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INVITED REVIEW

System-level optimal yield: increased value, less risk, improved stability, and better fisheries

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Abstract: The discipline and practice of fisheries science and management have had an useful, successful, and interesting history. The discipline has developed over the past century and a half into a very reductionist, highly quantitative, socially impactful endeavor. Yet given our collective successes in this field, some notable challenges remain. To address these challenges, many have proposed ecosystem-based fisheries management that takes a more systematic approach to the management of these living marine resources. Here I describe systems theory and associated constructs underlying system dynamics, elucidate how aggregate properties of systems can and have been used, contextualize these aggregate features relative to optimal yield, and note how this approach can produce useful estimates and outcomes for fisheries management. I explore two contrasting examples where this approach has and has not been considered, highlighting the benefits of applying such an approach. I conclude by discussing ways in which we might move forward with a portfolio approach for both the discipline and practice of fisheries science and management.

Résumé : La discipline et la pratique des sciences halieutiques et de la gestion des pêches ont une histoire utile, intéressante et couronnée de succès. La discipline est devenue, au cours du dernier siècle et demi, une entreprise très réductionniste, hautement quantitative et à forte incidence sociale. Malgré nos réussites collectives dans ce domaine, des défis notables demeurent. Pour relever ces défis, de nombreuses personnes ont proposé une gestion écosystémique des pêches qui adopte une approche plus systématique de gestion de ces ressources marines vivantes. Je décris la théorie des systèmes et les constructions associées qui sous-tendent la dynamique des systèmes, j'explique comment les propriétés groupées de systèmes peuvent être utilisées et l'ont été, je mets ces éléments groupés dans un contexte de rendement optimal et je souligne comment cette approche peut produire des estimations et des résultats utiles pour la gestion des pêches. J'explore deux exemples dans lesquels cette approche a, d'une part, été prise en considération et, d'autre part, ne l'a pas été, afin de souligner les avantages de son application. Je conclus en discutant d'avenues possibles pour l'avenir qui reposent sur une approche de portefeuille tant pour la discipline que pour la pratique des sciences halieutiques et de la gestion des pêches. [Traduit par la Rédaction]

Introduction

There have been notable successes in fisheries science and management, such that now many parts of the world have well-managed fisheries and reasonably healthy fish stocks (Pitcher et al. 2009; Hilborn et al. 2015), even though this is not globally true (Pitcher et al. 2009; FAO 2016). Yet whether stocks are poorly or well managed, several other challenges to marine fisheries persist that have long been well chronicled (e.g., Botsford et al. 1997; Micheli 1999; Jackson et al. 2001; Pauly et al. 2002, 2003; Pikitch et al. 2004; Link 2010; Fogarty 2014). The list of challenges often involves the need to more clearly consider the broader impact of fisheries — on other biota, habitat and socioeconomic systems — as well as the impacts of those broader considerations on fisheries, and all that in a systematic, comprehensive manner. In the context of improving the status of fisheries and fish stocks — in both poorly and well-managed situations — numerous calls for ecosystem-based fisheries management (EBFM) have arisen (Garcia et al. 2003; Pikitch et al. 2004; Garcia and Cochrane 2005; Levin and Lubchenco 2008; Link 2010; Fogarty 2014). For "poorly managed stocks", providing additional information has the potential to better elucidate those stocks, their dynamics, and their status, and any such additional information will be an improvement over the status quo. It also makes sense that we see this evolution towards exploring ecosystem approaches for many "well-managed stocks" in that there are diminishing returns to greater investment (i.e., model and data improvements) in single-species assessments and management, whereas the potential benefit of incorporating ecosystem information can and has provided substantial returns on fisheries investments.

The call to execute EBFM is not novel. Over 150 years ago Baird (1873) noted factors and facets that resonate with themes noted in the modern vernacular of EBFM (Link 2010). Over time considerable debate has occurred on the topic, usually centered around the relative importance of density-dependent versus -independent factors and usually sharpened as the relative importance of fishing versus environmental or other exogenous (to the fish stock) factors that affect stock dynamics (Browman and Stergiou 2004, 2005). Events in the 1940s such as the Thompson–Burkenroad debate (cf. Skud 1975; Smith 1994), similar discourse again in the 1950s–1960s ecological literature (Andrewartha and Birch 1954; Hairston et al. 1960), and continuing to this day (Walters and Collie 1988; Rose 2000; Punt et al. 2014; Szuwalski et al. 2015) all represent snapshots of the overarching debate regarding the prominence of internal versus external factors that shape the dynamics of fish stocks. In my view, the debate has become somewhat artificial and has remained singularly philosophical rather than having pragmatic solutions — of course both density-dependent and -independent factors influence fish stocks; of course both fishing and the environment influence fish stocks.

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And of course the relative importance of each varies under any given set of conditions. The argument against a broader inclusion of environmental or socioeconomic or density-independent elements really centers on the following:

- the analytical level of emphasis (single stock, multispecies, aggregate group, entire fisheries, or entire fishing and ecological system);
- whether including other factors in a model makes a significant difference in statistical performance or outcomes of models and management advice based thereon (Burgess et al. 2017);
- whether we will ever understand the ocean enough to model functional forms of all salient processes;
- whether governance institutions are structured to address trade-offs across stocks; or
- concerns over limited data.

The issues of data and governance structure remain a legitimate concern, although perhaps less so than is typically assumed (Patrick and Link 2015*a*). I would argue that in some instances these broader considerations are not needed to manage some fisheries, albeit trade-offs in objectives among and across fisheries would still remain. I would also argue that the other objections to broader inclusion of ecosystem considerations are also valid in certain situations, but there are in fact solutions around those concerns. A systems approach could be one such solution to address many of them.

Considering a systems approach is not novel in fisheries (Walters 1971, 1980; Walters and Hilborn 1976; Allen and McGlade 1987) and seems to be noted at regular intervals (Charles 2001; Apollonio 2002, 2010; Mahon et al. 2008; Fogarty et al. 2016), but it is also not widespread. Systems-level thinking examines processes in a more integrated, comprehensive matter. Doing so provides a plethora of emergent properties, cybernetics, and related information used in a wide range of applications. The challenge is that some of the information that emerges from systems analysis is not entirely intuitive or always easily interpretable, but advances in the systems analytic discipline over the past several decades have increased the familiarity and ease of interpretation of these outputs (Adams 2015). In essence, a systems approach acknowledges that there are myriad processes, connections, and "components" in any given system and that combined these are both highly complex and interactive. Yet a systems approach also emphasizes the collective whole and emergent features therefrom by examining the system at a higher hierarchical level of organization. If one can overcome historical precedence and associated objections of not managing fisheries on a stock-by-stock basis, a systems approach, and its associated analyses and perspectives, can offer solutions to the concerns identified in the ecosystem considerations debate. A systems approach similarly offers some promise towards implementing EBFM, thereby potentially improving the management of fisheries by addressing the concerns it seeks to ameliorate.

Optimal yield (OY) has been a prominent part of fisheries science and management. These discussions typically revolve around both how to define it and how to achieve it. Often associated with maximum sustainable yield (MSY) and related proxies (Restrepo et al. 1998; Rindorf et al. 2017*a*), the concept of yield has engendered considerable debate in the discipline (Schaefer 1954; Larkin 1977; Sissenwine 1978; Mace 2001; Hilborn 2010; Legovic et al. 2010; Finley 2011; Finley and Oreskes 2013; Rindorf et al. 2017*a*). The debate over whether OY should be measured as MSY or something else aside, and the debate over the politics of how the science to support and implement it arose also aside, the simple fact is that we need to have some concept — or preferably some type of measure — of the amount of fish biomass that is harvestable at any given place and time (Mace 1994; Fogarty 2014), an estimate that is loosely associated with the productivity of the fishes involved and the carrying capacity of the ecosystem that limits such productivity (Graham and Edwards 1962; Gulland 1970; Ryther 1969; Iverson 1990; Pauly and Christensen 1995; Ware 2000; Friedland et al. 2012; Heath 2012; Fogarty 2014). Most countries with any fisheries jurisdiction at all have some form of OY espoused in their fisheries polices (Mardle et al. 2002; Marchal et al. 2009; Dichmont et al. 2010; Mesnil 2012; Patrick and Link 2015*b*; Rindorf et al. 2017*a*). The reality is that the concept of OY can be applied to a population, a guild of species, an entire biotic community, or a fishery or group of fisheries or even an entire ecosystem. Patrick and Link (2015*b*) explored this theme, to the point that many facets of OY can be recognized as the embodiment of EBFM.

Here I would like to explore systems-level thinking as it pertains to OY as one possible means to improve fisheries science and management in the context of EBFM. My assertions are that we can learn from other disciplines, that the theory behind systems thinking is well developed, just not widely familiar to the fisheries discipline, and that this approach may ultimately prove beneficial to fisheries. Here I provide a more detailed primer on the systems approach, followed by some policy context of OY, then end with case studies exploring the application of worked examples for this approach.

A brief primer on systems-level thinking

Systems theory

Given that marine ecosystems are highly complex, exhibit nonlinear interactions, rarely experience equilibria, are open systems, have a myriad of dynamics, drivers, and interactions operating at multiple scales, and function in environments that are hard to observe, it may be an understatement to say that they are difficult to understand and predict. Certainly several end-to-end models are attempting to capture the broad suite of dynamics that constitute marine ecosystems (e.g., Fulton and Link 2014; Fulton et al. 2011, 2014) to the point of forecasting some scenarios, of which some models have been employed with a modicum of success (Fulton et al. 2014). But those models are understood to be representative of general, strategic perspectives and not the detailed, tactical predictions that are routinely required in a fisheries context. Further, one of the chief criticisms of an ecosystem approach is that the complexity, and hence data and information needs, are too high, thereby making any such understanding or predictions essentially infeasible. It is in this context that a systems perspective (which in fact many marine ecosystem end-to-end models adopt; Wu and Marceau 2002) is potentially quite helpful.

Although I provide a cursory presentation on the topic of systems theory here, I acknowledge that it can be a dense subject for those outside of that field. As such I have attempted to avoid both jargon-specific terms and highly mathematical representations associated with systems theory. There are copious, very detailed books on the topic that present those aspects in much more specificity (e.g., Axelrod and Cohen 2001; Weinberg 2001; Jackson 2003; Meadows 2008; Luhmann 2013). As an intermediate step, a reader interested in more details might find several primers on systems thinking and related topics helpful (e.g., Allen 2009; Mele et al. 2010; Wu 2013).

Systems science is the interdisciplinary study of systems in general, aiming to elucidate patterns and principles that can be discerned from and applied to all types of systems. (The terms systems science, thinking, theory, or approach are often used synonymously, although there are minor distinctions among them. For the purposes here, these are not distinguished.) A system is broadly defined as a collection of objects tied together by some form of interaction or interdependence. Ludwig von Bertalanffy, a name that should be familiar to most fisheries scientists from his commonly used growth curves (von Bertalanffy 1938), is widely recognized as one of the fathers of systems thinking. His work on general systems theory (von Bertalanffy 1968; cf. Luhmann 2013; Capra and Luisi 2014) was presented as a response to counteract the increasing reductionism he saw in the sciences, recognizing a need for and value of taking a holistic, or systematic, view. Rather than reducing an object to its component parts, systems theory focuses on the ordering of, and relationships between, the parts that connect them into a whole. Thus, a systems perspective emphasizes holism and argues that it is difficult to fully comprehend an object simply by breaking it into its component parts and then reconstituting it. In other words, even if one starts from the analysis of the component parts of an object, to fully comprehend the object completely one needs to also observe it from a higher level of organization (i.e., a holistic perspective; von Bertalanffy 1968). Systems theory does not ignore components; rather, it stresses the interrelations between components. Thus, a fundamental feature of systems theory is a focus on interactions. Given this emphasis, an important feature of general systems theory is complexity.

Complex adaptive systems

Coupled social–ecological systems, such as management of marine ecosystems and their component fisheries stocks, continue to defy reductionism. The number of components in such a system is sufficiently large that the usual treatments and representations thereof (i.e., a series of differential equations, mechanistic characterization of all salient processes, identification of appropriate functional relationships, etc.) become nearly impractical, as we tend to see in some ecosystem models. Further, such representations reach an effective limit in our ability to understand the system at this level of detail, particularly because the components dynamically interact. Systems like this, such as fishing in marine ecosystems, are known as complex adaptive systems (or more simply known as complex systems; both are used interchangeably here). Complex adaptive systems are defined as a complex, self-similar collection of interacting, adaptive components able to self-organize (Levin 1998, 2003; Lansing 2003; Levin et al. 2013). A complex adaptive system is a system in which even a perfect understanding of the individual components does not automatically convey a perfect understanding of the whole system and its behavior (Levin 1998, 1999; Jørgensen and Müller 2000; Miller and Page 2007). For example, even within an individual organism such as a fish, it is clear that highly detailed understanding of genes, biochemistry, and metabolic processes is not nearly sufficient to understand the complexity of a cell, much less the behavior of an entire organism.

The features of complex adaptive systems are well chronicled, but warrant summary here. Such systems exhibit adaptability, high numbers of connections and interactions, a preponderance of nonlinear dynamics, openness (i.e., difficulty in defining boundaries), lack of equilibria, and hysteresis (Cilliers 1998; Levin 1998, 1999). Complex adaptive systems can also be characterized by emergent properties, multiscale interactions, unexpected behaviors, and self-organization (Jørgensen 1995, 1997; Levin 1999; Wu 1999; Nielsen and Müller 2000). These characteristics of complex adaptive systems can result in challenges for understanding and prediction. The overall behavior of the system is hard if not impossible to predict by the behavior of the individual components. Yet a consequence of these properties of complex adaptive systems is that they result in a high degree of adaptive capacity, generally increasing resilience to perturbation (Jørgensen 1995, 1997; Jørgensen and Müller 2000; discussed further below). There are other benefits of complex system properties. Another value in examining complex systems comes from a focus on emergent properties of the system (any unique property that "emerges" when component objects are joined together in constraining relations to constitute a higher-level aggregate object, a novel property that unpredictably comes from a combination of two or more simpler components; Jørgensen 1997; Jørgensen and Müller 2000). Emergent properties are a key outcome from systems-level thinking applied to complex systems, with higher levels having emergent properties not exhibited by lower-level subsystems or components. It is the existence of emergent properties that allows us to better understand systems in a hierarchical manner.

Hierarchy theory

Hierarchy theory is a subset of systems theory. Hierarchy theory focuses upon levels of organization, issues of scale, and the importance of observation (Simon 1962; O'Neill et al. 1986; Wu 2013). This theory essentially decomposes a system into subsystems resulting in a hierarchical structure, capitalizing on emergent features at successively higher levels of a hierarchy to explain overall system and component dynamics. It is an important means to deal with complexity in systems. This quote from Wu (2013) summarizes nicely why a hierarchical approach has utility:

> "Complex systems are perceived by people as complex because their large number of interacting components resists easy description and understanding. Then, how do we approach such systems? One approach would be to treat them as "black boxes" — try to understand them by knocking on their walls and corners from the outside and then interpreting their responses without knowing anything inside. This would be an extremely holistic approach, which has proven to be of limited value. Another approach would be to treat them as nothing but the sum of their parts — an extreme reductionist perspective.… If the complex world is hierarchically or modularly structured, which seems true in many situations, none of the above mentioned approaches should work. In these cases, hierarchy theory has proven useful and effective."

A hierarchical system is composed of multiple levels, each consisting of one or more components or subsystems. Simon (1962) defines hierarchy in a systems context as "a system that is composed of interrelated subsystems, each of the latter being, in turn, hierarchic in structure until we reach some lowest level of elementary subsystem." Key elements of hierarchy theory emphasize that hierarchy is indeed repeatedly observed as a central facet of complex systems, levels tend to operate at the same rates, and there is flexibility (i.e., not tight, hard-wired coupling) among most system components (Simon 1962; Wu 2013). Allen (2009) further posits some principles of ecological hierarchy, chief of which are that higher levels operate more slowly and at a lower frequency than lower levels and that higher levels exert constraints on lower levels, which is routinely confirmed in observations (Wu 2013). That is, higher levels tend to be slower in time and larger in size, whereas lower levels tend to be faster in time and smaller in size (O'Neill et al. 1986, 1991; Urban et al. 1987; Wu 1999).

Another important aspect of hierarchy theory is the importance of an observer's role in understanding complex systems. The focal level at which an observer examines a hierarchical system strongly influences what is to be observed, how the system is perceived, and what properties emerge (Allen et al. 1984). Hierarchy theory often emphasizes the importance of the role of the observer such that it is sometimes known as "observation theory" (Allen and Starr 1982; Allen et al. 1984; Ahl and Allen 1996; Wu 1999, 2013). It is this feature of "hierarchical-ness" that on the one hand harkens caution in what and how a system is observed, but on the other hand lends to flexibility and opportunity given the concept of emergent properties at succeeding hierarchical levels. In other words, the specific viewpoint of one observing a system is critical to know, but allows one to explore facets of the system at levels above and below the level of observation.

Combining the observation about different rates at different levels in a hierarchy with the importance of the particular level of a hierarchy at which one is observing a system, a critical feature emerges. Because hierarchical systems generally consist of components that are loosely coupled (i.e., flexible in their interactions and connectivity), they are deemed near-decomposable. Near-decomposability is a core property of hierarchical systems and essentially conveys the observation that rates of interaction within components or subsystems at any level are much higher than rates of interaction between components and certainly between levels (Simon 1962, 1973). Full-decomposability occurs when system components are entirely decoupled from each other, which is not the case for complex systems. If the components are strongly coupled, the system cannot be "decomposed", and then its description requires consideration of all components. It is this near-decomposability that allows for the simplification necessary to describe and comprehend complexity (Simon 1962, 1973, 1976, 1996; Wu 1999, 2013). In other words, to describe the dynamics of a hierarchical system parsimoniously and adequately, again quoting Wu (2013), one would

> "…select a focal level, treat slow behaviors at the higher levels *[effectively]* as constants and fast behaviors at the lower levels as averages or equilibrium values. For a specific problem, it is not only possible but also wise to "scale off" the relevant levels from those above and below, thus achieving a greater simplification and better understanding (Simon 1962, 1973, 1996)."

The fact that most complex systems have a nearly decomposable, hierarchic structure is a major factor enabling understanding and description of systems and their components.

Simon and Ando (1961) describe how to utilize the near-decomposability of hierarchical systems into schemes of aggregation. This effectively allows one to analyze, measure, and quantify important facets of complex systems, particularly emergent properties. In doing so, the aggregative properties of higher hierarchical levels are used to describe the interactions of both that focal level and components at lower levels. In essence this allows one to reduce the complexity of complex hierarchical systems by combining interactions into aggregates of the components, thereby facilitating better description of a system.

Hierarchy theory has been applied to other complex systems for decades (e.g., Pattee 1973; Rosen 1969; Simon 1962). Hierarchy theory has similarly received much attention in the ecological literature (e.g., Allen and Starr 1982; O'Neill et al. 1986; Ahl and Allen 1996), but much less so in the fisheries literature (yet cf. Apollonio 2002, 2010). Even though hierarchy theory is not widely recognized within fisheries, it can offer both insights into the structure and function of marine ecosystems (e.g., Allen and Starr 1982; O'Neill et al. 1986; Ahl and Allen 1996) and potential utility in dealing with the complex dynamics of fish stocks and their management, particularly as these are coupled social–ecological systems (Apollonio 2002, 2010, 2015). Taking the concept of near-decomposability from hierarchy theory can result in at least one pragmatic approach to better understand, predict, and even inform management of marine ecosystems and fisheries.

The portfolio effect

Thus far we've established that marine ecosystems and the fisheries associated with them are near-decomposable, hierarchal, complex, adaptive systems with emergent properties. So what? Recall that by taking advantage of the feature of near-decomposability found in these hierarchical systems, one can aggregate components at different levels of a system and then quantify salient emergent properties of that system (Simon and Ando 1961; Simon 1962; Jørgensen 1995, 1997; Nielsen and Müller 2000; Wu 2013). This ability, due to near-decomposability coupled with the similarity of rates at a given hierarchical level (identified by Allen 2009), has quite literally been capitalized on, formally, for over 70 years in financial systems (Markowitz 1952). Especially with respect to the specific properties of aggregate and individual stock value, aggregate stability and variance, and distributed risk, these features form what is known as a portfolio. A portfolio is a specific application of hierarchy theory applied to complex systems.

Represented pseudo-mathematically, a portfolio seeks to find aggregate solutions, whereby

$$\text{(1)}\quad \text{obj } f(\Sigma x) \cap \text{Min}[\text{Risk}(\Sigma x)], \text{Min}[\text{Risk}(x)], \text{Min}[\text{Var}(\Sigma x)], \text{Max}[\text{Stability}(\Sigma x)], \text{Max}[\text{Value}(\Sigma x)], \text{Max}[\text{Value}(x)], \text{Max}[\text{Productivity}(\Sigma x)], \text{Max}[\text{Yield}(\Sigma x)], \text{Min}\{\text{Effort}[\text{Max}(\Sigma x)]\}$$

where x is any asset (in this case, a fish stock), and Σx is the aggregate sum of said assets. The portfolio approach is a specific application of near-decomposable properties of a hierarchical, complex system of assets, where an asset can be one of any number of things, synonymous with the term "components" noted above. This has resulted in the proliferation of aggregate products associated with financial markets. Given the main objectives of a portfolio (eq. 1) that capitalize on hierarchy theory, what essentially results is that one seeks to arrange an aggregate group of assets (i.e., diversify) to stabilize and maximize their collective performance, known as reaching the efficiency frontier of a portfolio. This is done cognizant of the level of risk facing the portfolio; in a fisheries context that would be primarily to not have any stock overfished. The dynamics of this aggregate level are almost always much less variable than that of the individual components (Simon and Ando 1961; Booth and Fama 1992; Brown et al. 2016). It is well known in financial contexts that more diversified portfolios are less volatile (e.g., Markowitz 1952; Brigham and Gapenski 1988; Elton and Gruber 1977, 1995; Choueifaty and Coignard 2008; Fabozzi and Markowitz 2011; Marston 2011). A key reason why more aggregate levels may be more stable than individual components is because of the effects of statistical averaging (e.g., discussed in an ecological context; Doak et al. 1998). Resisting the urge to replicate equations of covariance that seem replete in these discussions (Doak et al. 1998; Tilman et al. 1998; Tilman 1999; Halpern et al. 2011; Sethi 2010; Anderson et al. 2013; Thibault and Connolly 2013), essentially what happens is the collective variance of a set of assets gets dampened via the addition of covariance terms, spreading the variability across more assets. Under many circumstances, in both financial (Markowitz 1952; Elton and Gruber 1977, 1995) and ecological (Tilman et al. 1998; Schindler et al. 2015) contexts, the sum of several randomly and independently varying assets is less variable than the average asset. This is known as the portfolio effect.

Diversification and portfolio theory rely on two main features: statistical averaging and correlations among the component stocks or assets. Substantial debate has occurred regarding the exact and precise form of this representation (Doak et al. 1998; Tilman et al. 1998; Tilman 1999; Cottingham et al. 2001; Thibault and Connolly 2013) and whether the portfolio effect is solely a result of mathematical combination of hierarchical assets (there is no doubt that by structure at least mathematically it has to contribute to some of the portfolio effect) or whether properties of the individual components and system dynamics also contribute to the spread of risk and dampened variability. Besides statistical averaging, other features have been identified as important determinants of the portfolio effect. These include asynchrony in asset fluctuations, weighting of the different components of the portfolio, and diversity of assets relative to total aggregate-level values (Tilman et al. 1998; Cottingham et al. 2001; Anderson et al. 2013). The general conclusion is that statistical averaging is an important facet driving the portfolio performance, but that other considerations can and do amplify it.

The portfolio effect has been utilized in financial markets for a long time (Markowitz 1952; Roy 1952). The dynamics of many ecological and biological systems are also almost always less variable than the individual components they are composed of, similarly exhibiting portfolio effects. This concept is well represented in the ecological literature (e.g., Waltho and Kolasa 1994; Tilman et al. 1998; Thibault and Connolly 2013; Schindler et al. 2015; Brown et al. 2016), particularly related to biodiversity. The concept has had some attention in the fisheries literature (Edwards et al. 2004; Rădulescu et al. 2010; Schindler et al. 2010; Yang 2011; DuFour et al. 2015; Jin et al. 2016), but the application has not been widely implemented. The theory and mathematics have been explored in a fisheries context, often with particular respect to EBFM (Sanchirico et al. 2008; Jin et al. 2016), such that the discipline is at the point where these academic examples could start moving into applied, operational management practice.

The salient point of the portfolio effect in a marine fisheries and ecosystem context is this: how would one like to manage fisheries in marine ecosystems such that

- risk of overfishing is minimized (i.e., above agreed upon levels or at least not substantially above natural levels);
- populations of fishes, catches, and profits are stable;
- overall value across all stocks is maximized;
- bureaucratic oversight and regulatory interventions are minimized;
- catch and yield are optimized;
- biomass of the resource (in aggregate) is maximized;
- stakeholder disenfranchisement and legal challenges are minimized;
- catch per unit effort is optimized; and
- risk of ancillary ecosystem impacts are minimized (again, above agreed upon levels or at least not substantially above natural levels)?

That may sound like an overstatement or even "pollyanna-ish", but if even close to true it is a statement that every fishery manager needs to underline, muse upon, and strongly consider. In cases where worked examples have been examined (Edwards et al. 2004; Sanchirico et al. 2008; Jin et al. 2016), categorically the benefits of this application of a systems approach are always predicted to be greater than what was actually realized from the actualized blend of the portfolio relative to the portfolio efficiency frontier (Sanchirico et al. 2008; Rădulescu et al. 2010). This is in terms of biomass of the stock, value and profit to the fleet, and risk to the stock and fishery, which were all suboptimal compared with what could have been obtained (Edwards et al. 2004; Sanchirico et al. 2008). That is, by focusing solely on the component stocks in an ecosystem instead of an aggregate group as part of a complex, hierarchical system and by ignoring the properties of marine ecosystems that result in a portfolio effect, collectively we have left profit on the table, lost productivity, and exercised undue risk.

Given the demonstrable and predictable benefits of the portfolio effect, that its theory and mathematics have been worked out, and that it has clear application in a marine fisheries context, let me raise two questions. Is the approach applicable and consistent with OY considerations, and are there broad-scale, long-term examples that can more fully and clearly demonstrate the utility of the approach?

Addressing pragmatic considerations of system-level thinking for OY

The theoretical construct for systems thinking (and the particular instance of it in the portfolio effect) has a solid basis. There are also several practical implications from that theory. But do those have appropriate application in a fisheries context that is typically required, in one form or another, to consider OY? I would argue that there are at least three considerations that should serve as criteria to answer that question: Is it legal? Is it estimable? Is it advisable?

Is it legal? In all jurisdictions, certainly it is not illegal. That is, what fisheries mandate or manager would be opposed to the elements noted in eq. 1? What is less clear is not if a systems approach is allowable, but if it is called for. In the US context, Patrick and Link (2015*a*, 2015*b*) looked at the Magnuson–Stevens Act (Magnuson–Steven Fishery Conservation and Management Act (MSA) 1976; (16 USC 1801, et seq.)) and related National Standard guidelines (Darcy and Matlock 1999; Methot et al. 2014) to see how OY and EBFM are related. What they and others have found is that, in fact, at least under US fisheries law, the need for OY and associated management measures is not legally mandated on a stock-by-stock basis, but rather mandated on a fishery basis. In practice OY has played out as being applied on a stock-by-stock basis, but it does not have to be done that way. Several jurisdictions are exploring or have proposed a systems approach to broaden the stock-specific application of OY (e.g., Mueter and Megrey 2006; Fogarty et al. 2012; Fogarty 2014; Rosenberg et al. 2014; Gaichas et al. 2017; Rindorf et al. 2017*a*, 2017*b*). When I advise governments and engage with counterparts in countries all around the world, for example, Norway, Canada, Australia, South Africa, New Zealand, Japan, the EU, Iceland, etc. (cf. Mardle et al. 2002; Marchal et al. 2009; Dichmont et al. 2010; Mesnil 2012; Patrick and Link 2015*b*; Rindorf et al. 2017*a*), and broach the topic, not one has objected nor indicated that considering OY in aggregate is not allowed; rather, they typically tend to simply not think of it. This is also true in numerous intergovernmental organizations, such as the International Council for the Exploration of the Sea, Food and Agricultural Organization of the United Nations, or other regional fisheries management organisations — not one objects to aggregate OY on a legal basis. Rather, the concerns either center on technical capacity to estimate it or more so that these organizations are simply not thinking about this issue. Probably the most notable exception is the Convention for the Conservation on Antarctic Marine Living Resources, which has considered aggregate OY to the point of including facets of it in their krill harvest control rules (Constable et al. 2000; Constable 2001, 2011).

In the US context, there are provisions in the national standard guidelines and associated technical guidance (e.g., Darcy and Matlock 1999; Federal Register 2009; Methot et al. 2014) for using stock complexes (i.e., aggregated groups of stocks). Thus, doing an aggregate OY is clearly allowable. Yet, although there are no specific requirements for OY at a system level, increasingly there are recommendations for it. Apart from the provision in technical guidance for OY, the US National Marine Fisheries Service (NMFS) has multiple strategic documents that call for a systematic look at how fisheries yield is determined (and coordinated across stocks), including the updated Stock Assessment Improvement Plan, Stock Assessment Prioritization effort, NMFS Climate Science Strategy, Habitat Assessment Improvement Plan, and EBFM Road Map[1]. Additionally, it is NMFS policy (EBFM Policy[2]) to explore and begin to use system-level reference points. Thus, although not common nor always clear, adopting a systems approach to implement OY is legally defensible and being increasingly explored as preferred policy.

Is it estimable? Beyond philosophical considerations about whether we should even be measuring it or what that would entail, objections to estimating aggregate OY typically revolve around applying a reductionist approach to higher levels of aggregation

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[1] http://www.st.nmfs.noaa.gov/StockAssessment/; http://www.st.nmfs.noaa.gov/stock-assessment/stock-assessment-prioritization; http://www.st.nmfs.noaa.gov/ecosystems/climate/national-climate-strategy; https://www.st.nmfs.noaa.gov/ecosystems/habitat/publications/haip/index; https://www.st.nmfs.noaa.gov/ecosystems/ebfm/creating-an-ebfm-management-policy.

[2] http://www.nmfs.noaa.gov/op/pds/documents/01/01-120.pdf.

and as such are ignorant of the benefits of near-decomposability and associated facets of hierarchy theory. Certainly one can use complicated age- or length-based multispecies models, size spectra models, food web models, and even end-to-end ecosystem models to estimate aggregate OY (Christensen et al. 2009; Link 2010; Fulton and Link 2014; Fogarty 2014). In some instances where the data and modeling capacity are available to do so, executing these models to provide estimates of aggregate OY provides a useful comparison with other methods and affords the opportunity to test the robustness of model results in a multimodel ensemble (Townsend et al. 2014). Yet most of the approaches to estimating system-level or aggregate OY use relatively simple and straightforward forms of trophic transfer models or aggregated production models (Ware 2000; Fogarty et al. 2008, 2012; Rosenberg et al. 2014). Perhaps without even explicitly considering it, these latter approaches are taking advantage of the near-decomposability of hierarchical systems and emphasizing emergent properties found in aggregate measures. In essence, using widely available data common to fisheries (e.g., survey biomass indices, catch, etc.), one can execute commonly available models, simply ignore the stock or species delineations, combine the data into an aggregate group, and obtain model outputs for the group as a whole. Several theoretical, simulation, and contextual considerations have explored this approach (Gamble and Link 2009, 2012; Worm et al. 2009; Bundy et al. 2012; Fay et al. 2013, 2015; Fogarty 2014; Gaichas et al. 2017; Rosenberg et al. 2014; Rindorf et al. 2017*a*) and found the method performs as one would expect from hierarchy theory, namely that resulting model outputs are consistent with the portfolio effect (Link et al. 2012; Jin et al. 2016). Many works (e.g., Mueter and Megrey 2006; Fogarty et al. 2008, 2012; Christensen et al. 2009; Gaichas et al. 2012; Lucey et al. 2012; Rosenberg et al. 2014; Rindorf et al. 2017*a*, 2017*b*) have examined these approaches, tested the analytical methods, explored the robustness and vagaries of the results, and generally arrived at the conclusion that at the very least, the analytical methodologies and considerations are not a limitation for the adoption of this systems approach. More so, the results are generally repeatable, robust, and ecologically defensible. And finally these results conform to predictions consistent with that predicted from portfolio theory.

Is it advisable? To answer this question, one can compare and contrast simulations of a system-level versus a stock-by-stock-level approach to estimating OY and then evaluate in silico management performance at aggregate levels. Several instances where such simulation testing has occurred demonstrate improvements across a range of not only yield, but other fisheries objectives (Worm et al. 2009; Link et al. 2010; Fulton et al. 2011; Kaplan and Leonard 2012; Fay et al. 2013, 2015; Smith et al. 2015; Jacobsen et al. 2017). Additional contrasts have similarly elucidated the performance of OY estimates empirically, with a range of studies having similarly explored this issue (e.g., Ralston and Polovina 1982; Mueter and Megrey 2006; Fogarty et al. 2008, 2012; Worm et al. 2009; Sparholt and Cook 2010; Gaichas et al. 2012; Lucey et al. 2012; Rosenberg et al. 2014). A few common outcomes emerge from this collective body of work. One is that the aggregate OY, or more specifically estimates of aggregate MSY, are on average about 25% lower than summed estimates of single-stock MSY in a given system (Worm et al. 2009; Hilborn et al. 2012). Yet when one considers that OY is usually lowered from estimates of MSY, the distinction becomes rather small if not indistinguishable (Restrepo et al. 1998; Patrick and Link 2015*b*). Another common theme is that the risk of overfishing when managing at an aggregate OY level is much less than managing at component stocks' OYs (Worm et al. 2009; Link et al. 2012; Fogarty 2014). For small reductions in overall yield (5%–10%), the risk of overfishing any component stock is lessened in many instances by 50%–60% (Worm et al. 2009; Hilborn et al. 2012; Link et al. 2012; Fogarty 2014); the economic benefits from a constant revenue stream and not having to implement rebuilding measures likely outweighs any small, short-term reduction in yield (Worm et al. 2009; Hilborn et al. 2012; Fay et al. 2013, 2015; Kaplan and Leonard 2012). Additionally, other work has shown that ecosystem processes and functioning, and hence related OY, is achieved more efficiently when managed at the aggregate or systems level (Jacobsen et al. 2017). It has been repeatedly shown that maintaining all stocks in an ecosystem simultaneously at a single-species maximum sustainable or maximum economic yield is an impossibility due to biological and socioeconomic constraints (Au 1973; Crutchfield 1973; May 1975; Pope 1975, 1979; Fukuda 1976; Brown et al. 1976; Larkin 1977; May et al. 1979; Mayo et al. 1992; Collie and Gislason 2001; Walters et al. 2005; Mueter and Megrey 2006; Legovic et al. 2010; Legovic and Gecek 2011; Steele et al. 2011; Fogarty et al. 2012; Heath 2012; Link et al. 2012). Thus, given the energetic constraints of both species production and ecological interactions, coupled with some core tenets of portfolio and hierarchy theory, an aggregate approach is actually more theoretically defensible than attempting to simultaneously maximize all single component stocks or species yield in an entire ecosystem. In some instances, particularly such as in data-poor situations, highly diverse systems where species delineations are challenging, or where the risk or history of overfishing is high, the aggregate approach is preferable to accommodate those biological and informational limitations (sensu Gaichas et al. 2012; Link et al. 2012). The theory, simulations, empirical evidence, and contrast in performance all show that adopting an aggregate OY has notable benefits and in many instances is indeed advisable, if not even preferable.

The value and benefits of system-level OY — examples from two ecosystems

The application of systems thinking to marine fisheries ecosystems is appropriate and is not constrained by an OY context. But are there are any worked examples where this has been applied, perhaps contrasted in a similar instance where it has not, to document real-word, realized benefits from adopting such an approach?

The Eastern Bering Sea in Alaska and Georges Bank – Gulf of Maine in the Northeast US have both had a long and important history of fisheries (Witherell et al. 2000; Hollowed et al. 2011; and Fogarty and Murawski 1998; Link et al. 2011, respectively). They have broadly similar taxa, high levels of productivity, highly lucrative fisheries, and due to their rich history of scientific studies and long-term data collection, have been used in many comparative ecosystem analyses (e.g., Gaichas et al. 2009, 2012; Link et al. 2009; Megrey et al. 2009). There are many similarities in the fisheries context for both ecosystems (Table 1). Yet there are distinctions between the two regions. There are differences in oceanography, latitude, regional human populations, regional cultures, and fisheries management measures employed between the two regions. Certainly, the nuances of these factors contribute to the different overall value of the fishery in each ecosystem. Yet in a systems context, one such difference is that one fisheries management system has adopted key facets of an aggregate portfolio approach, and the other has not.

Utilizing information from national, publicly available databases, I examined total and individual stock landings (t), the value of those landings (in 2013 US dollars)[3], and the proportion of

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[3] https://www.st.nmfs.noaa.gov/commercial-fisheries/.

Table 1. Distinctions between ecosystems and fisheries in the Northeast regional and Alaskan regional fishery management systems.

	Northeast	Alaska
No. of managed species	~70	~70
No. of managed fishery species	39	32
No. of fishery management plans (FMPs)	14	7
Are there estimates of ecosystem (or aggregate)-level optimal yield (OY)?	Yes	Yes
Are ecosystem-level OY estimates used for management?	**No**	**Yes**
Are there Ecosystem Status Reports?	Yes	Yes
Primary fisheries management jurisdiction	RO, 2 FMCs[a]	RO, 1 FMC[a]
No. of other jurisdictions	4 organizations	3 organizations
% Change in no. of active vessels, 2010–2015	–5.8%	–6.8%
Change in no. of jobs (without imports)[b]		
% Difference in no. of jobs, 2010–2014	**–21.2%**	**+12.5%**
% Difference in no. of jobs, 2006–2014	**–23.6%**	**+51.5%**
No. of groundfish FMP amendments and framework adjustments, 2010–present[c]	**35**	**20**
No. of significant violations 2010–2015 (enforcement decisions and orders)[d]	14	8
Total no. of lawsuits filed against agency, 2010–2016[e]	**20**	**14**

Note: Sources of information are provided below; bold items are those which are notably different between the two regions (χ^2 test, $p < 0.05$).

[a] RO, regional office; FMC, fisheries management council.

[b] From FEUS in each year noted; https://www.st.nmfs.noaa.gov/economics/publications/feus/fisheries_economics_2014/index.

[c] NPFMC — https://www.npfmc.org/; NEFMC — http://www.nefmc.org/; MAFMC — http://www.mafmc.org/.

[d] http://www.gc.noaa.gov/enforce-office6.html.

[e] From NOAA Office of General Counsel database.

stocks that are overfished[4]. A couple of obvious things emerge from these data (Fig. 1). The first is that landings of the many individual stocks shown (as examples here) fluctuate quite dynamically (Figs. 1A, 1B), similar to the biomass of these taxa (EAP 2012; Zador 2015). And although the absolute amount of landings is higher in Alaska (reflective of the area fished and slightly higher primary productivity; Hollowed et al. 2011; Zador 2015), both sets of aggregate landings are relatively stable over the extent of the time series shown (Figs. 1A, 1B), consistent with systems thinking and hierarchy theory. What is most noteworthy is that in one region (New England), the fisheries are managed on a stock-by-stock basis. Whereas in the other region (Alaska), the stocks are managed with an aggregate cap on landings of groundfish; the 2 million metric ton total allowable catch in Alaska (Witherell et al. 2000) effectively serves as an aggregate OY in practice. Most other metrics are quite comparable (Table 1).

In both regions individual stocks are tracked. The constraints and emergent properties of full system dynamics are not taken into account in the stock-by-stock case, whereas these portfolio effects directly provide an overarching context in the other. In Alaska, when the proposed catches of all stocks combined is projected to exceed the limit of the aggregate portfolio OY, a process is then enacted where stock-oriented and fisheries-specific negotiations occur. These discussions proceed under a set of constraints whereby the catch limits for each stock in the portfolio is then lowered subject to an overall, 2 million metric ton cap on all groundfish (Witherell et al. 2000), no one stock is overfished, and economic impacts are (at least attempted to be) minimized. The result is a lowering of overall fishing rate on all stocks (inclusive of bycatch) while maintaining total catch and value for all the groundfish collectively in the portfolio. The overall portfolio cap and the stock-based overfishing constraints combine to provide the parameters of the "trade-space" as seen analogously in financial markets. In essence, the Alaska region uses the properties of complex systems and hierarchy theory by directing management at one hierarchical level above the level where the intended effects are desired. In New England, stocks are managed individually with no such portfolio considerations. There are certainly single-stock objectives, but none collectively for the entire set of fisheries. Thus, individual stock risk and productivity are certainly considered, but an overall strategy using a portfolio-defined trade space is not, thereby overlooking some of the cumulative and aggregate risk and production considerations. To be fair and complete in this description, the New England region has at times considered and is currently considering a portfolio approach, but has not operationally enacted such an approach (Fogarty 2014; M. Fogarty, personal communication).What are the outcomes of adopting this portfolio approach (in Alaska) or not (in New England)?

Figure 2 shows that in Alaska, the value of seminal groundfishes has steadily increased and the number of stocks that are overfished has remained low (<5%). Whereas in New England, the opposite occurred; the value of seminal groundfishes has remained relatively flat and low, and the number of stocks overfished is close to 30%–35%, with a peak over 40%. This implies that in the region that uses the aggregate estimate of OY as a hard cap for fisheries management (i.e., that adopts parts of a portfolio approach), taking advantage of near-decomposability, the risk of overfishing is lower and the long-term value is higher than the region that has not adopted such an approach.

In addition to capitalizing on the collective resource (i.e., all fish stocks) stability inherent in aggregate, emergent features of higher hierarchical levels, the region that adopted the aggregate approach also accrued other benefits. There was regulatory stability, as seen by both the fewer number of fishery management plans and amendments to those plans (Table 1). There is economic stability occurring, with greater value (Fig. 2) in the region that adopted an aggregate OY approach. Being able to predict, plan for, and count on there being a consistent amount of fish available to the fleet contributes to that stability. Additionally, there is greater fisheries stability as seen by lower turnover not in the number of fishing vessels (changes thereto are about equal; Table 1), but rather in the stark difference in the number of jobs over time. This has an implied lower risk to fishing communities that are experiencing other social changes (Jepson and Colburn 2013; Himes-Cornell and Kasperski 2016). There is also generally greater stakeholder buy-in to the management decisions, as seen by the fewer number of lawsuits against the NMFS and fewer number of significant violations (Table 1). Where there were no-

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[4] https://www.st.nmfs.noaa.gov/sisPortal/.

Fig. 1. Total landings (metric tons) of major, exemplary fished taxa in the New England region (A) and the Alaska region (B), with total landings of all species in each region.

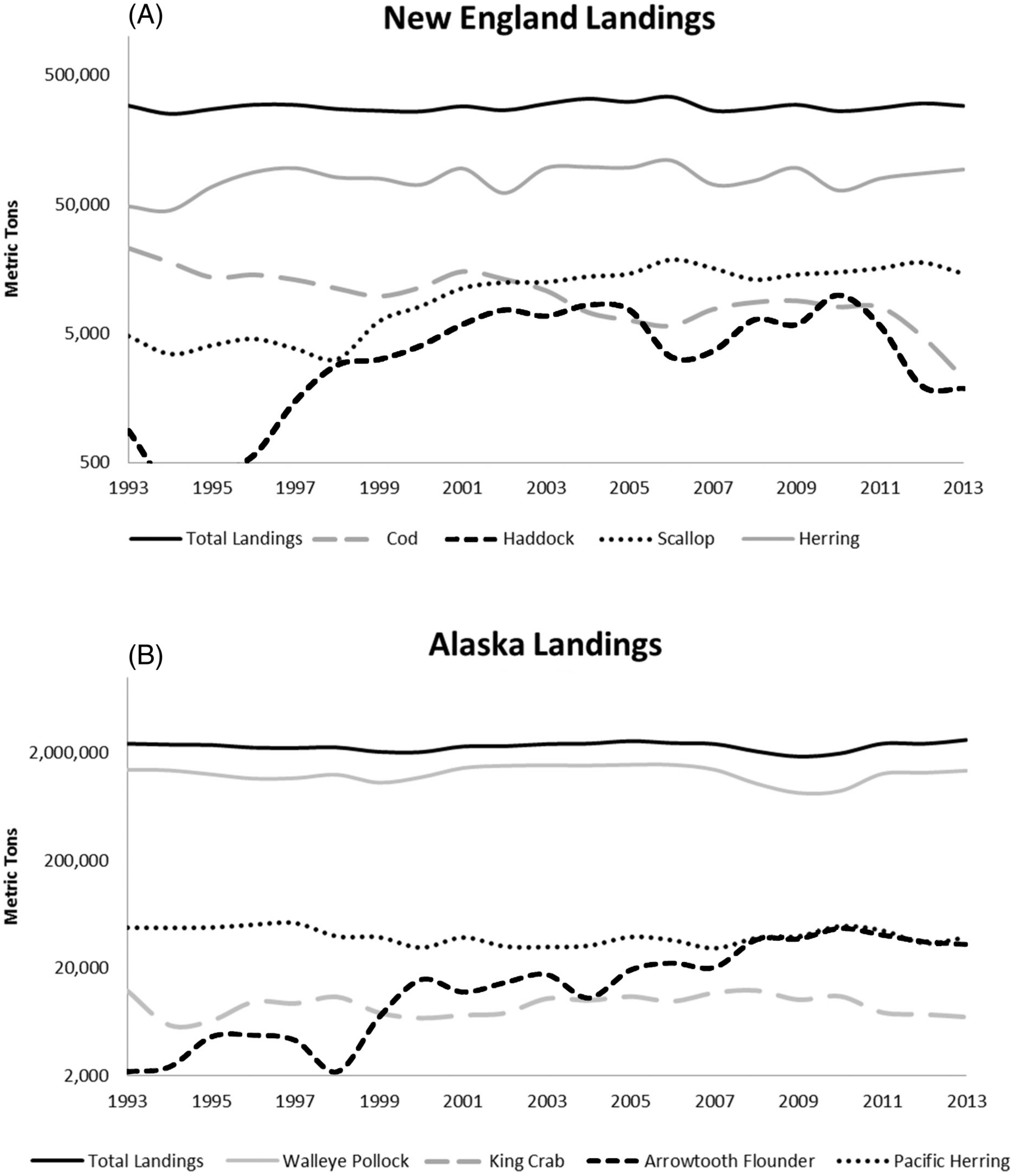
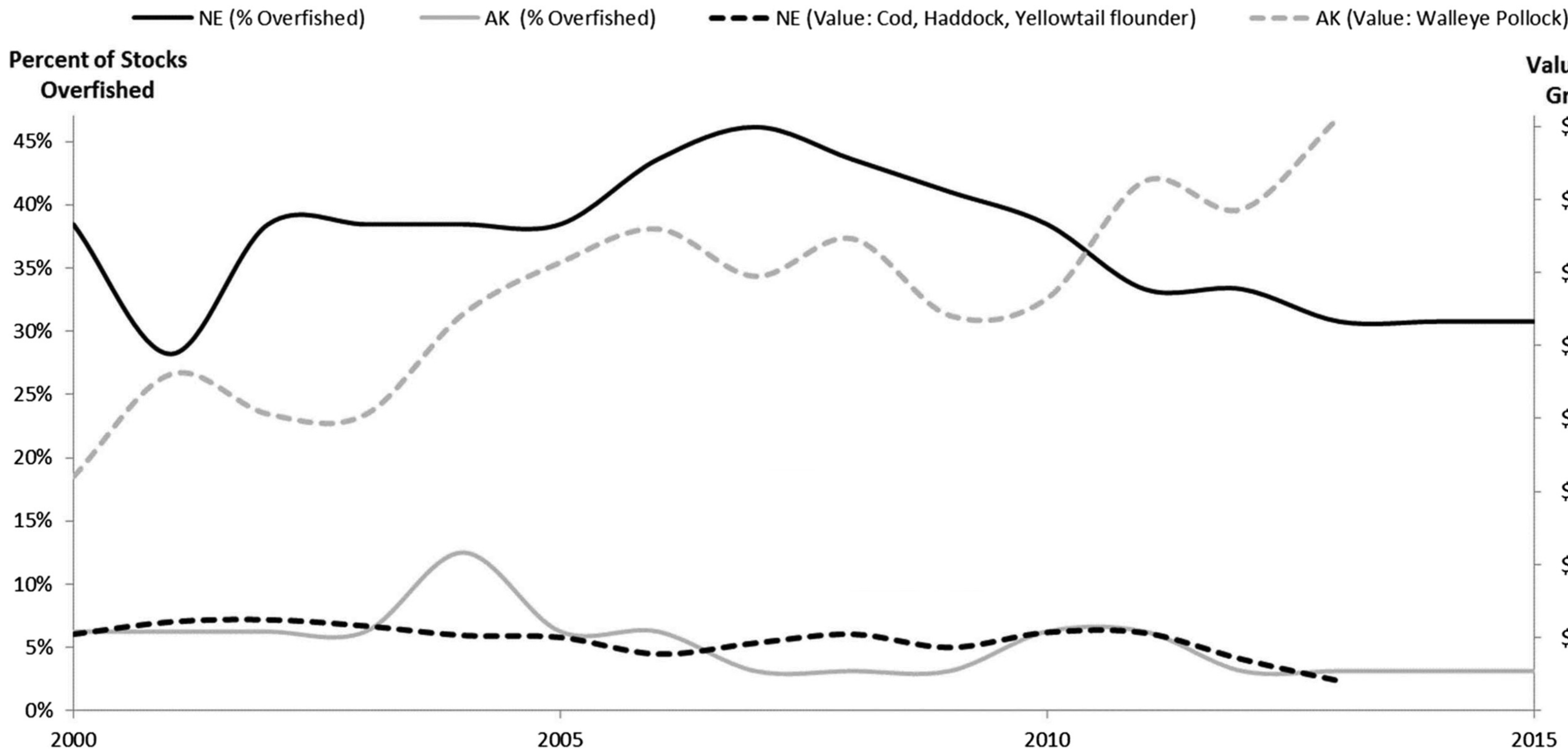


table differences, categorically the opposite responses were observed in the region that did not adopt an aggregate OY approach and stuck to a focal emphasis at lower levels of the hierarchical system. In sum, the region that adopted an aggregate OY approach, essentially managing for a portfolio rather than individual, component stocks, took advantage of the inherent trade-offs and true constraints imposed by both ecology and properties of hierarchical systems and, as such, reaped clear benefits.

Certainly there are caveats for each region. Alaska (and vessels from other ports fishing there) has a smaller human population, has a relatively newer fisheries, has fewer permit-holders and fishermen involved in the fishery, was an earlier adopter of catch limits and catch shares, and has had major contributions to the overall landed biomass dominated by one species, the walleye pollock (*Gadus chalcogrammus*; Hollowed et al. 2011; Zador 2015). New England has a larger human population, longer history of industrial fishing, greater number of permit-holders and fishermen, emphasized effort controls rather than catch limits as a management measure (although that has been equalized since the 2006 reauthorization of the MSA that required annual catch limits), and has had a diversity of species dominating the overall catch over time. There are more distinctions noted elsewhere (Fogarty and Murawski 1998; Gaichas et al. 2009, 2012; Link et al. 2009; Megrey et al. 2009; Link et al. 2011), and certainly these could

Fig. 2. Time series of the percentage of overfished stocks (solid line, left axis) and value (in 2013 US dollars for seminal groundfish species; dashed line, right axis) from the New England (black) and Alaska (grey) regions.



have been factors at least partially accounting for the different outcomes we have observed. But many other regions in the US could have easily substituted for New England with similar outcomes; I chose to contrast New England with Alaska because there were the greatest number of similarities in those two regions across a range of factors, which better facilitated comparison (Table 1; Megrey et al. 2009). The other regions would all have exhibited similar patterns as New England. The main distinction between these other regions and Alaska would not have been the number of stocks managed or stocks fished, governance structures, productivity of the ecosystem, human population density, or number of FMPs; the most obvious distinction between them and Alaska is that none of the other regions manage their fisheries using an aggregate OY, portfolio approach.

Discussion

Here the application of systems thinking applied as the portfolio approach has been theoretically established, predicted key outcomes, and been shown to have demonstrable benefits consistent with theory and predictions in real-world situations. As a relatively unknown or at least underutilized approach in fisheries, it holds promise for the implementation of EBFM to better ameliorate the many challenges facing the discipline and practice of fisheries management.

Certainly there are caveats to interpreting, understanding, and adopting a portfolio approach in fisheries. Obviously, specific management measures, other constraints due to protected species or habitats, other ocean-use activities, and related facets of fisheries and ocean governance structures need to be considered. Additionally, the size and capacity of fleets, the value of what is landed, the broader economic context, the number of individuals participating in a fishery, and related human dimensions are also important. And certainly oceanographies, climatologies, biodiversities, component species, and related ecosystem dynamics can all influence outcomes predicted from a portfolio approach. Indeed, all these considerations varied in the two example regions shown and undoubtedly contributed to distinctions in fisheries performance observed between the two systems. I acknowledge that on the surface that may make these comparisons circumstantial. Certainly, formally executing Smith's "fishery autopsy" (Smith 1998; Smith and Link 2005) would rule out a range of competing hypotheses; however, in large part that was how this contrast was informally structured. Furthermore, of all the factors between these two and many other regions, the one major distinction is the Alaskan region utilized an aggregate OY. And the results were consistent with what is predicted from portfolio theory. Thus, given all these caveats, the value of using properties of complex adaptive systems such as these, and particularly embodied as the portfolio effect, is that they are flexible enough to accommodate such distinctions among components and still provide robust predictions. Thus, there will always be stock-specific considerations in a given ecosystem that has active fisheries that will need to be accommodated, but the portfolio approach is robust enough to accommodate them and still provide reasonable predictions and realized outcomes.

To be clear, by proposing a portfolio approach to achieve aggregate OY, I am not espousing the overfishing of component stocks. Just as in financial markets, the overall value of a portfolio increases when all stocks increase; that should be the goal. Similarly, in financial markets there are limits as to what the performance of a portfolio is relative to how much risk an investor is willing to tolerate. Thus, fishery portfolios can similarly be developed with constraints to avoid risks of overfishing. A major difference between financial and fisheries portfolios is that ultimately the health (and survival) of minor stock contributors to the portfolio is inconsequential in a financial context. Yet in fisheries, minor components contributing to the portfolio can be rare, but we simply cannot allow them to "exit the portfolio" existentially speaking, and in such instances we do need to enact some protection to minimize this risk to individual stocks. Furthermore, some rare components of fisheries portfolios can actually be quite valuable even if limited in abundance or biomass, thereby increasing fishing pressure on them. The set of risks are thus not entirely analogous, but the portfolio approach is actually designed to account for these constraints to minimize such risk.

Another important caveat here that may distinguish fisheries from financial or other portfolio analogues pertains to energetic constraints. It is well documented that one cannot maximize the yield of all fish stocks in a system simultaneously (Au 1973; Crutchfield 1973; May 1975; Pope 1975, 1979; Fukuda 1976; Brown et al. 1976; Larkin 1977; May et al. 1979; Mayo et al. 1992; Collie and Gislason 2001; Walters et al. 2005; Mueter and Megrey 2006; Legovic et al. 2010; Legovic and Gecek 2011; Steele et al. 2011; Fogarty et al. 2012; Heath 2012; Link et al. 2012). Beyond all the fisheries-related debates over this point, this is simply a matter of the second law of thermodynamics and related applications of ecological-mass balance (Jørgensen 1995, 1997; Jørgensen and Müller 2000). Thus, fisheries portfolios need to be constructed with recognition of the constraints placed on multiple stock maximization that is set by ecosystem productivity. The proposal by Fogarty (2014) to embody these constraints as "ceilings and floors" in a fishery portfolio is certainly feasible, can help define the portfolio frontier, has been explored in a fisheries portfolio context (Edwards et al. 2004; Sanchirico et al. 2008; Rădulescu et al. 2010; Schindler et al. 2010; Yang 2011; DuFour et al. 2015; Jin et al. 2016), has been proposed for ecological portfolios addressing biodiversity (Tilman et al. 1998; Thibault and Connolly 2013; Schindler et al. 2015; Brown et al. 2016), and has actually been routinely adopted in financial portfolio approaches (Markowitz 1952; Brigham and Gapenski 1988; Elton and Gruber 1977, 1995; Choueifaty and Coignard 2008; Fabozzi and Markowitz 2011; Marston 2011). This portfolio cap constraint, coupled with the overfishing risk constraints, provides the useful trade space that is needed in many fisheries negotiations, helping to focus the debate on wise and equitable allocations among the stocks.

Another important distinction of the portfolio effect evinced in fisheries as compared with financial markets is the relative weightings of component stocks. In the one example shown, there is notable stability of a major component stock (i.e., walleye pollock in Alaska), which largely dominated the portfolio, whereas there is a broader diversity of catch in the New England region. In part that is due to the vagaries of the Bering Sea ecosystem, with the regime shift from invertebrates to conditions more suitable to walleye pollock well-chronicled (Hare and Mantua 2000; Hollowed et al. 2001, 2011). The New England region currently is likely undergoing such changes[5] (cf. EAP 2012; Pershing et al. 2015), with the outcome less clear. Thus, the distinction here is that in some ways, unlike financial markets, the conditions and health of stocks that lead to choices of stock-weightings in a portfolio are less able to be controlled. Yet even financial markets exhibit such indeterminate dynamics in individual stocks, with the allowance for changes to stock-weightings in financial portfolios. More so, the policy choices in Alaska largely prioritized walleye pollock, which gave it more weight in the portfolio and allowed both the overall portfolio and that individual stock to be managed consistent with that weighting. Certainly good management of the individual stock contributed to the results seen in Alaska, as it does in financial portfolios. But having the overall portfolio goal

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[5] https://www.nefsc.noaa.gov/ecosys/ecosystem-status-report/executive-summary.html.

allowed for some buffering of risk and, as has been argued in systems theory, led to a positive feedback loop, which reinforced the desired weighting. That is, as decisions in that region were made to prioritize (i.e., give higher weight to) walleye pollock and deemphasize other stocks, all cognizant of the risk of overfishing any one component species, over time the system responded to reflect that weighting, subject to the constraints of productivity in the entire system. So although the specific values and suggested weightings they imply are always going to be relatively imprecise in a fisheries context, they can be set and routinely evaluated for performance in a portfolio framework that is conducive to achieving overall fishery goals consistent with those weightings. I recognize that in a multispecies, multistock, multifleet context, the process for deriving those weightings of public resources is difficult, is not trivial, and will need to (continue to) be rather delicate. Such a "multi-multi" context is precisely why application of systems thinking can be beneficial, not only to drive the discussion of those weightings, but more so to accommodate such complexity and also better elucidate the collective benefits from the portfolio effect.

Another reason for broader consideration of the portfolio effect in fisheries is one of limitations to analytical capacity. There is simply no way the practice of the discipline is going to be able to assess all targeted stocks on a routine and regular basis. This is certainly true globally and even for well-resourced countries. In US marine ecosystems, there has been a renewed effort to develop a schema for stock assessment prioritization (Methot 2015). The specific criteria proposed are quite reasonable. Yet the fact that there is a need for this in well-resourced fisheries situations underscores the challenges that are highlighted from systems theory — that is, the challenges of understanding the dynamics of complex adaptive systems by taking a reductionist approach to understand the dynamics of every component. More so, this need for prioritization reflects the simple infeasibility of ever regularly doing stock-by-stock assessments beyond major or prioritized stocks and in essence ignores hierarchy theory. The need to take advantage of the near-decomposability and the portfolio effect is heightened because of this limitation. That is, adopting a portfolio approach may further assist in identifying higher prioritized — or weighted — stocks (relative to portfolio frontiers) and thereby free up analytical resources to focus on areas of greatest need. It can also provide an overall assessment of the entire fishery and thereby provide assessment coverage for all fished stocks relative to an overall OY as well as reduce collective and individual risk to that group of stocks.

Why do the barriers to adopting a holistic systems approach remain? These approaches to estimate production of fisheries in the ocean have been considered for over half a century (Graham and Edwards 1962; Schaefer 1965; Ryther 1969; Gulland 1970, 1971; cf. Smith 1994), but they have not become commonly operationalized. Instances in fisheries where this has been actually implemented — not just hypothetically tested or explored, but actually operationalized in a management context — have clearly demonstrated positive and desirable outcomes (i.e., Alaska). So why is there resistance to adopting this systems approach? I submit that there are at least four reasons. The first is simply unfamiliarity with the broad concepts and approaches of systems thinking as applied to the discipline of fisheries. That can be rectified by this paper and works like it as they continue to elaborate on systems-related approaches (Jackson 2003; Allen 2009; Mele et al. 2010; Wu 2013) and reinforcing works specific to fisheries (Walters 1971, 1980; Walters and Hilborn 1976; Charles 2001; Apollonio 2002, 2010). I would also encourage fisheries schools and other training fora to present this systems approach to better broaden awareness of its existence and the probable benefits from its application.

The second reason is simply historical inertia of this (or any) discipline. Largely, the theory and practice of fisheries have been developed on a stock-oriented basis, emphasizing component parts and a reductionist perspective. There is a long history of why and how this came about (Smith 1994; Finley 2011). In many ways this has been embodied in the either–or debates noted above (Andrewartha and Birch 1954; Hairston et al. 1960; Skud 1975; Smith 1994; Walters and Collie 1988; Rose 2000; Punt et al. 2014; Szuwalski et al. 2015), with reductionism leading to interminable debates, whereas holism leading to systematic solutions — that is, where there is clear recognition that a range of factors can and does influence fish stock and aggregate groups thereof. Reasonable minds are recognizing that we need to maintain the stock-oriented perspective in many instances, but that the scale and scope of challenges facing the discipline of fisheries is forcing an evaluation of methods and means towards a broader consideration of issues. Evaluating and tweaking the underlying philosophy (and associated world view) of how one understands phenomenon, and even approaches such understanding, is certainly difficult, but is also the hallmark of scientific progress.

The third reason is that there remains skepticism to novel approaches that is beyond just resistance to change, but healthy in a scientific context. Until the theory, clear predictability, mathematical representations, applied outcomes, and benefits of a new or different approach are demonstrated, such an approach will nearly always be slow to be adopted. As a corollary, most management or governance systems tend to oppose change until they understand and become familiar with the background and behavior of the proposed change. For instance, directing management action at one level in the hierarchy above the level where effects are desired is an elementary feature (and recommendation) of hierarchy theory, but may seem counterintuitive at first and as such would need familiarization and socialization before being enacted. I trust that the short primer on systems theory coupled with the worked examples shown and cited here escalate the consideration of this approach.

The fourth reason is one that relates to the first — the paralysis leading to inaction that stems from the recognition of overwhelming complexity in these systems. That thinking decries the limited data, difficulty in understanding, and near infeasibility of predictions associated with marine ecosystems and their fisheries. I and several others before me (von Bertalanffy 1968; cf. Luhmann 2013; Capra and Luisi 2014; Apollonio 2002, 2010) clearly recognize the essentially impossible task of understanding these coupled social–ecological marine ecosystems in a mechanistic, reductionist matter for each and every component. Yet this thinking entirely misses the utility of adopting a systems approach, namely in that it frees one from having to have such detailed data and understanding for each and every component of the system. Rather, from hierarchy theory, we do not have to have that level of understanding and in fact may actually glean further understanding of the system by adopting a systems approach that elucidates emergent properties. I am under no illusion that these barriers to systems thinking will resolve any time soon. Yet I am confident that at least some readers will give systems thinking due consideration for their work and application.

Perhaps a way around many of these obstacles is altering our thinking about OY. In particular, managing for aggregate OY can accrue all the benefits from adopting parts of a portfolio approach. Doing so translates what we manage for, at least in terms of focal levels, from a stock focus to an emphasis on the aggregate and emergent properties of fisheries systems. Many of the data, outputs, and reference points are the same, they would simply be based on systems thinking and take advantage of hierarchy theory and portfolio predictions of the aggregate. Thus, one can wonder if a paradigm shift is needed for the discipline, and I am inclined to think that it is. I would argue that a shift in emphasis in focal level, changing from a stock to an aggregate focus, is needed. Yet I also think many of our protocols for managing fisheries are robust to this potential shift in emphasis, and I think our governance structures can reasonably accommodate such a shift in emphasis. But to execute any shift in emphasis would require an associated change in our thinking and schools of thought, certainly in what is taught as part of fisheries science and management courses, but also in training for those actively making fisheries management decisions, all with an eye to the particular need of enhancing familiarity with the portfolio approach and its benefits. For example, simply conveying that the further a system is away from the portfolio frontier, the more revenue is going to be sacrificed is crucially important, but is also not routinely discussed in most fisheries contexts. As part of this shift, I think we need to alter what we monitor and track in fisheries portfolios. Currently in the US and in many other contexts, we monitor and manage off of some form of annual catch limits, which are generally based on MSY derivatives. I would argue that in addition to some variant of MSY-related biomass and fishing-rate reference points, OY should also explicitly and quantitatively consider the need to minimize variance of the fisheries catch across the entire fisheries portfolio and minimize distance from the portfolio efficiency frontier. Until we specifically and formally state how much risk we are willing to accept across **all** stocks and threats, how stable we would like our systems to be, and how much revenue is left on the table, I do not think we will ever fully encapsulate the intent of achieving OY for all fisheries in an ecosystem.

There has been repeated recognition of the increasing importance of ecosystem-level reference points (ELRPs; Link 2010; Shin et al. 2010). Many have been proposed, including a range of statistical and theoretical approaches (e.g., Pauly and Christensen 1995; Kerr and Dickie 2001; Jennings and Blanchard 2004; Bundy et al. 2005; Gascuel et al. 2005; Coll et al. 2008; Libralato et al. 2008; Heymans et al. 2014; Link et al. 2015). I certainly advocate for continued research and operational development of these decision support tools. At their core, these ELRPs seek to capitalize on emergent properties of ecosystems that arise from systems thinking. Such ELRPs have the benefit of capturing major signals more quickly, as they impact a broader set of the ecosystem versus waiting to detect signals from meta-analyses on individual components. They also have value in efficiently identifying dynamics and stressors that affect all system components simultaneously. Aggregate OY estimates would be a specific form of ELRPs particular to the fisheries sector. Although capitalizing on systems thinking and taking advantage of the portfolio effect, the resulting metrics from such an aggregate OY set of ELRPs points would have the added value of being rather easily interpretable and relative familiar, which is not always true for other emergent property ELRPs (Jørgensen 1997; Nielsen and Müller 2000). This is an important and active area of research, with copious opportunity for theoretical exploration and practical application.

I will conclude by repeating the “pollyanna-ish” statement: how would one like to manage fisheries in marine ecosystems such that

- risk of overfishing is minimized (i.e., above agreed upon levels or at least not substantially above natural levels);
- populations of fishes, catches, and profits are stable;
- overall value across all stocks is maximized;
- bureaucratic oversight and regulatory interventions are minimized;
- catch and yield are optimized;
- biomass of the resource (in aggregate) is maximized;
- stakeholder disenfranchisement and legal challenges are minimized;
- catch per unit effort is optimized; and
- risk of ancillary ecosystem impacts are minimized (again, above agreed upon levels or at least not substantially above natural levels)?

I recognize that achieving all this simultaneously is probably impossible in a practical sense. But wouldn’t it be nice to begin to approach that aspiration? By adopting a systems approach emphasizing fishery portfolios, we can begin to achieve such goals and thereby more of what is fully intended by the objectives of OY. I also recognize that the presentation of this topic may strike some as rather strident, with borderline advocacy for a paradigm shift and emphasis on the benefits of systems thinking beyond a typical proposal to consider such a thesis. That may be. Yet I fear that otherwise, without such a presentation, systems thinking and its associated demonstrated benefits will continue to be overlooked. All of us in the practice of fisheries science and management sincerely want to see fisheries stocks and marine ecosystems sustainably managed. Thus, when one identifies an approach that is theoretically defensible, has been proven in simulation and worked examples, represents an improvement to business-as-usual, and has case studies of clearly proven benefits, it is incumbent on one to escalate the debate over the uptake of such an approach. It is in this context of improving the status of fisheries and fish stocks — in both poorly and well-managed situations — that EBFM should continue to espouse a systems approach (Garcia et al. 2003; Pikitch et al. 2004; Garcia and Cochrane 2005; Levin and Lubchenco 2008; Link 2010; Fogarty 2014). The systems thinking and particularly portfolio approach highlighted herein represents a pragmatic way in which to implement EBFM and thereby reap the benefits it seeks to attain.

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References

Adams, K.M.G. 2015. Non-functional requirements in systems analysis and design. Springer, New York.

Ahl, V., and Allen, T.F.H. 1996. Hierarchy theory: a vision, vocabulary, and epistemology. Columbia University Press, New York.

Allen, P.M., and McGlade, J.M. 1987. Modelling complex human systems: a fisheries example. Eur. J. Op. Res. **30**: 147–167. doi:10.1016/0377-2217(87)90092-0.

Allen, T.F.H. 2009. Hierarchy theory in ecology. *In* Ecosystem ecology — a derivative of encyclopedia of ecology. *Edited by* S.E. Jørgensen. Elsevier, Amsterdam. pp. 114–120.

Allen, T.F.H., and Starr, T.B. 1982. Hierarchy: perspectives for ecological complexity. University of Chicago Press, Chicago.

Allen, T.F.H., O’Neill, R.V., and Hoekstra, T.W. 1984 Inter level relations in ecological research and management: some working principles from hierarchy theory. USDA Forest Service General Tech Report RM-110, Rocky Mountain Forest and Range Experiment Station, Fort Collins.

Anderson, S.C., Cooper, A.B., and Dulvy, N.K. 2013. Ecological prophets: quantifying metapopulation portfolio effects. Methods Ecol. Evol. **4**(10): 971–981. doi:10.1111/2041-210X.12093.

Andrewartha, H.G., and Birch, L.C. 1954. The distribution and abundance of animals. University of Chicago Press, Chicago.

Apollonio, S. 2002. Hierarchical perspectives on marine complexities: searching for systems in the Gulf of Maine. Columbia University Press, New York.

Apollonio, S. 2010. A testable hypothesis for ecosystem-based fisheries management. Rev. Fish. Sci. **18**(2): 183–188. doi:10.1080/10641262.2010.490885.

Apollonio, S. 2015. A hierarchical concept of ecosystems: an alternative to sector management for multi-species fisheries. Rev. Fish. Sci. Aquacult. **23**(1): 11–22. doi:10.1080/23308249.2014.988867.

Au, D.W.K. 1973. Total sustainable yield from Subareas 5 and 6 based on yield per recruit and primary production consideration. ICNAF Research Document 73/10. Serial No. 2912. ICNAF, Dartmouth, N.S.

Axelrod, R., and Cohen, M.D. 2001. Harnessing complexity. Basic Books, New York.

Baird, S.F. 1873. Report on the Condition of the Sea Fisheries of the South Coast of New England in 1871 and 1872. Government Printing Office, Washington, D.C.

Booth, D., and Fama, E. 1992. Diversification and asset contributions. Financial Analysts Journal, **48**(3): 26–32. doi:10.2469/faj.v48.n3.26.

Botsford, L.W., Castilla, J.C., and Peterson, C.H. 1997. The management of fisheries and marine ecosystems. Science, **277**: 509–515. doi:10.1126/science.277.5325.509.

Brigham, E.F., and Gapenski, L.C. 1988. Financial management: theory and practice. 5th ed. Dryden Press, Chicago.

Browman, H.I., and Stergiou, K.I. 2004. Perspectives on ecosystem-based approaches to the management of marine resources. Mar. Ecol. Prog. Ser. **274**: 269–303. doi:10.3354/meps274269.

Browman, H.I., and Stergiou, K.I. 2005. Politics and socio-economics of ecosystem-based management of marine resources. Mar. Ecol. Prog. Ser. **300**: 241–296. doi:10.3354/meps300241.

Brown, B., Breenan, J., Grosslein, M., Heyerdahl, E., and Hennemuth, R. 1976. The effect of fishing on the marine finfish biomass in the Northwest Atlantic from the Gulf of Maine to Cape Hatteras. ICNAF Res. Bull. **12**: 49–68.

Brown, B.L., Downing, A.L., and Leibold, M.A. 2016. Compensatory dynamics stabilize aggregate community properties in response to multiple types of perturbations. Ecology, **97**: 2021–2033. doi:10.1890/15-1951.1. PMID:27859207.

Bundy, A., Fanning, P., and Zwanenburg, K.C.T. 2005. Balancing exploitation and conservation of the eastern Scotian Shelf ecosystem: application of a 4D ecosystem exploitation index. ICES J. Mar. Sci. **62**: 503–510. doi:10.1016/j.icesjms.2004.12.008.

Bundy, A., Bohaboy, E.C., Hjermann, D.O., Mueter, F.J., Fu, C., and Link, J.S. 2012. Common patterns, common drivers: Comparative analysis of aggregate surplus production across ecosystems. Mar. Ecol. Prog. Ser. **459**: 203–218. doi:10.3354/meps09787.

Burgess, M.G., Giacomini, H.C., Szuwalski, C.S., Costello, C., and Gaines, S. 2017. Describing ecosystem contexts with single-species models: a theoretical synthesis for fisheries. Fish Fish. **18**: 264–284. doi:10.1111/faf.12179.

Capra, F., and Luisi, P.L. 2014. The systems view of life: a unifying vision. Cambridge University Press, Cambridge.

Charles, A. 2001. Sustainable fishery systems. Wiley-Blackwell, Oxford, UK.

Choueifaty, Y., and Coignard, Y. 2008. Toward maximum diversification. J. Portfolio Manage. **35**: 40–51. doi:10.3905/JPM.2008.35.1.40.

Christensen, V., Walters, C.J., Ahrens, R., Alder, J., Buszowski, J., Christiensen, L.B., et al. 2009. Database-driven models of the world's large marine ecosystems. Ecol. Model. **220**: 1984–1996. doi:10.1016/j.ecolmodel.2009.04.041.

Cilliers, P. 1998. Complexity and postmodernism: understanding complex systems. Routledge, London.

Coll, M., Libralato, S., Tudela, S., Palomera, I., and Pranovi, F. 2008. Ecosystem overfishing in the ocean. PLoS ONE, **3**(12): e3881. doi:10.1371/journal.pone.0003881. PMID:19066624.

Collie, J.S., and Gislason, H. 2001. Biological reference points for fish stocks in a multispecies context. Can. J. Fish. Aquat. Sci. **58**(11): 2167–2176. doi:10.1139/f01-158.

Constable, A.J. 2001. The ecosystem approach to managing fisheries: achieving conservation objectives for predators of fished species. CCAMLR Sci. **8**: 37–64.

Constable, A.J. 2011. Lessons from CCAMLR on the implementation of the ecosystem approach to managing fisheries. Fish Fish. **12**: 138–151. doi:10.1111/j.1467-2979.2011.00410.x.

Constable, A.J., de la Mare, W.K., Agnew, D.J., Everson, I., and Miller, D. 2000. Managing fisheries to conserve the Antarctic marine ecosystem: practical implementation of the Convention for the Conservation on Antarctic Marine Living Resources (CCAMLR). ICES J. Mar. Sci. **57**: 778–791. doi:10.1006/jmsc.2000.0725.

Cottingham, K.L., Brown, B.L., and Lennon, J.T. 2001. Biodiversity may regulate the temporal variability of ecological systems. Ecol. Lett. **4**: 72–85. doi:10.1046/j.1461-0248.2001.00189.x.

Crutchfield, J.A. 1973. Economic and political objectives in fishery management. Trans. Am. Fish. Soc. **102**: 481–491. doi:10.1577/1548-8659(1973)102<481:EAPOIF>2.0.CO;2.

Darcy, G.H., and Matlock, G.C. 1999. Application of the precautionary approach in the national standard guidelines for conservation and management of fisheries in the United States. ICES J. Mar. Sci. **56**: 853–859. doi:10.1006/jmsc.1999.0533.

Dichmont, C.M., Pascoe, S., Kompas, T., Punt, A.E., and Deng, R. 2010. On implementing maximum economic yield in commercial fisheries. Proc. Natl. Acad. Sci. **107**: 16–21. doi:10.1073/pnas.0912091107. PMID:20018676.

Doak, D.F., Bigger, D., Harding, E.K., Marvier, M.A., O'Malley, R.E., and Thomson, D. 1998. The statistical inevitability of stability–diversity relationships in ecology. Am. Nat. **151**: 264–276. doi:10.2307/2463348. PMID:18811357.

DuFour, M.R., May, C.J., Roseman, E.F., Ludsin, S.A., Vandergoot, C.S., Pritt, J.J., Fraker, M.E., Davis, J.J., Tyson, J.T., Miner, J.G., Marschall, E.A., and Mayer, C.M. 2015. Portfolio theory as a management tool to guide conservation and restoration of multi-stock fish populations. Ecosphere, **6**(12): 296. doi:10.1890/ES15-00237.1.

EAP. 2012. Ecosystem Status Report For the Northeast U.S. Continental Shelf Large Marine Ecosystem — 2011. Northeast Fisheries Science Center Reference Document, 12-07. Ecosystem Assessment Program.

Edwards, S.F., Link, J.S., and Rountree, B.P. 2004. Portfolio management of wild fish stocks. Ecol. Econ. **49**: 317–329. doi:10.1016/j.ecolecon.2004.04.002.

Elton, E.J., and Gruber, M.J. 1977. Risk reduction and portfolio size: an analytical solution. The Journal of Business, **50**: 415–437. doi:10.1086/295964.

Elton, E.J., and Gruber, M.J. 1995. Modern portfolio theory and investment analysis. Wiley, New York.

Fabozzi, F.J., and Markowitz, H.M. 2011. The theory and practice of investment management. John Wiley and Sons, Inc., Hoboken, N.J.

FAO. 2016. State of the world's fisheries and aquaculture. Food and Agricultural Organization of the United Nations, Rome.

Fay, G., Large, S.I., Link, J.S., and Gamble, R.J. 2013. Testing systemic fishing responses with ecological indicators: an MSE approach. Ecol. Model. **265**: 45–55. doi:10.1016/j.ecolmodel.2013.05.016.

Fay, G., Large, S.I., Link, J.S., and Gamble, R.J. 2015. Management performance of ecological indicators in the Georges Bank finfish fishery. ICES J. Mar. Sci. **72**(5): 1285–1296. doi:10.1093/icesjms/fsu214.

Federal Register. 2009. National Standard 1 Guidelines — Final Rule, vol. 74 [online]. Available from https://www.federalregister.gov/ [accessed 16 January 2009].

Finley, C. 2011. All the fish in the sea: maximum sustainable yield and the failure of fisheries management. University of Chicago Press, Chicago.

Finley, C., and Oreskes, N. 2013. Maximum sustained yield: a policy disguised as science. ICES J. Mar. Sci. **70**: 245–250. doi:10.1093/icesjms/fss192.

Fogarty, M.J. 2014. The art of ecosystem-based fishery management. Can. J. Fish. Aquat. Sci. **71**(3): 479–490. doi:10.1139/cjfas-2013-0203.

Fogarty, M.J., and Murawski, S.A. 1998. Large-scale disturbance and the structure of marine systems: fishery impacts on Georges Bank. Ecol. Appl. **8**(S1): S6–S22.

Fogarty, M.J., Overholtz, W.J., and Link, J.S. 2008. Fishery production potential of the Northeast Continental Shelf of the United States. GARM WP 3.5 GARM – III - BRP Meeting.

Fogarty, M.J., Overholtz, W.J., and Link, J.S. 2012. Aggregate surplus production models for the demersal fish complex of the Gulf of Maine. Mar. Ecol. Prog. Ser. **459**: 247–258. doi:10.3354/meps09789.

Fogarty, M.J., Gamble, R., and Perretti, C.T. 2016. Dynamic complexity in exploited marine ecosystems. Front. Ecol. Evol. **4**: 68. doi:10.3389/fevo.2016.00068.

Friedland, K.D., Stock, C., Drinkwater, K.F., Link, J.S., Leaf, R.T., Shank, B.V., et al. 2012. Pathways between primary production and fisheries yields of large marine ecosystems. PLoS ONE, **7**(1): e28945. doi:10.1371/journal.pone.0028945.

Fukuda, Y. 1976. A note on yield allocation in multi-species fisheries. ICNAF Res. Bull. **12**: 83–87.

Fulton, E.A., and Link, J.S. 2014. Modeling approaches for marine ecosystem-based management. *In* The sea. Vol. 16. Marine ecosystem-based management. *Edited by* M.J. Fogarty and J.J. McCarthy. Harvard University Press, Boston, Mass. pp. 121–170.

Fulton, E.A., Link, J., Kaplan, I.C., Johnson, P., Savina-Rolland, M., et al. 2011. Lessons in modelling and management of marine ecosystems: The Atlantis experience. Fish Fish. **12**: 171–188. doi:10.1111/j.1467-2979.2011.00412.x.

Fulton, E.A., Smith, A.D.M., Smith, D.C., and Johnson, P. 2014. An integrated approach is needed for ecosystem based fisheries management: insights from ecosystem-level management strategy evaluation. PLoS ONE, **9**(1): e84242. doi:10.1371/journal.pone.0084242. PMID:24454722.

Gaichas, S., Skaret, G., Falk-Petersen, J., Link, J.S., Overholtz, W., Megrey, B.A., et al. 2009. A comparison of community and trophic structure in five marine ecosystems based on energy budgets and system metrics. Prog. Oceanogr. **81**: 47–62. doi:10.1016/j.pocean.2009.04.005.

Gaichas, S., Gamble, R., Fogarty, M., Benoît, H., Essington, T., Fu, C., Koen-Alonso, M., and Link, J. 2012. Assembly rules for aggregate-species production models: simulations in support of management strategy evaluation. Mar. Ecol. Prog. Ser. **459**: 275–292. doi:10.3354/meps09650.

Gaichas, S.K., Fogarty, M., Fay, G., Gamble, R., Lucey, S., and Smith, L. 2017. Combining stock, multispecies, and ecosystem level fishery objectives within an operational management procedure: simulations to start the conversation. ICES J. Mar. Sci. **74**: 552–565. doi:10.1093/icesjms/fsw119.

Gamble, R.J., and Link, J.S. 2009. Analyzing the tradeoffs among ecological and fishing effects on an example fish community: a multispecies (fisheries) production model. Ecol. Model. **220**: 2570–2582. doi:10.1016/j.ecolmodel.2009.06.022.

Gamble, R.J., and Link, J.S. 2012. Using an aggregate production simulation model with ecological interactions to explore effects of fishing and climate on a fish community. Mar. Ecol. Prog. Ser. **459**: 259–274. doi:10.3354/meps09745.

Garcia, S.M., and Cochrane, K.L. 2005. Ecosystem approach to fisheries: a review of implementation guidelines. ICES J. Mar. Sci. **62**: 311–318. doi:10.1016/j.icesjms.2004.12.003.

Garcia, S.M., Zerbi, A., Aliaume, C., Do Chi, T., and Lasserre, G. 2003. The ecosystem approach to fisheries: issues, terminology, principles, institutional foundations, implementation and outlook. FAO Fisheries Technical Paper 443. FAO, Rome, Italy.

Gascuel, D., Bozec, Y., Chassot, E., Colomb, A., and Laurans, M. 2005. The trophic spectrum: theory and application as an ecosystem indicator. ICES J. Mar. Sci. **62**: 443–452. doi:10.1016/j.icesjms.2004.12.013.

Graham, H.W., and Edwards, R.L. 1962. The world biomass of marine fishes. *In* Fish in nutrition. *Edited by* E. Heen and R. Kreuzer. Fishing New Books, London.

Gulland, J.A. 1970. The fish resources of the ocean. FAO Fisheries Technical Paper No. 97. FAO, Rome.

Gulland, J.A. 1971. The fish resources of the ocean. Fishing New Books, West Byfleet, UK.

Hairston, N.G., Smith, F.E., and Slobodkin, L.B. 1960. Community structure, population control, and competition. Am. Nat. **94**: 421–425. doi:10.1086/282146.

Halpern, B.S., White, C., Lester, S.E., Costello, C., and Gaines, S.D. 2011. Using portfolio theory to assess tradeoffs between return from natural capital and social equity across space. Biol. Conserv. **144**(5): 1499–1507. doi:10.1016/j.biocon.2011.01.019.

Hare, S.R., and Mantua, N.J. 2000. Empirical evidence for North Pacific regime shifts in 1977 and 1989. Prog. Oceanogr. **47**: 103–145. doi:10.1016/S0079-6611(00)00033-1.

Heath, M. 2012. Ecosystem limits to food web fluxes and fisheries yields in the North Sea simulated with an end-to-end model. Prog. Oceanogr. **102**: 42–66. doi:10.1016/j.pocean.2012.03.004.

Heymans, J.J., Coll, M., Libralato, S., Morissette, L., and Christensen, V. 2014. Global patterns in ecological indicators of marine food webs: a modelling approach. PLoS ONE, **9**(4): e95845. doi:10.1371/journal.pone.0095845. PMID: 24763610.

Hilborn, R. 2010. Pretty good yield and exploited fishes. Mar. Pol. **34**: 193–196. doi:10.1016/j.marpol.2009.04.013.

Hilborn, R., Stewart, I.J., Branch, T.A., and Jensen, O.P. 2012. Defining trade-offs among conservation, profitability, and food security in the California Current bottom-trawl fishery. Conserv. Biol. **26**: 257–266. doi:10.1111/j.1523-1739.2011.01800.x. PMID:22443131.

Hilborn, R., Fulton, E.A., Green, B.S., Hartmann, K., Tracey, S.R., and Watson, R.A. 2015. When is a fishery sustainable? Can. J. Fish. Aquat. Sci. **72**(9): 1433–1441. doi:10.1139/cjfas-2015-0062.

Himes-Cornell, A., and Kasperski, S. 2016. Using Socioeconomic and Fisheries Involvement Indices to Understand Alaska Fishing Community Well-Being. Coastal Manage. **44**(1): 36–70. doi:10.1080/08920753.2016.1116671.

Hollowed, A.B., Hare, S.R., and Wooster, W.S. 2001. Pacific Basin climate variability and patterns of Northeast Pacific marine fish production. Prog. Oceanogr. **49**: 257–282. doi:10.1016/S0079-6611(01)00026-X.

Hollowed, A.B., Aydin, K.Y., Essington, T.E., Ianelli, J.N., Megrey, B.A., Punt, A.E., and Smith, A.D.M. 2011. Experience with quantitative ecosystem assessment tools in the northeast Pacific. Fish Fish. **12**: 189–208. doi:10.1111/j.1467-2979.2011.00413.x.

Iverson, R.L. 1990. Control of marine fish production. Limnol. Oceanogr. **35**(7): 1593–1604. doi:10.4319/lo.1990.35.7.1593.

Jackson, J.B.C., Kirby, M.X., Berger, W.H., Bjorndal, K.A., Botsford, L.W., Bourque, B.J., et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science, **293**: 629–638. doi:10.1126/science.1059199. PMID:11474098.

Jackson, M. 2003. Systems thinking: creative holism for managers. John Wiley & Sons, Ltd., Chichester.

Jacobsen, N.S., Bergess, M.G., and Anderson, K.H. 2017. Efficiency of fisheries is increasing at the ecosystem level. Fish Fish. **18**: 199–211. doi:10.1111/faf.12171.

Jennings, S., and Blanchard, J. 2004. Fish abundance with no fishing: predictions based on macroecological theory. J. Anim. Ecol. **73**: 632–642. doi:10.1111/j.0021-8790.2004.00839.x.

Jepson, M., and Colburn, L.L. 2013. Development of Social Indicators of Fishing Community Vulnerability and Resilience in the U.S. Southeast and Northeast Regions. US Dept. of Commerce., NOAA Technical Memorandum NMFS-F/SPO-129.

Jin, D., DePiper, G., and Hoagland, P. 2016. Applying portfolio management to implement Ecosystem-Based Fishery Management (EBFM). N. Am. J. Fish. Manage. **36**(3): 652–669. doi:10.1080/02755947.2016.1146180.

Jørgensen, S.E. 1995. Complex ecology in the 21st century. *In* Complex ecology: the part-whole relation in ecosystems. *Edited by* B.C. Patten, S.E. Jørgensen, and S.I. Auerbach. Prentice-Hall, Englewood Cliffs, N.J. pp. xvii–xix.

Jørgensen, S.E. 1997. Integration of ecosystem theories: a pattern. Kluwer, Boston.

Jørgensen, S.E., and Müller, F. (*Editors*). 2000. Handbook of ecosystem theories and management. Lewis Publishers, Boca Raton, Fla.

Kaplan, I.C., and Leonard, J. 2012. From krill to convenience stores: forecasting the economic and ecological effects of fisheries management on the US west coast. Mar. Pol. **36**: 947–954. doi:10.1016/j.marpol.2012.02.005.

Kerr, S.R., and Dickie, L.M. 2001. The biomass spectrum. Columbia University Press, New York.

Lansing, J.S. 2003. Complex adaptive systems. Ann. Rev. Anthropol. **32**: 183–204. doi:10.1146/annurev.anthro.32.061002.093440.

Larkin, P.A. 1977. An epitaph for the concept of maximum sustained yield. Trans. Am. Fish. Soc. **106**: 1–11. doi:10.1577/1548-8659(1977)106<1:AEFTCO>2.0.CO;2.

Legovic, T., and Gecek, S. 2011. Impact of maximum sustainable yield on independent populations. Ecol. Model. **221**: 2108–2111. doi:10.1016/j.ecolmodel.2010.05.015.

Legovic, T., Klanjscek, J., and Gecek, S. 2010. Maximum sustainable yield and species extinction in ecosystems. Ecol. Model. **221**: 1569–1574. doi:10.1016/j.ecolmodel.2010.03.024.

Levin, S.A. 1998. Ecosystems and the biosphere as complex adaptive systems. Ecosystems, **1**: 431–436. doi:10.1007/s100219900037.

Levin, S.A. 1999. Fragile dominions: complexity and the commons. Perseus Books, Reading.

Levin, S.A. 2003. Complex adaptive systems: exploring the known, the unknown, and the unknowable. Bull. Am. Math. Soc. **40**: 3–19.

Levin, S.A., and Lubchenco, J. 2008. Resilience, robustness, and marine ecosystem-based management. BioScience, **58**: 27–32. doi:10.1641/B580107.

Levin, S., Xepapadeas, T., Crepin, A.S., Norberg, J., De Zeeuw, A., Folke, C., Hughes, T., Arrow, K., Barrett, S., Daily, G., Ehrlich, P., Kautsky, N., Maler, K.G., Polasky, S., Troell, M., Vincent, J.R., and Walker, B. 2013. Social–ecological systems as complex adaptive systems: modeling and policy implications. Environ. Dev. Econ. **18**: 111–132. doi:10.1017/S1355770X12000460.

Libralato, S., Coll, M., Tudela, S., Palomera, I., and Pranovi, F. 2008. Novel index for quantification of ecosystem effects of fishing as removal of secondary production. Mar. Ecol. Prog. Ser. **355**: 107–129. doi:10.3354/meps07224.

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

Link, J.S., Stockhausen, W.T., Skaret, G., Overholtz, W., Megrey, B.A., Gjøsæter, H., Gaichas, S., Dommasnes, A., Falk-Petersen, J., Kane, J., Mueter, F.J., Friedland, K.D., and Hare, J.A. 2009. A comparison of biological trends from four marine ecosystems: synchronies, differences, and commonalities. Prog. Oceanogr. **81**: 29–46. doi:10.1016/j.pocean.2009.04.004.

Link, J.S., Fulton, E.A., and Gamble, R.J. 2010. The Northeast US Application of ATLANTIS: A full system model exploring marine ecosystem dynamics in a living marine resource management context. Prog. Oceanogr. **87**: 214–234. doi:10.1016/j.pocean.2010.09.020.

Link, J.S., Bundy, A., Overholtz, W.J., Shackell, N., Manderson, J., Duplisea, D., Hare, J., Koen-Alonso, M., and Friedland, K.D. 2011. Ecosystem-based fisheries management in the Northwest Atlantic. Fish Fish. **12**: 152–170. doi:10.1111/j.1467-2979.2011.00411.x.

Link, J.S., Gaichas, S., Miller, T.J., Essington, T., Bundy, A., Boldt, J., Drinkwater, K.F., and Moksness, E. 2012. Synthesizing lessons learned from comparing fisheries production in 13 Northern Hemisphere ecosystems: emergent fundamental features. Mar. Ecol. Prog. Ser. **459**: 293–302. doi:10.3354/meps09829.

Link, J.S., Pranovi, F., Libralato, S., Coll, M., Christensen, V., and Solidoro, C. 2015. Emergent properties delineate marine ecosystem perturbation and recovery. Trends Ecol. Evol. **30**: 649–661. doi:10.1016/j.tree.2015.08.011. PMID:26456382.

Lucey, S.M., Cook, A.M., Boldt, J., Link, J.S., Essignton, T., and Miller, T.J. 2012. Comparative analyses of surplus production dynamics of functional feeding groups across 12 northern hemisphere marine ecosystems. Mar. Ecol. Prog. Ser. **459**: 219–229. doi:10.3354/meps09825.

Luhmann, N. 2013. Introduction to systems theory. [Translated by P. Gilgen.] Polity, Malden, Mass.

Mace, P.M. 1994. Relationships between common biological reference points used as thresholds and targets of fisheries management strategies. Can. J. Fish. Aquat. Sci. **51**(1): 110–122. doi:10.1139/f94-013.

Mace, P.M. 2001. A new role for MSY in single-species and ecosystem approaches to fisheries stock assessment and management. Fish Fish. **2**: 2–32. doi:10.1046/j.1467-2979.2001.00033.x.

Mahon, R., McConney, P., and Roy, R.N. 2008. Governing fisheries as complex adaptive systems. Mar. Pol. **32**: 104–112. doi:10.1016/j.marpol.2007.04.011.

Marchal, P., Lallemand, P., Stokes, K., and Thébaud, O. 2009. A comparative review of the fisheries resource management systems in New Zealand and in the European Union. Aquat. Living Resour. **22**: 463–481. doi:10.1051/alr/2009032.

Mardle, S., Pascoe, S., Boncoeur, J., LeGallic, B., García-Hoyo, J.J., Herrero, I., Jimenez-Toribio, R., Cortes, C., Padilla, N., Nielsen, J.R., and Mathiesen, C. 2002. Objectives of fisheries management: case studies from the UK, France, Spain and Denmark. Mar. Pol. **26**: 415–428. doi:10.1016/S0308-597X(02)00022-2.

Markowitz, H. 1952. Portfolio selection. Journal of Finance, **7**: 77–91. doi:10.1111/j.1540-6261.1952.tb01525.x.

Marston, R.C. 2011. Portfolio design: a modern approach to asset allocation. John Wiley and Sons, Inc., Hoboken, N.J.

May, A.W. 1975. Report of standing committee on research and statistics. ICNAF Seventh Special Commission Meeting — September 1975. International Convention for the Northwest Atlantic Fisheries, Dartmouth, N.S.

May, R.M., Beddington, J.R., Clark, C.W., Holt, S.J., and Laws, R.M. 1979. Management of multispecies fisheries. Science, **205**: 267–277. doi:10.1126/science.205.4403.267. PMID:17747032.

Mayo, R.K., Fogarty, M.J., and Serchuk, F.M. 1992. Aggregate fish biomass and yield on Georges Bank, 1960-87. J. Northw. Atl. Fish. Sci. **14**: 59–78. doi:10.2960/J.v14.a4.

Meadows, D.H. 2008. Think in systems: a primer. Chelsea Green Publishing, White River Junction, Vermont.

Megrey, B.A., Link, J.S., Hunt, G.L., and Moksness, E. 2009. Comparative marine ecosystem analysis: applications, opportunities, and lessons learned. Prog. Oceanogr. **81**: 2–9. doi:10.1016/j.pocean.2009.04.002.

Mele, C., Pels, J., and Polese, F. 2010. A brief review of systems theories and their managerial applications. Service Science, **2**(1–2): 126–135. doi:10.1287/serv.2.1_2.126.

Mesnil, B. 2012. The hesitant emergence of maximum sustainable yield (MSY) in fisheries policies in Europe. Mar. Pol. **36**: 473–480. doi:10.1016/j.marpol.2011.08.006.

Methot, R.D. (*Editor*). 2015. Prioritizing fish stock assessments. US Dep. Commer., NOAA Tech. Memo. NMFS-F/SPO-152.

Methot, R.D., Tromble, G.R., Lambert, D.M., and Greene, K.E. 2014. Implementing a science-based system for preventing overfishing and guiding sustainable fisheries in the United States. ICES J. Mar. Sci. **71**: 183–194. doi:10.1093/icesjms/fst119.

Micheli, F. 1999. Eutrophication, fisheries, and consumer–resource dynamics in marine pelagic ecosystems. Science, **285**: 1396–1398. doi:10.1126/science.285.5432.1396. PMID:10464097.

Miller, J.H., and Page, S.E. 2007. Complex adaptive systems: an introduction to computational models of social life. Princeton University Press, Princeton, N.J.

Mueter, F.J., and Megrey, B.A. 2006. Using multi-species surplus production models to estimate ecosystem-level maximum sustainable yields. Fish. Res. **81**: 189–201. doi:10.1016/j.fishres.2006.07.010.

Nielsen, S.N., and Müller, F. 2000. Emergent properties of ecosystems. *In* Handbook of ecosystem theories and management. *Edited by* S.E. Jørgensen and F. Müller. Lewis Publishers, Boca Raton, Fla. pp. 195–210.

O'Neill, R.V., DeAngelis, D.L., Waide, J.B., and Allen, T.F.H. 1986. A hierarchical concept of ecosystems. Princeton University Press, Princeton, N.J.

O'Neill, R.V., Gardner, R.H., Milne, B.T., Turner, M.G., and Jackson, B. 1991. Heterogeneity and spatial hierarchies. *In* Ecological heterogeneity. *Edited by* J. Kolasa and S.T.A. Pickett. Springer, New York. pp. 85–96.

Patrick, W.S., and Link, J.S. 2015*a*. Myths that continue to impede progress in ecosystem-based fisheries management. Fisheries, **40**: 155–160. doi:10.1080/03632415.2015.1024308.

Patrick, W.S., and Link, J.S. 2015*b*. Hidden in plain sight: using optimum yield as a policy framework to operationalize ecosystem-based fisheries management. Mar. Pol. **62**: 74–81. doi:10.1016/j.marpol.2015.08.014.

Pattee, E.E. (*Editor*). 1973. Hierarchy theory: the challenge of complex systems. George Braziller, New York.

Pauly, D., and Christensen, V. 1995. Primary production required to sustain global fisheries. Nature, **376**(6537): 279–279. doi:10.1038/376279b0.

Pauly, D., Christensen, V., Guénette, S., Pitcher, T.J., Sumaila, U.R., Walters, C.J., Watson, R., and Zeller, D. 2002. Towards sustainability in world fisheries. Nature, **418**: 689–695. doi:10.1038/nature01017. PMID:12167876.

Pauly, D., Alder, J., Bennett, E., Christensen, V., Tyedmers, P., and Watson, R. 2003. The future for fisheries. Science, **302**: 1359–1361. doi:10.1126/science.1088667. PMID:14631031.

Pershing, A.J., Alexander, M.A., Hernandez, C.M., Kerr, L.A., Le Bris, A., Mills, K.E., et al. 2015. Slow adaptation in the face of rapid warming leads to collapse of the Gulf of Maine cod fishery. Science, **350**(6262): 809–812. doi:10.1126/science.aac9819. PMID:26516197.

Pikitch, E.K., Santora, C., Babcock, E.A., Bakun, A., Bonfil, R., Conover, D.O., et al. 2004. Ecosystem-based fishery management. Science, **305**: 346–347. doi:10.1126/science.1098222. PMID:15256658.

Pitcher, T.J., Kalikoski, D., Short, K., Varkey, D., and Pramod, G. 2009. An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. Mar. Pol. **33**: 223–232. doi:10.1016/j.marpol.2008.06.002.

Pope, J.G. 1975. The application of mixed fisheries theory to the cod and redfish stocks of subarea 2 and division 3K. ICNAF Res Doc75/IX/126. International Convention for the Northwest Atlantic Fisheries, Dartmouth, N.S., Canada.

Pope, J. 1979. Stock assessment in multistock fisheries, with special reference to the trawl fishery in the Gulf of Thailand. SCS/DEV/79/19. FAO, Manila.

Punt, A.E., A'mar, T., Bond, N.A., Butterworth, D.S., de Moor, C.L., De Oliveira, J.A.A., Haltuch, M.A., Hollowed, A.B., and Szuwalski, C. 2014. Fisheries management under climate and environmental uncertainty: control rules and performance simulation. ICES J. Mar. Sci. **71**: 2208–2220. doi:10.1093/icesjms/fst057.

Rădulescu, M., Rădelscu, C.Z., Rahoveanu, M.T., and Zbăganau, G. 2010. A portfolio theory approach to fishery management. Stud. Inform. Control, **19**(3): 285–294.

Ralston, S., and Polovina, J.J. 1982. A multispecies analysis of the commercial deep-sea handline fishery in Hawaii. Fish. Bull. **80**: 435–448.

Restrepo, V.R., Thompson, G.G., Mace, P.M., Gabriel, W.L., Low, L.L., MacCall, A.D., et al. 1998. Technical guidance on the use of precautionary approaches to implementing National Standard 1 of the Magnuson–Stevens Fishery Conservation and Management Act. NOAA Technical Memorandum NMFS-F/SPO 31.

Rindorf, A., Dichmont, C.M., Levin, P.S., Mace, P., Pascoe, S., Prellezo, R., Punt, A.E., Reid, D.G., Stephenson, R., Ulrich, C., Vinther, M., and Clausen, L.W. 2017*a*. Food for thought: pretty good multispecies yield. ICES J. Mar. Sci. **74**: 475–486.

Rindorf, A., Dichmont, C.M., Thorson, J., Charles, A., Clausen, L.W., Degnbol, P., Garcia, D., Hintzen, N.T., Kempf, A., Levin, P., Mace, P., Maravelias, C., Minto, C., Mumford, J., Pascoe, S., Prellezo, R., Punt, A.E., Reid, D.G., Rockmann, C., Stephenson, R.L., Thebaud, O., Tserpes, G., and Voss, R. 2017*b*. Inclusion of ecological, economic, social, and institutional considerations when setting targets and limits for multispecies fisheries. ICES J. Mar. Sci. **74**: 453–463. doi:10.1093/icesjms/fsw226.

Rose, K.A. 2000. Why are quantitative relationships between environmental quality and fish populations so elusive? Ecol. Appl. **10**: 367–385. doi:10.1890/1051-0761(2000)010[0367:WAQRBE]2.0.CO;2.

Rosen, R. 1969. Hierarchical organization in automata theoretic models of biological systems. *In* Hierarchical structures. *Edited by* L. Whyte, A.G. Wilson, and D. Wilson. Elsevier, New York. pp. 179–199.

Rosenberg, A.A., Fogarty, M.J., Cooper, A.B., Dickey-Collas, M., Fulton, E.A., Gutierrez, N., et al. 2014. Developing new approaches to global stock status assessment and fishery production potential of the seas. FAO Fisheries and Aquaculture Circular No. 1086. Rome, FAO.

Roy, A.D. 1952. Safety first and the holding of assets. Econometrica, **20**: 431–449. doi:10.2307/1907413.

Ryther, J.H. 1969. Photosynthesis and fish production in the sea. Science, **166**: 72–76. doi:10.1126/science.166.3901.72. PMID:5817762.

Sanchirico, J.N., Smith, M.D., and Lipton, D.W. 2008. An empirical approach to ecosystem-based fishery management. Ecol. Econ. **64**: 586–596. doi:10.1016/j.ecolecon.2007.04.006.

Schaefer, M.B. 1954. Some aspects of the dynamics of populations important to the management of commercial marine fisheries. Bull. Int.-Am. Trop. Tuna Comm. **1**: 27–56.

Schaefer, M.B. 1965. The potential harvest of the sea. Trans. Am. Fish. Soc. **94**(2): 123–128. doi:10.1577/1548-8659(1965)94[123:TPHOTS]2.0.CO;2.

Schindler, D.E., Hilborn, R., Chasco, B., Boatright, C.P., Quinn, T.P., Rogers, L.A., and Webster, M.S. 2010. Population diversity and the portfolio effect in an exploited species. Nature, **465**: 609–612. doi:10.1038/nature09060. PMID:20520713.

Schindler, D.E., Armstrong, J.B., and Reed, T.E. 2015. The portfolio concept in ecology and evolution. Front. Ecol. Environ. **13**(5): 257–263. doi:10.1890/140275.

Sethi, S.A. 2010. Risk management for fisheries. Fish Fish. **11**: 341–365. doi:10.1111/j.1467-2979.2010.00363.x.

Shin, Y.-J., Shannon, L.J., Bundy, A., Coll, M., Blanchard, J.L., Borges, M.-F., Cotter, J., Cury, P.M., Heymans, J.J., Jouffre, D., Link, J.S., Möllmann, C., Ojaveer, H., Abdallahi, K.O.M., Rochet, M.-J., and Yemane, D. 2010. Using indicators for evaluating, comparing and communicating the ecological status of exploited marine ecosystems. Part 2: Setting the scene. ICES J. Mar. Sci. **67**: 692–716. doi:10.1093/icesjms/fsp294.

Simon, H.A. 1962. The architecture of complexity. Proc. Am. Philos. Soc. **106**(6): 467–482.

Simon, H.A. 1973. The organization of complex systems. *In* Hierarchy theory: the challenge of complex systems. *Edited by* H.H. Pattee. George Braziller, New York. pp. 1–27.

Simon, H.A. 1976. How complex are complex systems? *In* Proceedings of the Biennial Meeting of the Philosophy of Science Association, Ann Arbor. pp. 507–522.

Simon, H.A. 1996. The sciences of the artificial. 3rd ed. The MIT Press, Cambridge.

Simon, H.A., and Ando, A. 1961. Aggregation of variables in dynamic systems. Econometrica, **29**: 111–138. doi:10.2307/1909285.

Sissenwine, M.P. 1978. Is MSY an adequate foundation for optimum yield? Fisheries, **3**: 22–42. doi:10.1577/1548-8446(1978)003<0022:IMAAFF>2.0.CO;2.

Skud, B.E. 1975. Revised estimates of halibut abundance and the Thompson-Burkenroad debate. Report of the International North Pacific Halibut Commission.

Smith, L., Gamble, R., Gaichas, S., and Link, J. 2015. Simulations to evaluate trade-offs among marine mammal consumption needs, commercial fishing fleets and finfish biomass. Mar. Ecol. Prog. Ser. **523**: 215–232. doi:10.3354/meps11129.

Smith, T.D. 1994. Scaling fisheries: the science of measuring the effects of fishing, 1855–1955. Cambridge University Press, New York.

Smith, T.D. 1998. "Simultaneous and complementary advances": Mid-century expectations of the interaction of fisheries science and management. Rev. Fish Biol. Fish. **8**: 335–348. doi:10.1023/A:1008823503867.

Smith, T.D., and Link, J.S. 2005. Autopsy your dead…and living: a proposal for fisheries science, fisheries management and fisheries. Fish Fish. **6**: 73–87. doi:10.1111/j.1467-2679.2005.00176.x.

Sparholt, H., and Cook, R.M. 2010. Sustainable exploitation of temperate fish stocks. Biol. Lett. **6**(1): 124–127. doi:10.1098/rsbl.2009.0516. PMID:19740893.

Steele, J.H., Gifford, D.J., and Collie, J.S. 2011. Comparing species and ecosystem-based estimates of fisheries yields. Fish. Res. **111**: 139–144. doi:10.1016/j.fishres.2011.07.009.

Szuwalski, C.S., Vert-Pre, K.A., Punt, A.E., Branch, T.A., and Hilborn, R. 2015. Examining common assumptions about recruitment: a meta-analysis of recruitment dynamics for worldwide marine fisheries. Fish Fish. **16**: 633–648. doi:10.1111/faf.12083.

Thibault, L.M., and Connolly, S.R. 2013. Understanding diversity–stability relationships: towards a unified model of portfolio effects. Ecol. Lett. **16**: 140–150. doi:10.1111/ele.12019. PMID:23095077.

Tilman, D. 1999. The ecological consequences of changes in biodiversity: a search for general principles. Ecology, **80**: 1455–1474. doi:10.2307/176540.

Tilman, D., Lehman, C.L., and Bristow, C.E. 1998. Diversity–stability relationships: statistical inevitability or ecological consequence? Am. Nat. **151**: 277–282. doi:10.2307/2463349. PMID:18811358.

Townsend, H.M., Harvey, C., Aydin, K.Y., Gamble, R., Grüss, A., Levin, P.S., Link, J.S., Osgood, K.E., Polovina, J., Schirripa, M.J., and Wells. B. (*Editors*). 2014. Report of the 3rd National Ecosystem Modeling Workshop (NEMoW 3): Mingling Models for Marine Resource Management – Multiple Model Inference. NOAA Tech. Memo. NMFS-F/SPO-149.

Urban, D.L., O'Neill, R.V., and Shugart, H.H. 1987. Landscape ecology: a hierarchical perspective can help scientists understand spatial patterns. Landsc. Ecol. **37**: 119–127. doi:10.2307/1310366.

von Bertalanffy, L. 1938. A quantitative theory of organic growth (inquiries on growth laws. II). Hum. Biol. **10**: 181–213.

von Bertalanffy, L. 1968. General system theory: foundations, development, applications. George Braziller, New York.

Walters, C.J. 1971. Systems ecology: the systems approach and mathematical models in ecology. *In* Fundamentals of ecology. *Edited by* E.P. Odum. Saunders, Philadelphia. pp. 276–292.

Walters, C.J. 1980. Systems principles in fisheries management. *In* Fisheries management. *Edited by* R.T. Lackey and L.A. Nielsen. Wiley, New York. pp. 167–183.

Walters, C.J., and Collie, J.S. 1988. Is research on environmental factors useful to fisheries management? Can. J. Fish. Aquat. Sci. **45**(10): 1848–1854. doi:10.1139/f88-217.

Walters, C.J., and Hilborn, R. 1976. Adaptive control of fishing systems. J. Fish. Res. Bd. Can. **33**(1): 145–159. doi:10.1139/f76-017.

Walters, C.J., Christensen, V., Martell, S.J., and Kitchell, J.F. 2005. Possible ecosystem impacts of applying MSY policies from single-species assessment. ICES J. Mar. Sci. **62**: 558–568. doi:10.1016/j.icesjms.2004.12.005.

Waltho, N., and Kolasa, J. 1994. Organization of instabilities in multispecies systems, a test of hierarchy theory. Proc. Natl. Acad. Sci. **91**: 1682–1685. doi:10.1073/pnas.91.5.1682. PMID:11607461.

Ware, D.M. 2000. Aquatic ecosystems: properties and models. *In* Fisheries oceanography: an integrative approach to fisheries and ecology and management. *Edited by* P.J. Harrison and T.R. Parsons. Blackwell Science, Oxford, UK.

Weinberg, G.M. 2001. An introduction to general systems thinking. Dorset House Publishing Company, New York.

Witherell, D., Pautzke, C.P., and Fluharty, D. 2000. An ecosystem-based approach for Alaska groundfish fisheries. ICES J. Mar. Sci. **57**: 771–777. doi:10.1006/jmsc.2000.0719.

Worm, B., Hilborn, R., Baum, J.K., Branch, T.A., Collie, J.S., Costello, C., Fogarty, M.J., Fulton, E.A., Hutchings, J.A., Jennings, S., Jensen, O.P., Lotze, H.K., Mace, P.M., McClanahan, T.R., Minto, C., Palumbi, S.R., Parma, A.M., Ricard, D., Rosenberg, A.A., Watson, R., and Zeller, D. 2009. Rebuilding global fisheries. Science, **325**(5940): 578–585. doi:10.1126/science.1173146. PMID:19644114.

Wu, J. 1999. Hierarchy and scaling: extrapolating information along a scaling ladder. Can. J. Rem. Sens. **25**: 367–380. doi:10.1080/07038992.1999.10874736.

Wu, J.G. 2013. Hierarchy theory: an overview. *In* Linking ecology and ethics for a changing world: values, philosophy, and action. *Edited by* R. Rozzi, S.T.A. Pickett, C. Palmer, J.J. Armesto, and J.B. Callicott. Springer, New York. pp. 281–301.

Wu, J., and Marceau, D. 2002. Modeling complex ecological systems: an introduction. Ecol. Model. **153**: 1–6. doi:10.1016/S0304-3800(01)00498-7.

Yang, M.M. 2011. The portfolio analysis for multi-species fisheries management in New Zealand: an application in a fishing firm. Doctoral dissertation, University of Auckland, Auckland, New Zealand.

Zador, S. (*Editor*). 2015. Ecosystems Considerations 2015: Status of Alaska's marine ecosystems. NPFMC, Anchorage, Alaska.