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Remaining questions in the case for balanced harvesting

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Abstract

Balanced harvesting – harvesting all species and sizes in an ecosystem in proportion to their productivity – is a fisheries management strategy that has been suggested recently to increase yields, while reducing overall ecosystem impact. However, some aspects of balanced harvesting are controversial, including its call for extensive harvesting of juveniles and forage fish. Balanced harvesting also calls for targeting species and size-classes that are not currently marketable, possibly at a significant economic cost. Some have argued that this cost is outweighed by the ecological benefits of maintaining the ecosystem size and trophic structures and by the benefits of extra yield for food security. There is broad consensus that balanced harvesting would require major changes to fishery management institutions and consumer behaviour, and it is unclear to what extent it is physically possible with current technologies. For this reason, we argue that steps to implement balanced harvesting are difficult to justify until the case for it is more clearly resolved. We outline some of the pivotal questions that must be answered to make a convincing case for or against balanced harvesting, many of which can be answered

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empirically. In identifying these questions, we hope to offer a constructive path forward in resolving some of the key issues in the balanced harvesting debate.

Keywords Ecosystem-based fishery management, efficiency frontier, size limit, size-at-entry, size-spectrum, yield-per-recruit

Introduction

Fisheries management must balance human needs for food, profit, recreation and other benefits, while minimizing adverse ecological impacts and degradation of ecosystem services (Beddington *et al.* 2007; Worm *et al.* 2009; Chan *et al.* 2012). There is a broad scientific consensus calling for a transition to ecosystem-based fishery management (EBFM) (Pikitch *et al.* 2004; Francis *et al.* 2011), which explicitly considers interactions among ecosystem components and holistically manages the fisheries in an ecosystem, rather than managing each species in isolation. Concrete policy commitments and some steps towards implementation of EBFM have been made in parts of the United States, European Union, Australia, Canada and a few other countries (Tromble 2008; Jennings and Rice 2011; Link *et al.* 2011; Micheli *et al.* 2014). However, in many of the world's fisheries, we lack clear EBFM goals, specific data- and institution-tested strategies for achieving these goals, or both (Hilborn 2011; Link *et al.* 2011; Skern-Mauritzen *et al.* 2015).

One EBFM strategy that has been suggested in several recent studies is 'balanced harvesting' (Zhou *et al.* 2010; Garcia *et al.* 2012; Law *et al.* 2014). Under this strategy, all age classes and species are harvested at rates proportional to their productivity. Although the operational definition of 'productivity' varies somewhat among studies (see Law *et al.* 2014), the motivations are similar. The rationale is that balanced harvesting might offer an opportunity to increase overall fishery yields (Garcia *et al.* 2012; Law *et al.* 2012; Zhou *et al.* 2014), which is beneficial to food security (Zhou *et al.* 2014), while also providing diverse environmental benefits – reducing the number of population collapses (Law *et al.* 2012, 2014; Zhou *et al.* 2014), maintaining the ecosystem size-structure and trophic structure (Zhou *et al.* 2010, 2014; Garcia *et al.* 2012; Jacobsen *et al.* 2014), increasing resilience (Law *et al.* 2012, 2014) and reducing the effects of fishery-induced evolution (Law *et al.* 2015; see also Borrell 2013). Fishing patterns similar to balanced harvesting, and arguably some of its benefits, have been observed in some small-scale fisheries in developing countries (e.g. Kolding and van Zwieten 2014).

In contrast to these possible benefits, studies proposing balanced harvesting (e.g. Law *et al.* 2014; Zhou *et al.* 2014) recognize that it may also have economic costs, because the catch under this strategy becomes dominated by small, likely low-value fish (Jacobsen *et al.* 2014; see also Diekert *et al.* 2010). Given such potential trade-offs, a key question is whether the benefits for ecosystems and food security outweigh potential economic losses (Law *et al.* 2014; Zhou *et al.* 2014), and if so, who would pay the cost.

Balanced harvesting generates other controversies as well. Notably, it calls for high fishing mortality on juveniles and forage fish, which both tend to be productive. This challenges the conventional wisdom from single-species yield-per-recruit theory (e.g. Beverton and Holt 1957; see also Borrell 2013) that yield is maximized by only harvesting mature fish and is at odds with other EBFM studies (e.g. Smith *et al.* 2011), which have called for reductions in fishing mortality on lower trophic levels to protect a vital food resource for top predator species, both fished and unfished.

Because implementing balanced harvesting on a large scale would require major shifts in current fishery management paradigms at potentially high costs (Zhou *et al.* 2014), it is critical to empirically resolve the case for balanced harvesting before taking steps towards implementation. Thus far, studies proposing balanced harvesting and studies seemingly at odds with balanced harvesting have been difficult to compare. Either they focus on different objectives (e.g. profits and conservation of predators vs. overall yield and maintaining size-structure) and ignore their potential trade-offs or they use different model-types (e.g. single-species models with discrete age-structure vs. size-spectrum ecosystem models). Studies seemingly at odds with aspects of balanced harvesting have often not squarely aimed their criticism at the general concept of balanced harvesting. As a result, the important debate remains poorly framed. Moreover, studies evaluating the merits of balanced harvesting at a global scale have largely used numerical simulations of complex ecosystem and size-spectrum models (e.g. Garcia *et al.* 2012; Jacobsen *et al.* 2014; Law *et al.* 2014), whose complexity can make it challenging to discern the mechanisms driving key results and to translate these into empirically testable hypotheses.

Here, we outline several clear and critical questions needing answers before we can determine whether or where balanced harvesting is in fact a desirable and feasible strategy. Although our list is unlikely comprehensive, these questions offer a constructive path towards empirically resolving some of the key issues surrounding balanced harvesting, and more generally EBFM.

Unresolved questions in the case for balanced harvesting

What is balanced harvesting and to what extent is it technologically feasible?

A precise operational definition of balanced harvesting is important to designing its implementation, but the operational definition has varied across proposals. The main element of the balanced harvesting strategy is harvesting each ecosystem component (each unique combination of size and species) in proportion to a measure of its productivity. However, the specific measure of productivity to use is a subject of active debate (see Law *et al.* 2014). For example, Garcia *et al.* (2012) define productivity as the 'amount of new organic matter produced per biomass unit during a given period of time' (units of time^{-1}), whereas Law *et al.* (2014) define productivity as the amount of organic matter produced per unit volume per unit time (units of mass × volume^{-1} × time^{-1}).

These definitions are qualitatively different. They imply a different ratio between the fishing mortality of different components of the ecosystem, and they differ in their approach to management. The first definition implies a more static management where fishing mortality may be set independent of the abundance of the fished ecosystem component. The other definition implies a more dynamic management where the fishing mortality is proportional to the abundance of the fished component. This latter definition leads to a strongly stabilizing effect of fishing (Law *et al.* 2014), with decreasing fishing mortality at low abundance to allow a recovery, and increasing mortality to maximize yield when the abundance is high. However, adjusting harvest rates dynamically to abundance at the size-by-species level would require both technological precision and significant monitoring efforts comparable to modern stock assessments, but now for all species in the ecosystem.

Implementing balanced harvesting faces other more general technological challenges. For example, in multispecies fisheries, it is rarely possible to exactly match catches to quotas because of the way in which species are caught together by many gear types (e.g. Branch and Hilborn 2008). For similar reasons, it is likely not possible to exactly balance exploitation rates across species or sizes with any measure of productivity. It is unknown how sensitive any ecological and yield benefits of balanced harvesting would be to constraints on the precision of implementation, or to empirical uncertainties in estimating productivity. Many of these sensitivities could be estimated using management strategy evaluation (Smith *et al.* 1999). However, in the remaining discussion, we consider potential benefits and costs of balanced harvesting, assuming that implementation under either definition is technologically feasible.

What sizes should be off-limits to fishing?

All definitions of balanced harvesting controversially call for high harvest rates on small fish, including juveniles and forage fish, because they are highly productive (Law *et al.* 2012; Jacobsen *et al.* 2014). In contrast, conventional fisheries management widely employs minimum size limits protecting juveniles (Hilborn 2011). Minimum size limits in management are often motivated by classic yield-per-recruit (YPR) theory of harvesting a single age-structured population, which predicts yield losses from harvesting juvenile fish (Beverton and Holt 1957; Froese and Binohlan 2000). Similarly, for managing entire fish communities, some studies have called for reducing exploitation of forage fish low in the food web to protect the yields and revenues from more valuable predatory fish (Smith *et al.* 2011) and to protect unfished higher trophic level conservation targets (e.g. marine mammals, penguins). Maximum size limits have been implemented or proposed in some fisheries to protect old 'mega-spawners' (e.g. Froese 2004) [also called 'big old fat fecund female fish' (BOFFFFs) (Hixon *et al.* 2014)] or guard against fishery-induced evolution (e.g. Conover and Munch 2002).

Less widely appreciated is the fact that recent studies proposing balanced harvesting actually also suggest a minimum size limit, at least for maximizing yields (Jacobsen *et al.* 2014; Law *et al.* 2014). Thus, the concept of balanced harvesting is not necessarily incompatible with minimum size limits, *per se*. Instead, the key conceptual difference between some balanced harvesting proposals (e.g. Law *et al.* 2014) and conventional fisheries theory seems to be how small this minimum size should be.

The logic underlying a minimum size limit is that harvesting a small fish provides little yield directly (because it is small), but comes at a high indirect yield cost, because that small fish could have grown into a larger fish, and eventually reproduced, or been eaten and converted into biomass of a larger fish (in an ecosystem), had it not been harvested. In single-species YPR theory, the yield-maximizing size limit – where somatic growth exactly balances natural mortality – is typically slightly larger than the size at first maturity (see Froese and Binohlan 2000 for review). For Atlantic cod (*Gadus morhua*, Gadidae), for example, size at first maturity is approximately 2.5 kg (Froese and Pauly 2013). In contrast, studies of balanced harvesting [on both single-species (Law *et al.* 2015) and ecosystems (Jacobsen *et al.* 2014; Law *et al.* 2014)] using size-spectrum models suggest an optimal minimum size limit for yield maximization of roughly 1 g (see Figure A1 in Jacobsen *et al.* 2014; Figure 1c,d in Law *et al.* 2014), which is small enough to be trivial in practice.

If fundamentally different models (e.g. YPR models vs. size-spectrum models) produce radically different recommended size limits, how do we empirically determine which recommendation is most appropriate? First, we can determine which differences in model assumptions are key drivers of the different predictions and test these against data. Second, we can look for directly empirically comparable or testable predictions in one or more of the models.

A key difference between the approaches is what they assume about the sources of natural mortality rates. YPR theory assumes natural mortality is exogenous (constant in time and typically also over age/size groups). Therefore, catching large fish does not reduce the future natural mortality rates of smaller fish. By contrast, the size-spectrum models assume that a major component of the natural mortality rate of small fish is predation by larger fish, including conspecifics (Law *et al.* 2012, 2014; Jacobsen *et al.* 2014). Thus, the indirect yield costs of harvesting all but the very smallest fish are smaller in size-spectrum models than in YPR theory, because harvesting today reduces future predation on small size-classes, in addition to preventing future reproduction and growth. Predation, including cannibalism, is known to contribute to the natural mortality of juvenile fish (see Engelhard *et al.* 2014 for review), but the scope of studies is limited and future research could evaluate the importance of predation relative to other sources of natural mortality empirically. Sensitivity analyses adjusting the importance of predation to natural mortality in size-spectrum models could further illuminate their degree of importance to model predictions.

The importance of predation to the natural mortality of small fish assumed in size-spectrum models has interesting, empirically testable, consequences. For example, YPR models generally predict a peak in the cohort biomass for cod at >1 kg (see Froese and Binohlan 2000; Froese and Pauly 2013). In contrast, Law *et al.* (2014) predict a second, order of magnitude larger peak at 0.1 y age (<1 g size). More intriguingly, size-spectrum models often predict counterintuitive biomass relationships between predatory fish species and their prey because of predation by large prey individuals on juvenile predators (the concept of 'overcompensation'; see De Roos and Persson 2002). For example, Law *et al.* (2014) predict (see their Figure 3), under roughly realistic current knife-edge selectivity at 100 g for mackerel (*Scomber scombrus*, Scombridae) and 1 kg for Atlantic cod, that if cod fishing mortality, F_c, is fixed at $F_c = 0.5$ years^{-1} and mackerel fishing mortality, F_m, is increased from ~ 0.5 years^{-1} to $F_m = 6$ (a twelve-fold increase), cod biomass should increase by roughly a factor of five. For comparison, the mean fishing mortality (F_c) estimated among assessed Atlantic cod stocks in the RAM Legacy Database in 2008 ($n = 20$) was 0.55 years^{-1}, and the mean F among assessed forage fish in 2008 ($n = 36$) was 0.47 years^{-1} (Ricard *et al.* 2012). Predictions from the above size-spectrum models imply that forage fish of currently harvested sizes have greater limiting effects through predation on juveniles than facilitating effects on predatory fish growth. The interaction between cod and mackerel, for example, is therefore more competitive than predatory. This hypothesis, if correct, would constitute a major paradigm shift in our understanding of marine food webs and merits a commensurate level of scrutiny.

What is the economic cost of balanced harvesting?

Balanced harvesting probably would incur significant economic costs relative to the status quo, at least at a global scale (Law *et al.* 2014; Zhou *et al.* 2014). There are three reasons for this: First and foremost, balanced harvesting calls for new and expanded harvest on species and sizes that are currently not targeted. Both the costs of developing technologies to be able to harvest fish in size ranges between 1 and 10 g and the costs of hauling the required quantities of these small fish are unknown, but they are likely to be substantial. The market potential for these types of products is not yet explored, but new fisheries that have been recently developed have been mostly low biomass and low revenue (Sethi *et al.* 2010), suggesting few remaining economic opportunities in unexploited species. Second, many of the small forage fish that make up most of the catch under balanced harvesting currently have a low market value (Andersen *et al.* 2015), although catches of some valuable low trophic invertebrates could increase, and future increases in demand for animal feeds (Tacon and Metian 2013) might buffer forage fish prices somewhat against the consequences of a supply glut. Third, increasing harvest rates on small and low trophic species could reduce yields of large-bodied, predatory species and individuals (e.g. Diekert *et al.* 2010; Smith *et al.* 2011), although some size-spectrum models have predicted the opposite (Houle *et al.* 2013; Law *et al.* 2014). Empirical, experimental (in the case of new technologies) and simulation-based research quantifying these economic costs at relevant scales is a critical prerequisite to implementation.

Some recent studies offer preliminary insights into the scale of the cost of maximizing yield globally under balanced harvesting. Garcia *et al.*'s (2012) analysis suggests that ecosystems generate nearly twice as much yield on average under yield-maximizing balanced harvesting relative to yield-maximization under existing selectivity patterns, although other analyses (Jacobsen *et al.* 2014) have suggested a much smaller difference. Average prices under balanced harvesting are almost certain to be lower than under current selectivity, because balanced harvesting increases the overall fish supply (Garcia *et al.* 2012) and shifts production towards small fish of mostly lower value (Jacobsen *et al.* 2014). Thus, total revenues under balanced harvesting would be at most double revenues under maximum yield with current selectivity, but probably much less. The question is whether added costs would more than offset these overly generous revenue estimates.

Sumaila *et al.* (2012) estimate that rebuilding existing fisheries globally to achieve maximum sustainable yield would require halving current fishing effort, which would also halve equilibrium fishing costs. They estimate that the resulting equilibrium revenue stream would be just over double the resulting costs. Applying these estimates to a global balanced harvesting scenario, if revenues were to double from higher yields, breaking even would require aggregate costs to be at most quadruple the aggregate costs under rebuilding – double the current costs. The break-even cost threshold for global balanced harvesting is likely much lower in reality, because revenues are likely to be much less than double the revenues under rebuilding.

It is difficult to exactly predict how much global yield-maximizing balanced harvesting would cost compared to current global fishing costs, but one possible benchmark for comparison is global average harvest rate (biomass harvested × total biomass^{-1} × time^{-1}). In a single-species fishery, for example, costs are proportional to average harvest rate if catchability and costs per-unit-effort are constant. Zhou *et al.* (2014) estimate that the sustainable ecosystem-wide harvest rate under balanced harvesting is at least twice the current rate (see their Figure 1). Thus, if fishing costs scale with average harvest rate, yield-maximizing balanced harvesting would almost certainly fail to break even on its costs.

Of course, there are a number of reasons fishing costs might not scale with average harvest rate at a global ecosystem scale. For example, marginal costs of the average harvest rate might diminish if there were economies of scale in fishing effort, or if balanced harvesting were achieved primarily by making fishing gears less selective. However, because much of the new mortality under balanced harvesting would be directed at sizes and species we are currently technologically ill-equipped to fish, it is also possible that the marginal costs of the average harvest rate would be higher under balanced harvesting. Much work needs to be carried out to resolve the economic costs of global balanced harvesting, but it is a non-negligible possibility that it could result in negative aggregate economic returns to fishing. This possibility merits careful further study.

What are the ecological benefits of balanced harvesting?

Balanced harvesting is posited to offer the following ecological benefits: (i) preservation of the pre-harvest ecosystem size and trophic structure (Garcia *et al.* 2012), (ii) decreased rates of fishery-induced evolution (Law *et al.* 2015) and (iii) increased resilience (Law *et al.* 2014, 2015). There is some empirical evidence for the first hypothesis (e.g. Kolding and van Zwieten 2014), but support for the latter two benefits remains theoretical. With respect to evolution changes, in so far as balanced harvesting reduces fishing pressure, it will likely also reduce the effects of fishing induced evolution (Andersen and Brander 2009). The logic for balanced harvesting increasing resilience is twofold. First, there is evidence that a truncated age-structure in harvested fish populations is destabilizing (Anderson *et al.* 2008). By distributing harvesting pressure over (almost) the entire age-spectrum, balanced harvesting could reduce some of the instability associated with fishing (Law *et al.* 2015). Second, an adaptive form of balanced harvesting, whereby harvests rates on particular sizes and species change as their productivities change, would act as a stabilizing force on populations, ecosystems and fisheries on its own (Law *et al.* 2014).

Although the benefits of maintaining the unfished size and trophic structure of an ecosystem could be significant, balanced harvesting reduces the biomass of all ecosystem components, which could have a significant ecological cost depending on the severity of the reduction. For example, Kolding and van Zwieten (2014) found that balanced harvesting in Lake Kariba, Zimbabwe did indeed preserve the ecosystem size-structure, but implied a depletion of roughly 80% of total ecosystem biomass (see their Figure 7). Balanced harvesting in other systems may not deplete the community biomass quite so severely, but the question of the relative importance of the ecosystem benefits from maintaining size-structure of an ecosystem vs. the benefits from maintaining higher species- or community-level biomass merits much greater attention. One of the likely sources of trade-offs might be the role of major components of system biomass in carbon storage. In terrestrial communities, forest biomass plays a key role in carbon storage, as well as habitat construction (Díaz *et al.* 2005). Emerging evidence from oceans suggests that deep-sea fish biomass can play a similar key role in sequestering carbon from shallow waters into the deep ocean (Trueman *et al.* 2014).

Is balanced harvesting efficient with respect to the environment-economy trade-off?

If the status quo outperforms balanced harvesting in economic terms, but balanced harvesting outperforms the status quo ecologically, we need to weigh economic and ecological objectives to arrive at an evaluation of the overall desirability of balanced harvesting as a fishing strategy. Agreeing upon subjective weights for economic and ecological objectives can be very difficult in practice, and the answer is likely to vary greatly among communities.

However, an equally important and likely easier question to answer is whether balanced harvesting is Pareto efficient with respect to the environment-economy trade-off (Fig. 1). Balanced harvesting is Pareto efficient if and only if there is no alternate strategy that outperforms balanced harvesting both economically and ecologically (see Polasky *et al.* 2008 for an analogous discussion of land use trade-offs). If such a strategy exists, then we would conclude that balanced harvesting is not a good strategy, at least with respect to these objectives (economic gains and environmental protection).

One strategy that is worth comparing balanced harvesting to at a global scale is not fishing at all. Not fishing seems likely to be the best possible strategy ecologically, at least from a preservationist viewpoint, if not also with respect to biodiversity and many measures of ecosystem function (e.g. carbon storage). Thus, if balanced harvesting cannot outperform not fishing economically, then it is clearly inefficient, regardless of how ecological and economic objectives are weighted (Fig. 1). For reasons discussed above, we hypothesize that balanced harvesting might indeed be worse economically

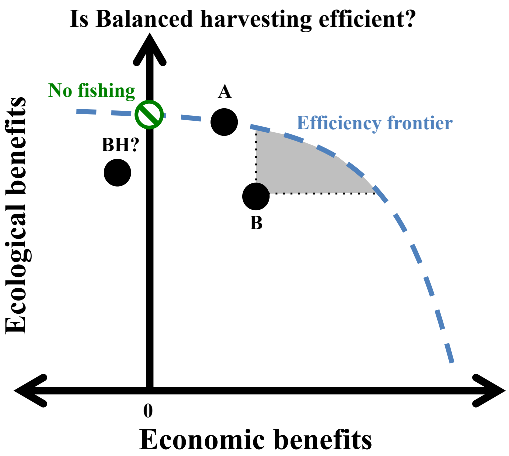

Figure 1 Is balanced harvesting efficient? Even without defined relative weights for ecological and economic fishery objectives, we can rule out strategies that produce inefficient outcomes. A fishery outcome is efficient with respect to the economy and the ecosystem if no other achievable outcome exists that performs at least as well on both dimensions, and better on at least one dimension. In the above diagram, the thick dashed line represents the efficiency frontier – the set of achievable outcomes that are efficient. Point A is efficient, for example, while point B is not, because all outcomes in the shaded region are achievable and better than point B on both dimensions. Assuming that no fishing (hatched) is the best ecological strategy, it is efficient, even though it produces no economic benefits. Our concern is that balanced harvesting (denoted 'BH') may provide negative economic benefits, and thus be inefficient – outperformed by no fishing both ecologically and economically (as illustrated).

than not fishing at all at a global scale and in some systems (e.g. the high seas), although perhaps not in some small-scale fisheries (e.g. Kolding *et al.* 2014). This hypothesis could be tested by future simulation studies informed by economic and ecological data. It is worth noting that the status quo is surely not efficient either by similar logic: Sumaila *et al.* (2012) estimate that current returns to global fishing are cumulatively negative. However, unlike balanced harvesting (or any other fishery management proposal), the desirability of status quo has already been widely rejected by fisheries scholars, often for precisely this reason.

What does balanced harvesting offer global food security?

Food security, specifically meeting a rising demand for protein, has been a cornerstone of the rationale for balanced harvesting (e.g. Zhou *et al.* 2014). The premise is that the extra protein produced through yield gains from balanced harvesting would help to meet the rapidly rising global protein demand (Garcia *et al.* 2012; Zhou *et al.* 2014). As demand for animal protein is projected to roughly double by 2050 (Alexandratos and Bruinsma 2012), increasing fisheries yields in ways that limit or even reduce environmental costs relative to expanded meat production on land could have significant benefits.

However, these increases in global protein demand will be largely driven by increasing per-capita demands associated with increasing wealth, coupled with preferences for protein-rich diets, rather than meeting the basic nutritional needs of a growing population (Fig. 2) (Tilman *et al.* 2011; Tilman and Clark 2014). Because much of the extra yield offered by balanced harvesting would come from sizes and species not currently consumed (Jacobsen *et al.* 2014), the degree to which it would meet a preference-driven demand increase is unclear. High substitutability between species and sizes seems to exist in some communities, particularly in developing countries (e.g. Kolding *et al.* 2014), where fish are also important sources of micronutrients (Béné *et al.* 2015). However, powerful counterexamples also exist. For example, Peru's Direct Human Consumption programme has faced significant challenges because of weak demand for anchovy consumption resulting from cultural dietary habits, despite widespread malnutrition (Fréon *et al.* 2014).

As the growth in animal protein demand with growing wealth will occur in developing countries whose dietary protein is generally more fish-oriented than many of today's largest meat consumers (Tilman and Clark 2014), balanced harvesting may indeed contribute to meeting rising protein demands. Under the best-case scenario, reforming fisheries using single-species approaches is projected to increase fisheries yields by only a few 10s of percent (Costello *et al.* 2012). If yields from balanced harvesting could significantly enhance this potential growth, fisheries could play a far larger role in the future, particularly if different sources of fish were substitutable.

One intriguing option for using some of the increased yields for species and sizes with little current demand is increasing feed availability for aquaculture. However, the extent to which this would reduce the environmental costs of meeting

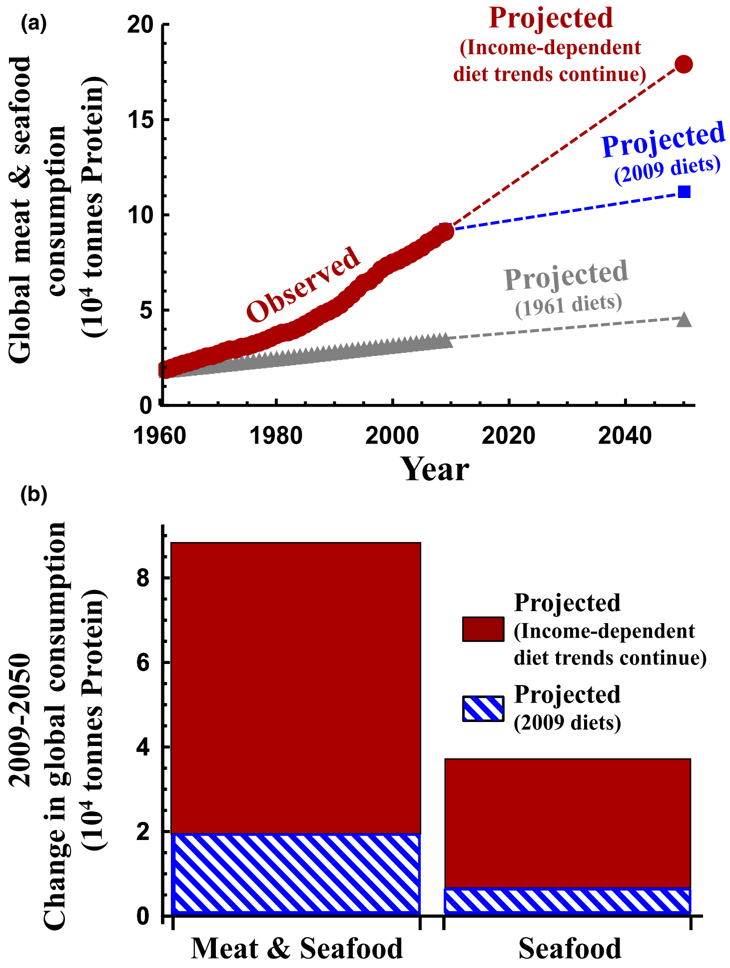

Figure 2 What can balanced harvesting contribute to global food security? Balanced harvesting creates additional animal protein in the face of rapidly rising global demand, but significantly changes the composition of fish available for consumption. Because historical and projected global demands for animal protein have been driven largely by changing consumption patterns associated with wealth, rather than population growth and nutritional needs, the contribution of balanced harvesting to food security may hinge on whether high substitutability exists between fish products. Panel (a) shows past trends (circles) in meat and seafood consumption from 100 of the world's most populous countries and projections of 2050 consumption under: projected population growth and income-dependent diet trends (circle and adjoining dashed line); projected population growth with 2009 diets (square and adjoining dashed line); and past (1961–2009) and projected (2050) population growth with 1961 diets (triangles and adjoining dashed line). Panel (b) shows projected changes in meat and seafood consumption in these countries under: projected population growth and income-dependent diet trends (solid); projected population growth with 2009 diets (hatched). All trends and projections shown are calculated using data deposited and methods described in Tilman and Clark (2014).

future protein demands would depend strongly on the additional effort costs of balanced harvesting, as fuel costs make up a substantial fraction of the greenhouse gas (GHG) costs of wild-caught fish (Tilman and Clark 2014). Due to the high economic and GHG cost of balanced harvesting in distant waters, and the preference-driven nature of global food demands, we suspect that the food security case for balanced harvesting may be stronger at local scales than at a global scale, but more research is needed to resolve this further.

Conclusions

Balanced harvesting challenges the established view in fisheries sciences. It proposes to advance the goals of EBFM by radically altering the selection pattern, harvesting all species and sizes in proportion to their productivity. Paradigm-challenging but specific proposals such as this are important steps towards EBFM, but implementation steps towards balanced harvesting are still premature until several key questions are addressed. First, we must clearly define balanced harvesting both conceptually and in practice. Second, we must answer some important questions to evaluate the conceptual case for balanced harvesting: What sizes should be fished to maximize yields? How realistic are assumptions in size-spectrum models, which suggest trivial size limits, and seemingly predict net competitive relationships between some marine predator species and their prey? What are the costs of harvesting currently unexploited species and sizes, and how do they compare to the gains or losses in terms of yield or revenue? What are the ecological benefits of balanced harvesting and how do we value these? Are there other EBFM strategies that outperform balanced harvesting both ecologically and economically? How large are the potential yield gains from balanced harvesting and what can they contribute to meeting global protein demands?

If a clear case for balanced harvesting emerges from answering these questions, we must evaluate the robustness of projected benefits to likely technological limitations on implementation. If balanced harvesting is determined to be costly but worthwhile, we must determine who pays the cost.

If balanced harvesting is not desirable or feasible at particular scales or locations, could it still be beneficially implemented only partially in some places and not at all in others? For example, it seems likely that the case for global balanced harvesting is weak, but perhaps cases exist for local balanced harvesting in parts of the developing world (e.g. Kolding and van Zwieten 2014). This possibility merits further consideration. The mechanics and ecological merits (if there are any) of partial implementation of balanced harvesting (e.g. harvesting some but not all currently unfished species or sizes in an ecosystem), in the light of surely common technological, economic, social or institutional constraints on full implementation, also merit careful consideration and evaluation.

In general, greater consensus on terms of reference for EBFM is needed, building on recent progress (e.g. Link and Browman 2014; Plagányi *et al.* 2014). The intersecting complexities of the economic, social and ecological objectives of EBFM, and of the social-ecological systems of fisheries themselves, make this understandably challenging. In such a complicated undertaking, the precautionary 'first, do no harm' principle would seem to apply as a basic threshold for evaluating all strategies (Pikitch *et al.* 2004). Balanced harvesting has not yet been shown to pass this test, but is nonetheless an interesting and innovative EBFM strategy that merits the due diligence.

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Conflict of interests

The authors declare no conflict of interests.

References

Alexandratos, N. and Bruinsma, J. (2012) World agriculture towards 2030/2050: the 2012 revision. ESA Working paper No. 12-03. Rome, FAO. Available at: http://www.fao.org/docrep/016/ap106e/ap106e.pdf (accessed 9 January 2015).

Andersen, K.H. and Brander, K. (2009) Expected rate of fisheries-induced evolution is slow. *Proceedings of the National Academy of Sciences of the United States of America* **106**, 11657–11660.

Andersen, K.H., Brander, K. and Ravn-Jonsen, L. (2015) Trade-offs between objectives for ecosystem management of fisheries. *Ecological Applications*. doi:10.1890/14-1209.1

Anderson, C.N.K., Hsieh, C.-H., Sandin, S.A. *et al.* (2008) Why fishing magnifies fluctuations in fish Abundance. *Nature* **452**, 835–839.

Beddington, J.R., Agnew, D.J. and Clark, C.W. (2007) Current problems in the management of marine fisheries. *Science* **316**, 1713–1716.

Béné, C., Barange, M., Subasinghe, R. *et al.* (2015) Feeding 9 billion by 2050 – Putting fish back on the menu. *Food Security* **7**, 261–274.

Beverton, R.J. and Holt, S.J. (1957) *On the Dynamics of Exploited Fish Populations*. HMSO, London, UK.

Borrell, B. (2013) A big fight over little fish. *Nature* **493**, 597–598.

Branch, T.L. and Hilborn, R. (2008) Matching catches to quotas in a multispecies trawl fishery: targeting and avoidance behavior under individual transferable quotas. *Canadian Journal of Fisheries and Aquatic Sciences* **65**, 1435–1446.

Chan, K.M.A., Satterfield, T. and Goldstein, J. (2012) Rethinking ecosystem services to better address and navigate cultural values. *Ecological Economics* **74**, 8–18.

Conover, D.O. and Munch, S.B. (2002) Sustaining fisheries yields over evolutionary time scales. *Science* **297**, 94–96.

Costello, C., Ovando, D., Hilborn, R., Gaines, S.D., Deschenes, O. and Lester, S.E. (2012) Status and solutions for the world's unassessed fisheries. *Science* **338**, 517–520.

De Roos, A.M. and Persson, L. (2002) Size-dependent life-history traits promote catastrophic collapses of top predators. *Proceedings of the National Academy of Sciences of the United States of America* **99**, 12907–12912.

Díaz, S., Tilman, D., Fargione, J. *et al.* (2005) Biodiversity regulation of ecosystem services. In: *Ecosystems and Human Well-Being: Current State and Trends, Volume I: Findings of the Condition and Trends Working Group* (eds R.M. Hassan, R. Scholes and N. Ash). Millennium Ecosystem Assessment, Island Press, Washington, DC, pp. 297–329.

Diekert, F.K., Hjermann, D.Ø., Nævdal, E. and Stenseth, N.C. (2010) Spare the young fish: optimal harvesting policies for North-East Arctic cod. *Environmental and Resource Economics* **47**, 455–475.

Engelhard, G.H., Peck, M.A., Rindorf, A. *et al.* (2014) Forage fish, their fisheries, and their predators: who drives whom? *ICES Journal of Marine Science* **71**, 90–104.

Francis, T.B., Harvey, C.J. and Levin, P.S. (2011) The perils and promise of futures analysis in marine ecosystem-based management. *Marine Policy* **35**, 675–681.

Fréon, P., Sueiro, J.C., Iriarte, F. *et al.* (2014) Harvesting for food versus feed: a review of Peru's fisheries in a global context. *Reviews in Fish Biology and Fisheries* **24**, 381–398.

Froese, R. (2004) Keep it simple: three indicators to deal with overfishing. *Fish and Fisheries* **5**, 86–91.

Froese, R. and Binohlan, C. (2000) Empirical relationships to estimate asymptotic length, length at first maturity and length at maximum yield per recruit in fishes, with a simple method to evaluate length frequency data. *Journal of Fish Biology* **56**, 758–773.

Froese, R. and Pauly, D. (2013) FishBase. Available at: http://www.fishbase.org (accessed 18 November 2014).

Garcia, S.M., Kolding, J., Rice, J. *et al.* (2012) Reconsidering the consequences of selective fisheries. *Science* **335**, 1045–1047.

Hilborn, R. (2011) Future directions in ecosystem based fisheries management: a personal perspective. *Fisheries Research* **108**, 235–239.

Hixon, M.A., Johnson, D.W. and Sogard, S.M. (2014) BOFFFFs: on the importance of conserving old-growth age structure in fishery populations. *ICES Journal of Marine Science* **71**, 2171–2185.

Houle, J.E., Andersen, K.H., Farnsworth, K.D. and Reid, D.G. (2013) Emerging asymmetric interactions between forage and predator fisheries impose management trade-offs. *Journal of Fish Biology* **83**, 890–904.

Jacobsen, N.S., Gislason, H. and Andersen, K.H. (2014) The consequences of balanced harvesting of fish communities. *Proceedings of the Royal Society B* **281**, 20132701.

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

Kolding, J. and van Zwieten, P. (2014) Sustainable fishing of inland waters. *Journal of Limnology* **73**(Suppl. 1), 138–148.

Kolding, J., Béné, C. and Bavinck, M. (2014) Chapter 22. Small-scale fisheries - importance, vulnerability, and deficient knowledge. In: *Governance for Marine Fisheries and Biodiversity Conservation. Interaction and Coevolution* (eds S. Garcia, J. Rice and A. Charles). Wiley-Blackwell, Chichester, Sussex, UK, pp. 317–331.

Law, R., Plank, M.J. and Kolding, J. (2012) On balanced exploitation of marine ecosystems: results from dynamic size spectra. *ICES Journal of Marine Science* **69**, 602–614.

Law, R., Plank, M.J. and Kolding, J. (2014) Balanced exploitation and coexistence of interacting, size-structured, fish species. *Fish and Fisheries* **17**, 281–302.

Law, R., Kolding, J. and Plank, M.J. (2015) Squaring the circle: reconciling fishing and conservation of aquatic ecosystems. *Fish and Fisheries* **16**, 160–174.

Link, J.S. and Browman, H.I. (2014) Integrating what? Levels of marine ecosystem-based assessment and management. *ICES Journal of Marine Science* **71**, 1170–1173.

Link, J.S., Bundy, A., Overholtz, W.J. *et al.* (2011) Ecosystem-based fisheries management in the Northwest Atlantic. *Fish and Fisheries* **12**, 152–170.

Micheli, F., De Leo, G., Butner, C., Martone, R.G. and Shester, G. (2014) A risk-based framework for assessing the cumulative impact of multiple fisheries. *Biological Conservation* **176**, 224–235.

Pikitch, E., Santora, E.A., Babcock, A. *et al.* (2004) Ecosystem-based fishery management. *Science* **305**, 346–347.

Plagányi, É.E., Punt, A.E., Hillary, R. *et al.* (2014) Multispecies fisheries management and conservation: tactical applications using models of intermediate complexity. *Fish and Fisheries* **15**, 1–22.

Polasky, S., Nelson, E., Camm, J. *et al.* (2008) Where to put things? Spatial land management to sustain biodiversity and economic returns. *Biological Conservation* **141**, 1505–1524.

Ricard, D., Minto, C., Jensen, O.P. and Baum, J.K. (2012) Evaluating the knowledge base and status of commercially exploited marine species with the RAM legacy stock assessment database. *Fish and Fisheries* **13**, 380–398.

Sethi, S.A., Branch, T.A. and Watson, R. (2010) Global fishery development patterns are driven by profit but not trophic level. *Proceedings of the National Academy of Sciences of the United States of America* **107**, 12163–12167.

Skern-Mauritzen, M., Ottersen, G., Handegard, N.O. *et al.* (2015) Ecosystem processes are rarely included in tactical fisheries management. *Fish and Fisheries* **17**, 165–175.

Smith, A.D.M., Sainsbury, K.J. and Stevens, R.A. (1999) Implementing effective fisheries-management systems – management strategy evaluation and the Australian partnership approach. *ICES Journal of Marine Science* **56**, 967–979.

Smith, A.D.M., Brown, C.J., Bulman, C.M. *et al.* (2011) Impacts of fishing low-trophic level species on marine ecosystems. *Science* **333**, 1147–1150.

Sumaila, U.R., Cheung, W., Dyck, A. *et al.* (2012) Benefits of rebuilding global fisheries outweigh costs. *PLoS ONE* **7**, e40542.

Tacon, A.G.J. and Metian, M. (2013) Fish matters: importance of aquatic foods in human nutrition and global food supply. *Reviews in Fisheries Science* **21**, 22–38.

Tilman, D. and Clark, M. (2014) Global diets link environmental sustainability and human health. *Nature* **515**, 518–522.

Tilman, D., Balzer, C., Hill, J. and Befort, B.L. (2011) Global food demand and the sustainable intensification of agriculture. *Proceedings of the National Academy of Sciences of the United States of America* **108**, 20260–20264.

Tromble, G.R. (2008) The ecosystem approach to fisheries management in the USA. In: *The Ecosystem Approach to Fisheries* (eds G. Bianchi and H.R. Skjoldal). FAO, Rome, Italy, pp. 301–308.

Trueman, C.N., Johnston, G., O'Hea, B. and MacKenzie, K.M. (2014) Trophic interactions of fish communities at midwater depths enhance long-term carbon storage and benthic production on continental slopes. *Proceedings of the Royal Society B* **281**, 2014 0669.

Worm, B., Hilborn, R., Baum, J.K. *et al.* (2009) Rebuilding global fisheries. *Science* **325**, 578–585.

Zhou, S., Smith, A.D.M., Punt, A.E. *et al.* (2010) Ecosystem-based fisheries management requires a change to the selective fishing philosophy. *Proceedings of the National Academy of Sciences of the United States of America* **107**, 9485–9489.

Zhou, S., Smith, A.D.M. and Knudsen, E. (2014) Ending overfishing while catching more fish. *Fish and Fisheries* **16**, 716–722.