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Trade-offs between objectives for ecosystem management of fisheries

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Abstract. The strategic objectives for fisheries, which are enshrined in international conventions, are to maintain or restore stocks to produce maximum sustainable yield (MSY) and to implement the ecosystem approach, requiring that interactions between species be taken into account and conservation constraints be respected. While the yield and conservation aims are, to some extent, compatible when a fishery for a single species is considered, species interactions entail that MSY for a species depends on the species with which it interacts, and the yield and conservation objectives therefore conflict when an ecosystem approach to fisheries management is required. We applied a conceptual size- and trait-based model to clarify and resolve these issues by determining the fishing pattern that maximizes the total yield of an entire fish community in terms of catch biomass or economic rent under acceptable conservation constraints. Our results indicate that the eradication of large, predatory fish species results in a potential maximum catch at least twice as high as if conservation constraints are imposed. However, such a large catch could only be achieved at a cost of forgone rent; maximum rent extracts less than half of the potential maximum catch mass. When a conservation constraint is applied, catch can be maximized at negligible cost in forgone rent, compared with maximizing rent. Maximization of rent is the objective that comes closest to respecting conservation concerns.

Key words: conservation; ecosystem approach to fisheries management; maximum economic yield; maximum sustainable yield; North Sea; size spectrum model.

Introduction

The development of agriculture over 10 000 years ago by deliberate transformation of terrestrial ecosystems resulted in increased food production from cultivated plants and domesticated animals, and allowed human population growth to accelerate (Harris 1996). In contrast, fishing remains a hunter-gathering activity, reliant on "natural" ecosystems despite the increasing sophistication, power, and efficiency of modern capture methods. With the exception of aquaculture, marine ecosystems have not been deliberately transformed to increase food production, although fishing and other human impacts have caused widespread unintended changes. By the middle of the 20th century, fishing had a global impact on marine ecosystems (Jackson et al. 2001), and after about 1990, the global yield from marine capture fisheries leveled off and began to decline (FAO 2014). A growing human population imposes an increasing demand for food, and the environmental costs of maximizing food production are obvious. Food production and conservation objectives are not fully compatible (Hilborn 2007), and the central task of modern management of global fisheries is to achieve the optimal balance between conservation and production of food and wealth. The aspiration to reconcile high levels of food production with sustainability and conservation is expressed by the maximum sustainable yield (MSY) concept and the "ecosystem approach to fisheries management."

The MSY concept views a fish stock as a production unit whose production should be maximized (in terms of food or wealth) without compromising the reproductive potential of the stock. MSY originated in realpolitik as well as science (Mesnil 2012), and has now become the overarching objective in fisheries management in Europe (EU Regulation 1380/2013, Article 2.2; EU 2013). International commitments to maintain and restore fish stocks to levels that can produce MSY were made at Rio de Janeiro (UN 1992) and Johannesburg (UN 2002). The latter required that MSY should be achieved by 2015. The aspiration to achieve food production and conservation goals is laudable and timely, but the use of a reassuring term like MSY does not of itself resolve the potential conflict between food production and conservation. In fact, it is evident that, in addition to this unresolved conflict, MSY as applied to individual species is not a well-defined objective and provides, at best, incomplete policy guidance for ecosystem sustainability (Gaichas 2008) since the productivity of any species depends on its interactions with predators and prey, which are also affected by fishing.

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The MSY concept is born of a tradition that considers each fish stock in isolation. However, fishing changes the abundances of predators and prey, and thus alters the yield of species affected by those predators and prey (Gislason 1999). Predator–prey interactions affect not only the maximum yield that can be taken from each individual species, but also the total fish yield of the system (Brander and Mohn 1991). Furthermore, when several species are taken in mixed fisheries there is a risk that vulnerable species (those most sensitive to fishing pressure) will be exploited above the level needed to produce MSY, as higher levels of fishing are applied to produce MSY for less vulnerable species. Some vulnerable species will be put at risk of local or global extinction (Dulvy et al. 2003). The single-stock approach to fisheries management is therefore unable to guide strategic management of an ecosystem (May et al. 1979). This realization has resulted in the adoption of the ecosystem approach to fisheries management (Reykjavik Declaration 2001, FAO 2001), which requires that fish production objectives are constrained by conservation objectives and that biological and technical interactions between species are taken into account.

If implementations of the MSY goal do not take biological and technical interactions into account they will be inconsistent with the ecosystem approach. In the EU, the Marine Strategy Framework Directive recognizes this shortcoming in the criteria being developed to define Good Environmental Status (in order to fulfill both MSY and conservation objectives). The scientific report dealing with the issue (Piet et al. 2010) says that decisions about how to resolve the interaction between species are a political, not a scientific, matter, but the response of the EU Commission (EU 2010) was that "Further research is needed to address the fact that [...] MSY may not be achieved for all stocks simultaneously due to possible interactions between them." This need for guidance on how to achieve an acceptable and economically sensible balance between food production and conservation objectives, which takes into account the technical and biological interactions between species, is evident (Gislason 1999, Worm et al. 2009, Voss et al. 2014) and is the subject of the present work.

At least two issues must be considered in order to resolve the conflict between food production and conservation objectives in an ecosystem context. First, what do we mean by MSY when the interactions between species are taken into account? For example, the value of a catch depends on the value of the individual species in the catch; a catch composed of forage fish is usually less valuable than the same biomass of fish for human consumption. The maximization of protein yield will therefore not automatically lead to a maximization of the wealth generated (the maximum economic yield [MEY]; Voss et al. 2014). Second, how can MSY be reconciled with the need to sustain the composition, structure, and function of the ecosystems concerned? These are clearly very difficult questions that could take a long time to work through, but the Johannesburg Declaration requires that MSY should be achieved by 2015.

The kinds of questions our work is intended to address are: 1) What is MSY as applied to a fish ecosystem with biological interactions? and, 2) What patterns of fishing will maximize either total catch or economic rent from such an ecosystem, with and without conservation constraints? We applied a size- and trait-based model that was parameterized to represent the fish and fisheries component of a generic ecosystem akin to the North Sea. Despite the model being a crude simplification of a complex natural ecosystem, it does provide general guidance on how predator–prey interactions shape the yield from the entire fish community of interacting species. Our exploration takes the form of scenarios where management adjusts the fishing pattern to optimize production either in terms of biomass or wealth, while respecting conservation constraints.

Methods

Community model

Size spectrum models provide a simple representation based on the size of individuals, of the dynamics of fish communities (Benoit and Rochet 2004, Andersen and Beyer 2006, Hartvig et al. 2011), and how they respond to fishing (Andersen and Pedersen 2010, Houle et al. 2013). They are based on a few simple and generally accepted assumptions, they contain a small set of species-independent parameters, and they are computationally efficient. The central process in the models is the predation by larger individuals on smaller individuals to fuel growth and reproduction. The models therefore resolve food-dependent somatic growth of individuals, which is neglected in classic food web models based on Lotka-Volterra type of equations. Here we used the previously published trait-based formulation of the size spectrum model, which represents the biomass density of individuals as a function of the size (mass) of individuals w and the asymptotic (maximum) size that the individual may reach W (Fig. 1A; Andersen and Pedersen 2010). The use of a trait (here W) as a continuous variable avoids the need to represent individual species (Norberg et al. 2001). The model employs size-based scaling and life history invariants to reduce the number of parameters to a general set that captures a typical fish community (Pope et al. 2006) such that the model generates quantitative predictions that are not specific to a particular ecosystem. To provide meaningful numbers in the economic part of the model, we have used prices and costs from the fishery in the North Sea. Full details of the concepts, assumptions, equations, and parameters of the model are provided in the Appendix.

Optimal harvesting

Each asymptotic size group is fished with a trawl-like selectivity pattern with a minimum size at 5% of the

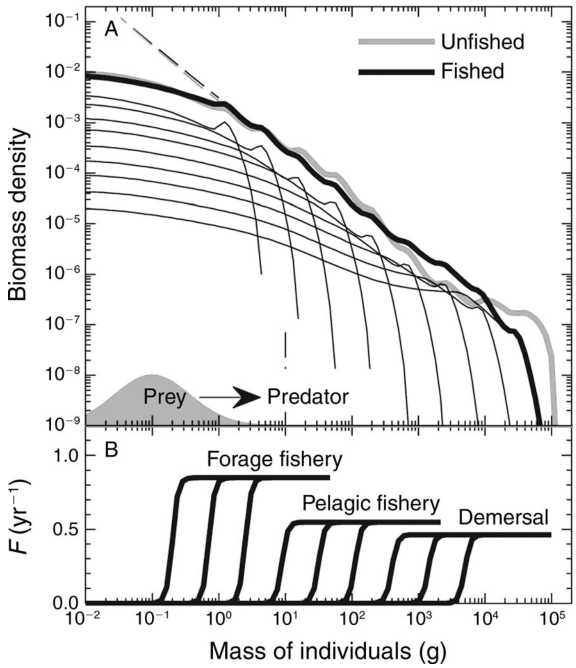

FIG. 1. (A) Example of an un-fished community (gray line) and the impact of fishing with a pattern of fishing mortality that maximizes rent (black line). Thick lines show the size distribution of the entire fish community and a zooplankton resource (dashed) as a function of individual mass. Individuals are characterized by a trait: asymptotic size. Each thin line shows the biomass of individuals from species with a given asymptotic size. The central process in the model is predation by larger predators on smaller prey described by log-normal size-selection function (gray area shown for a 10-g predator). The rest of the model is derived from a bioenergetic budget of energy within each individual fish and from keeping track of mass flows. (B) Fishing is described by a trawl-type size selection where individuals are targeted by the fishery when they reach a fraction of their asymptotic size. The fishing mortality (F) on the nine simulated species is grouped into three fisheries each targeting small (forage fish), medium (pelagic), or large species (demersal).

asymptotic size. The control variables in the model are the fishing mortalities, F_i, generated by three fisheries, each referred to by the subscript i taking values 1 to 3, that target different ranges of asymptotic size groups: a "forage" fishery for fish meal and oil (asymptotic mass $W < 100$ g), a "pelagic" fishery (W ranging from 100 to 5000 g), and a fishery for large "demersal" species ($W >$ 5000 g) (Fig. 1B). F_{mean}, the average of the three fishing mortalities, ranged from 0 to 5 yr^{-1}. The allocation of effort between the three fisheries for any given average fishing mortality level is adjusted to maximize the catch in terms of protein (the total mass of fish caught, Y_{catch}) or rent (the resource rent, Y_{rent}).

Three types of community-level maximization were performed, each for varying levels of average fishing mortality: maximizing catch, maximizing rent, and maximizing catch while preventing the spawning stock biomass of any species from falling below 20% of unfished spawning stock biomass (Table 1).

A simple bioeconomic model was used to calculate resource rent. Resource rent is total revenue minus total costs, and total costs include operational costs (fuel, labor, ice, maintenance, administration, et cetera), depreciation of capital (the loss of capital value during operation) and opportunity cost of capital (the forgone return on capital from the best alternative use, e.g., loan repayment). Rent differs from private profit because the latter does not consider alternative use of capital as a cost. The rent of fishery i is the biomass yield of all size groups $y_i(w)$ multiplied by a size-dependent price $p(w)$ minus the costs C_i of the fishing operation, as follows:

$$R_i = \int_{w_0}^{W_i} y_i(w)p(w)dw - C_i.$$

Price per mass is a function of individual mass: $p \propto w^c$. The exponent was determined by fitting to prices of fish from the North Sea as $c \approx 0.41$ (Appendix: Fig. A1). The cost of fishing was assumed proportional to the effort F_i with a constant of proportionality depending on the catchability of the stock $C_i = aF_iW_i^b$. The parameters a and b in the cost function were determined by reference to an average fishing pattern in the North Sea ($F_i \approx 0.7$ yr^{-1}; Pope et al. 2006). Since all asymptotic size groups are currently fished in the North Sea, we assumed that the rents of the three fishing fleets were approximately similar. This information was used to calibrate the parameter b. Further, it was assumed that overall the fishery is only marginally profitable (Statistics Denmark 2012), i.e., the rent is a small fraction of the revenue, which is used to calibrate a (Appendix: Fig. A2). We explored other cost functions where cost depends on abundance of the targeted stock to confirm

TABLE 1. Specification of objectives for community-level maximizations.

Scenario	Description
1a) "Catch"	Maximizing total catch: $\max_F\{Y_{\mathrm{catch}}\}$ where $Y_{\mathrm{catch}} = \sum Y_i$ is the yield measured in biomass per time from all the three fisheries, and F refers to the fishing mortalities in the three fisheries (i).
1b) "Zero"	The maximal total catch with zero resource rent.
2) "Rent"	Maximizing total resource rent: $\max_F\{Y_{\mathrm{rent}}\}$, where $Y_{\mathrm{rent}} = \sum R_i$ is the sum of the rent R_i from all three fisheries.
3) "Conserve"	Maximizing catch while preventing the spawning stock biomass of any species from falling below 20% of unfished spawning stock biomass.

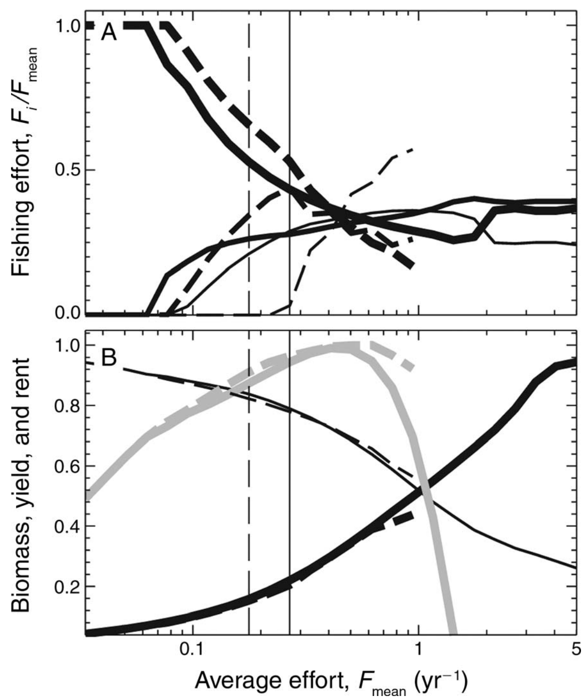

FIG. 2. Steady-state fishing mortality to maximize catch and rent. (A) Steady-state allocation of effort (mortality, F_i) between three fleets targeting small fish (forage species; thin lines), medium (pelagic species; medium lines), or large (demersal species; thick lines) when maximizing catch (solid) or rent (dashed). Panel (B) shows the result of the maximization as a function of the average fishing mortality of the three fleets: fished stock biomass in the community (thin black lines) and total catch Y_{catch} (thick black lines), both normalized by the biomass in the unfished community, and rent (gray lines) normalized by maximum rent. Results are shown when either catch Y_{catch} (solid lines) or rent Y_{rent} (dashed lines) are maximized. The lines showing the results of maximizing rent are cut short beyond the average fishing mortality, F_{mean}, which maximizes rent for clarity (see Appendix: Fig. A3 for full figure). The vertical lines are drawn at the effort where one species drops below 20% of unexploited biomass when maximizing catch (solid) or rent (dashed). Thus, the conservation constraint is violated on the right side of the vertical lines.

that the qualitative results are not overly sensitive to the cost function (Appendix: Fig. A3).

RESULTS

At low average fishing mortality (F_{mean}), both total catch (Y_{catch}) and rent (Y_{rent}) were maximized by allocating all the effort to the demersal (large fish) fishery (Fig. 2A; Appendix: Figs. A3A and A4A). As the average fishing mortality increases, fisheries for medium and small species begin. The pattern of effort allocation between the three fisheries diverges depending on whether the target is to maximize total catch or rent, with rent maximization prioritizing higher valued large fish over lower valued small fish. These differences, however, only have a small impact on total yield and rent (Fig. 2B). For an average fishing mortality, $F_{mean} > 0.3$ yr^{-1}, total catch is maximized when the effort is divided evenly between the three fleets.

Total catch continued to increase until an average fishing mortality was approximately equal to 5 yr^{-1}. This suggests that even in a highly exploited marine ecosystem such as the North Sea, where the average fishing mortality was approximately equal to 0.7 yr^{-1} in the mid 1980s and has since declined to $\sim$0.4 yr^{-1}, the total catch could be increased at least by a factor of two (Fig. 2B). Maximizing total catch requires that all except the smallest and most productive species are fished to extinction (Fig. 3), an ecosystem transformation that resembles agriculture, where removing unwanted predators and competitors maximizes the production of selected plants and herbivorous animals.

The largest species are the first to drop below 20% of unfished biomass (Fig. 3), and thus breach the conservation constraint. Reducing the proportion of average fishing mortality on large species could keep them within the conservation limit; however, this would result in increased predation pressure on medium-sized species (asymptotic mass $\approx$ 100 g) and rapidly cause them to drop below the 20% limit. It is therefore not possible to allocate a higher level of average fishing mortality between the three fisheries while keeping all species above the conservation limits (Appendix: Fig. A4).

DISCUSSION

Comparing the catch and rent between management objectives illustrates the trade-offs between the objectives (Fig. 4): (1) Maximizing the catch has a very high

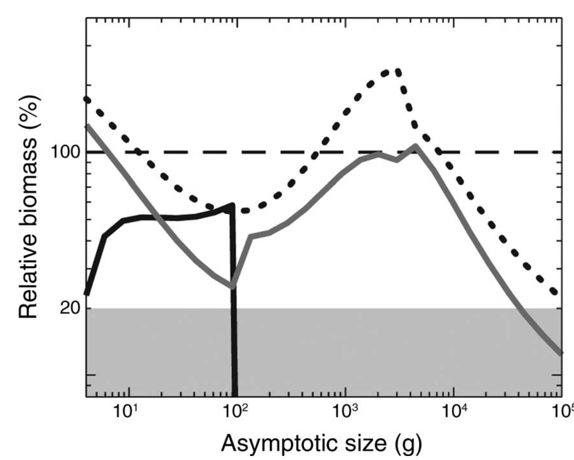

FIG. 3. Impact of optimal fishing on the spawning stock biomass (SSB) relative to the unfished situation (horizontal dashed line). SSB above (below) the black line means higher (lower) biomass than in the un-fished situation. The gray area is where the SSB is below 20% of the unfished SSB. Results are shown for the three fishing patterns maximizing catch (black solid line), rent (gray line), and catch while respecting the conservation constraint that no species is allowed to go below a biomass of 20% of the unfished biomass (dotted line).

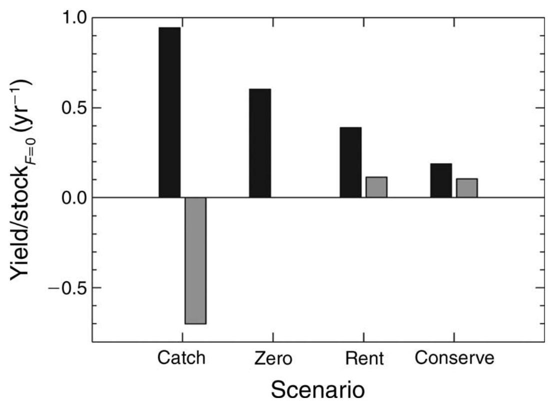

FIG. 4. Total catch (black) and rent (gray) for four possible objectives: maximizing catch (Catch), maximizing catch at zero rent (Zero), maximizing rent (Rent), or maximizing catch keeping SSB about 20% of unfished levels (Conserve; see Table 1 for further clarification of scenarios). Yield is shown relative to the standing stock, measured in biomass or price at zero fishing mortality.

cost; (2) the maximal catch where the rent is not negative occurs at a high average fishing mortality ($F_{\text{mean}} \approx 1.5$ yr^{-1}), and results in a catch about two-thirds of the maximum catch; (3) maximizing rent (at average fishing mortality ≈ 0.5 yr^{-1}) yields about half the maximum catch; and (4) respecting the conservation constraint (average fishing mortality ≈ 0.3 yr^{-1}) gives about one-fourth of the unconstrained maximum catch, but only a small reduction of rent compared to maximum rent. This supports the idea of a "pretty good yield" (Hilborn 2010), where forgoing a small proportion of the maximum yield, in this case in terms of rent, not biomass, results in a significant gain in resilience or conservation. The apparent compatibility of rent maximization with conservation makes it tempting to conclude that if fisheries management were left to unsubsidized market forces the conservation constraint would be self-generated. However, this conclusion ignores two crucial issues: Firstly, individual fishers will maximize their own rent, not the average rent of the entire ecosystem, and thereby fish in combination so all rent is dissipated (Gordon 1954). Secondly, vulnerable species may be more vulnerable to fishing than the "average" species represented in our model, and may require special protective measures (Burgess et al. 2013). Nevertheless, the simulations indicate that rent maximization is more compatible with conservation than is yield maximization. It would be worthwhile investigating whether special measures (selective gear that releases large species, protected areas, spatial planning) can be used to conserve the most sensitive species, since this may allow higher yields from the remaining species. Conservation efforts on land employ a mix of protected areas, spatial planning, and special conservation measures to protect vulnerable species and habitats while maximizing agricultural production (Brussaard et al. 2010).

Other size-based ecosystem models have found that the entire ecosystem collapses at levels of exploitation corresponding to an average mortality of $\sim$1.2 yr^{-1} (Worm et al. 2009) (collapse at a harvest rate $u \approx 0.7$, which corresponds to a fishing mortality of $-\ln(1-u) \approx 1.2$ yr^{-1}), whereas our results indicate that the system can be exploited much harder, albeit with a loss of large species. Why do two seemingly similar models give such qualitatively different results? We believe the difference stems from differences in the strength of coupling between the different species in the models through predation mortality. Both models enforce a balance between growth and inflicted predation mortality: For an individual to achieve a certain growth, a corresponding number of prey have to be eaten. In the Hall et al. (2006) model, the level of predation mortality is furthermore reduced by the introduction of "other food," which is used as a tuning parameter. The applied tuning results in predation mortalities that are much lower than independently estimated predation mortalities (Fig. 3d in Rochet et al. 2011). The parameterization of the model applied in Worm et al. (2009) is therefore representing the ecosystem as a set of weakly coupled single-species models. The low predation mortalities in the Hall et al. model mean that smaller species benefit little from release of predation when the larger species are fished out of the system and thus collapse at a low fishing mortality. When larger species are fished out in our model, the small species are released from predation mortality and can therefore tolerate higher fishing mortality before collapsing.

The results should be interpreted in the light of the limitations of the trait-based model. As with most food web models, we represent the mass flow between different parts of the ecosystems, while ignoring many other effects. We assume that fish species are characterized by just one trait (asymptotic size) and that all individuals are equally desirable targets for fishing. Variability between naturally occurring species with the same asymptotic size means that they have differing sensitivities to fishing (beyond that captured by size) and variability of prices between different species, despite same size, means that they will be targeted with variable intensity. In practice, the targeting of desirable species could lead to the system being taken over by non-desirable species. The model is also unable to resolve how fished species may have increased sensitivities to environmental fluctuations (Anderson et al. 2008). The model predicts that removing large predators can significantly increase biomass production, but in the real world there must be concern that such an impoverished system is liable to switch from a (forage) fish-dominated state to a jelly-dominated state (Richardson et al. 2009), such as was seen in the Black Sea (Daskalov et al. 2007). Other simplifications are the crude representation of fisheries selectivity and the assumption of omnipotent managers in the maximization. Taken together, these caveats mean that the results

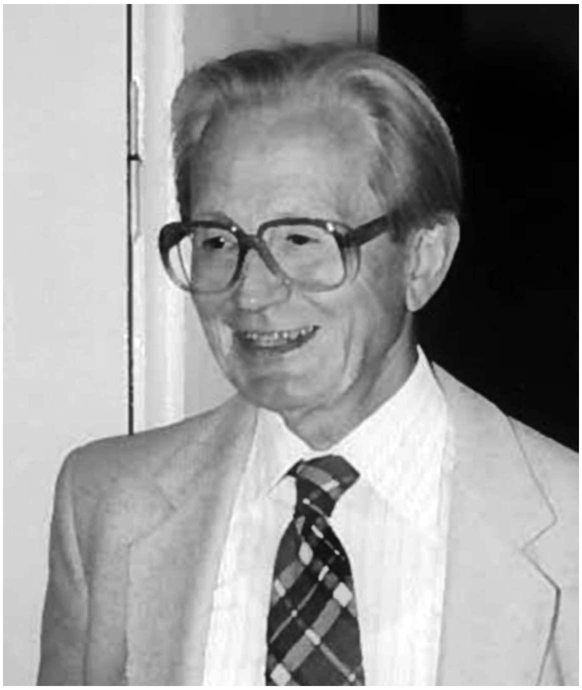

PLATE 1. Erik Ursin died on 14 April 2015 at the age of 92. In the early 1970s he, together with his colleague K. P. Andersen, formulated the first multispecies model of the North Sea ecosystem that incorporated predation and balanced the transfer of nutrients among species. The debt that we and others in this field owe to him is evident and we dedicate this paper to his memory. Erik was a courteous and perceptive debater who was generous in acknowledging the contribution that others made to his work. Keith Brander, Ken H. Andersen, and Lars Ravn-Jonsen. Photo credit: Erik Hoffmann.

should be considered as elaborate *Gendankenexperimente* ("thought experiments") to explore the dynamics of fish production at a strategic and conceptual level rather than as immediate operational management input.

When the costs and economics of fisheries are included, the conflicts between yield and conservation objectives almost disappear; conservation of large fish species can be achieved without foregoing a significant profit. This apparent compatibility between conservation and economic maximization arises because we have assigned higher value to larger individuals than to smaller individuals, as observed in the North Sea. This is, of course, subject to local effects, e.g., highly valued small species like shrimps or anchovies, and to the future price development of fishmeal. While the higher value of large individuals may well be representative of industrially exploited ecosystems in the northern hemisphere such as the North Sea, it is not generally valid (Sethi et al. 2010). In systems where price per mass is independent of body size the economic and conservation objectives are unlikely to be compatible and the result of economic maximization would be more similar to catch maximization.

A maximization of total fish production from the sea entails a transformation of the marine ecosystem to one dominated by small, fast-growing species feeding low down the food chain. This would require continuing costly fisheries to remove predators and would be both uneconomic and unacceptable on conservation grounds and incompatible with exploiting individual populations at or below MSY. With a human population exceeding seven billion and still growing, the forgone protein production that conservation entails is a pressing issue: You cannot have your fish and eat it too. However, the consequences of marine conservation do not stop at the shoreline; demand for marine protein that is not met, because of forgone marine production, is likely to affect terrestrial production systems that face similar conservation concerns (Hall et al. 2013).

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LITERATURE CITED

Andersen, K. H., and J. E. Beyer. 2006. Asymptotic size determines species abundance in the marine size spectrum. American Naturalist 168:54–61.

Andersen, K. H., and M. Pedersen. 2010. Damped trophic cascades driven by fishing in model marine ecosystems. Proceedings of the Royal Society of London B 277:795–802.

Anderson, C. N. K., C. Hsieh, S. A. Sandin, R. Hewitt, A. Hollowed, J. Beddington, R. M. May, and G. Sugihara. 2008. Why fishing magnifies fluctuations in fish abundance. Nature 452:835–839.

Benoit, E., and M.-J. Rochet. 2004. A continuous model of biomass size spectra governed by predation and the effects of fishing on them. Journal of Theoretical Biology 226:9–21.

Brander, K. M., and R. K. Mohn. 1991. Is the whole always less than the sum of the parts? ICES Marine Science Symposia 193:117–119.

Brussaard, L., P. Caron, B. Campbell, L. Lipper, S. Mainka, R. Rabbinge, D. Babin, and M. Pulleman. 2010. Reconciling biodiversity conservation and food security: scientific challenges for a new agriculture. Current Opinion in Environmental Sustainability 2:34–42.

Burgess, M. G., S. Polasky, and D. Tilman. 2013. Predicting overfishing and extinction threats in multispecies fisheries. Proceedings of the National Academy of Sciences USA 110:15943–15948.

Daskalov, G. M., A. N. Grishin, S. Rodionov, and V. Mihneva. 2007. Trophic cascades triggered by overfishing reveal possible mechanisms of ecosystem regime shifts. Proceedings of the National Academy of Sciences USA 104:10518.

Dulvy, N., Y. Sadovy, and J. Reynolds. 2003. Extinction vulnerability in marine populations. Fish and Fisheries 4:25–64.

EU. 2010. Official Journal of the European Union, L 232, section B 3.2.

EU. 2013. Official Journal of the European Union, L 354 22-48. http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2013:354:0022:0061:EN:PDF

FAO [Food and Agriculture Organization]. 2014. Reykjavik conference on responsible fisheries in the marine ecosystem. FAO, Rome, Italy. ftp://ftp.fao.org/fi/DOCUMENT/reykjavik/y2198t00_dec.pdf

FAO [Food and Agriculture Organization]. 2014. The state of world fisheries and aquaculture. FAO, Rome, Italy.

Gaichas, S. K. 2008. A context for ecosystem-based fishery management: Developing concepts of ecosystems and sustainability. Marine Policy 32:393–401.

Gislason, H. 1999. Single and multispecies reference points for Baltic fish stocks. ICES Journal of Marine Science 56:571–583.

Gordon, H. S. 1954. The economic theory of a common property resource: the fishery. Journal of Political Economy 62:124–142.

Hall, S. J., J. S. Collie, D. E. Duplisea, S. Jennings, M. Bravington, and J. Link. 2006. A length-based multispecies model for evaluating community responses to fishing. Canadian Journal of Fisheries and Aquatic Sciences 63:1344–1359.

Hall, S. J., R. Hilborn, N. L. Andrew, and E. H. Allison. 2013. Innovations in capture fisheries are an imperative for nutrition security in the developing world. Proceedings of the National Academy of Sciences USA 110:8393–8398.

Harris, D. R. 1996. Culture. Pages 552–573 *in* D. R. Harris, editor. The origins and spread of agriculture and pastoralism in Eurasia. UCL Press, London, UK.

Hartvig, M., K. H. Andersen, and J. E. Beyer. 2011. Food web framework for size-structured populations. Journal of Theoretical Biology 272:113–122.

Hilborn, R. 2007. Defining success in fisheries and conflicts in objectives. Marine Policy 31:153–158.

Hilborn, R. 2010. Pretty good yield and exploited fishes. Marine Policy 34:193–196.

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

Jackson, J. B., et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293:629–637.

May, R. M., J. R. Beddington, C. W. Clark, S. J. Holt, and R. M. Laws. 1979. Management of multispecies fisheries. Science 205:267–277.

Mesnil, B. 2012. The hesitant emergence of maximum sustainable yield (MSY) in fisheries policies in Europe. Marine Policy 36:473–480.

Norberg, J., D. P. Swaney, J. Dushoff, J. Lin, R. Casagrandi, and S. A. Levin. 2001. Phenotypic diversity and ecosystem functioning in changing environments: A theoretical framework. Proceedings of the National Academy of Sciences USA 98:11376–11381.

Piet, G., et al. 2010. Marine strategic framework directive. Task group 3 report: Commercially exploited fish and shellfish. JRC Scientific and Technical Reports Number 57750. Joint Research Centre, Ispra, Italy. http://ec.europa.eu/environment/marine/pdf/3-Task-Group-3.pdf

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

Richardson, A. J., A. Bakun, G. C. Hays, and M. J. Gibbons. 2009. The jellyfish joyride: causes, consequences and management responses to a more gelatinous future. Trends in Ecology and Evolution 24:312–322.

Rochet, M. J., J. S. Collie, S. Jennings, and S. J. Hall. 2011. Does selective fishing conserve community biodiversity? Predictions from a length-based multispecies model. Canadian Journal of Fisheries and Aquatic Science 68:469–486.

Sethi, S. A., T. A. Branch, and R. Watson. 2010. Global fishery development patterns are driven by profit but not trophic level. Proceedings of the National Academy of Sciences USA 107:12163.

Statistics Denmark. 2012. Account statistics for fishery 2010, Regnskabsstatistik for fiskeri 2010. Statistics Denmark, Copenhagen, Denmark.

UN. 1992. Convention on biological diversity. June, 1992. Rio de Janeiro, Brazil. https://www.cbd.int/doc/legal/cbd-en.pdf

UN 2002. Johannesburg Declaration on Sustainable Development, A/CONF.199/20, Chapter 1, Resolution 1, Johannesburg, September 2002.

Voss, R., M. Quaas, J. Schmidt, and J. Hoffmann. 2014. Regional trade-offs from multi-species maximum sustainable yield (MMSY) management options. Marine Ecology Progress Series 498:1–12.

Worm, B., et al. 2009. Rebuilding global fisheries. Science 325:578–585.

Supplemental Material

Ecological Archives

The Appendix is available online: http://dx.doi.org/10.1890/14-1209.1.sm