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# Accounting for indirect effects and non-commensurate values in ecosystem based fishery management (EBFM)

Kate Richerson<sup>a,b</sup>, Phillip S. Levin<sup>c</sup>, Marc Mangel<sup>b,\*</sup>

<sup>a</sup> MRAG Americas, P.O. Box 1490, Capitola, CA 95010, USA

<sup>b</sup> Center for Stock Assessment Research, Department of Applied Mathematics and Statistics, University of California Santa Cruz, CA 95064, USA

<sup>c</sup> Northwest Fisheries Science Center, NOAA Fisheries, 2725 Montlake Blvd E., Seattle, WA 98112, USA

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#### ABSTRACT

Ecosystem-based fishery management (EBFM) requires taking account of indirect effects (such as habitat destruction, incidental mortality, and competition between the fishery and marine mammals or birds) and dealing with non-commensurate values (such as yield from the fishery and production of offspring by the birds or mammals competing for the same resource). The perspective of EBFM requires that the rate of fishing mortality is less than the value that provides maximum sustainable yield (MSY), but the question is how far below this level should the fishery operate? For this problem in multiobjective programming, simple method of solution was developed and illustrated with the fishery for sandeels (Ammodytes spp.) in the Shetland Islands. The yield from the fishery at a given fishing mortality F is scaled by MSY (so that this quantity increases as fishing mortality increases from 0 to that giving MSY) and the breeding success of predators (black-legged kittiwakes Rissa tridactyla and Arctic terns Sterna paradisaea) at a given fishing mortality is scaled by that in the absence of fishing. The result is two non-dimensional quantities that can be combined into a single value function, which can then be explored or optimized. It is shown that a reduction of only about 20 percent in yield can nearly double the breeding performance of the more sensitive predator. Extensions of the method are discussed; these include the use of maximum economic yield (MEY) and state dependent life history, as implemented by stochastic dynamic programming.

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#### 1. Introduction

After half a century of traditional fishery management, the majority of the world's fisheries are either fully exploited or overexploited, resulting in a myriad of direct and indirect ecosystem effects [1]. While traditional management has aimed to determine the maximum sustainable yield (MSY) or the rate of fishing mortality  $F_{MSY}$  that a population can sustain, economic gains are likely to be maximized when the stock biomass is larger than that which provides MSY [2]. However, fisheries continue to exploit prey further than is economically optimal because of the "race to fish," where individual fishermen maximize their personal gain by outcompeting others [3].

Many scientists, policy-makers, and environmental activists have recommended ecosystem based fisheries management (EBFM), which aims to achieve better management of fisheries by considering larger ecosystem issues [4]. At the core of EBFM is the recognition that management of human activities must take

E-mail address: msmangel@ucsc.edu (M. Mangel).

into consideration more than the target species. One way of operationalizing this recognition lies in market-based strategies that affect consumer preferences [5]. Such social marketing [6] is most noticeably manifested in eco-labeling by fishery certification systems (FCSs) such as those of the Marine Stewardship Council (MSC, http://www.msc.org/). Eco-labels recognize well-managed fisheries that meet standards of sustainability set forth by the FCSs. Fisheries that achieve certification demonstrate management operations that maintain healthy populations of targeted species, protect the integrity of ecosystems, and balance biological, social, and commercial interests. Certified fisheries are entitled to use the distinguishing logo in the marketplace, and are well poised to meet the growing demand for sustainable seafood.

The MSC standard of sustainability is made up of three core principles that are considered to underpin sustainable and well managed fisheries: (1) the sustainable harvest of the target stock; (2) the acceptable impact of the fishery on the ecosystem; and (3) the effectiveness of the fishery management system. MSC criterion 1 is essentially single species management ensuring that the rate of fishing mortality is  $F_{MSY}$  or lower. MSC criterion 2 has a wide range of implications including minimizing bycatch, marine pollution, and habitat destruction. However, the MSC has



<sup>\*</sup>Corresponding author. Tel.: +18314595797.

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yet to describe a clear methodology for gauging acceptable ecosystem impact, and the guidelines state that "relatively few fisheries would have the information needed to address ecosystem issues quantitatively, and usually they will be assessed using surrogates, analogy, general observation, qualitative assessment and expert judgment" [7, p. 44].

Addressing the direct ecosystem impacts of fishing can be fairly straightforward. For example, in many cases harvest of a target species (e.g. Pacific hake Merluccius productus) is constrained by incidental mortality on other species (e.g. rockfish Sebastes spp.) [8]. In other cases, incidental mortality of non-target species has been implicated in the decline of a number of imperiled species including albatross, sea turtle and small cetaceans [9,10]. As a result, if incidental mortality cannot be effectively managed, then it is almost certain that a harvest policy aimed at reducing incidental mortality will lead to levels of fishing mortality less than  $F_{MSY}$ . In this case envisioning the tradeoff between target and non-target species is relatively straightforward, since the units of harvest of the target species (such as numbers of individuals or kg) and of the incidental take of nontarget species are easily made identical. For direct effects such as incidental mortality different management schemes can alter trade-offs. For example, full observer coverage in association with individually transferable quotas appears to reduce bycatch [11].

Addressing indirect impacts on the ecosystem is more complex. For instance, fisheries also have the potential to compete with marine mammals and seabirds for prey [12,13]. The case of a fishery targeting a species that is also a prey species of a marine mammal or bird is complicated because the accounting of the indirect effect is more convoluted (e.g. connecting removal of the target species with the population dynamics of the bird or mammal) and because of non-commensurate values (e.g. comparing kg of fish landed with number of offspring produced by the bird or marine mammal population). For example, in the Barents Sea ecosystem, the collapse of herring stocks as a result of intensive fishing was associated with the decline of Atlantic puffins. After the fishery was closed, herring populations recovered, and puffins showed an immediate response [14].

This paper demonstrates how to account for both indirect effects and non-commensurate values in EBFM, thus moving EBFM one more step forward towards operational implementation. The focus of the method is the target rate of fishing mortality F, which must be reduced from  $F_{MSY}$  for EBFM to be effective. This paper shows how this can be done, allowing all stakeholders to have a common analytical framework when a discussion of harvest rates occurs. This approach could be especially useful in informing FSCs, as well more standard fishery management systems.

#### 2. Methods

A standard age-structured population model [15,16] motivated by the sandeel (*Ammodytes* spp.) fishery in the UK [17] was used to illustrate the ideas set forth in this paper. The fundamental population variable in the model was the number of fish of age *a* in year *t*, denoted by N(a, t), for  $a=0,1,...,a_{max}$ , a known maximum age or plus age group. If Z(a) is the total mortality (fishing and natural) experienced by individuals of age *a*, then the population dynamics for a > 0 are

$$N(a,t) = N(a-1,t-1)e^{-Z(a-1)}$$
(1)

The number of new individuals N(0, t) born into the population at each year was assumed to follow a Beverton–Holt stock-recruitment

relationship

$$N(0,t+1) = \frac{a \cdot B_s(t)}{b + B_s(t)}$$
<sup>(2)</sup>

where *a* and *b* are constants representing maximum per capita reproduction and the strength of density dependence, respectively, and  $B_{s}(t)$  is the spawning biomass in year *t* given by

$$B_{\rm s}(t) = \sum_{a} N(a,t) \cdot W(a) \cdot P_m(a) \tag{3}$$

where  $P_m(a)$  is the probability an individual is mature at age *a* and W(a) is mass at age *a*. For simplicity, knife-edge maturity was assumed.

Total mortality Z for each age class is the sum of fishing mortality and age-specific natural mortality M(a):

$$Z(a) = M(a) + F \cdot S(a) \tag{4}$$

where S(a) is the selectivity of the fishing gear for individuals of age *a*. Age-specific selection by the fishery S(a) was assumed such that

$$S(a) = \frac{e^{(W(a) - W_{50})/\sigma_f}}{1 + e^{(W(a) - W_{50})/\sigma_f}}$$
(5)

where  $W_{50}$  is the mass at which an individual has a 50 percent chance of being removed by fishing and  $\sigma_f$  is the strength of selectivity. Values for M(a) and W(a) were taken from [18]. Recruitment parameters were chosen to give an unfished steady state population on the same order of magnitude as the maximum Shetland area sandeel population discussed in [19] (see [20] for a general description of sandeel stock recruitment in the North Sea).

These dynamics lead to a stable age distribution N(a) independent of time and from that a steady state biomass  $\overline{B}(F)$  that depends upon the rate of fishing mortality F

$$\bar{B}(F) = \sum_{a}^{a_{\max}} N(a) \cdot W(a)$$
(6)

and a steady state yield  $\bar{Y}(F)$  determined by the annual biomass removed by fishing,

$$\bar{Y}(F) = \sum_{a}^{a_{\max}} \bar{N}(a)(1 - e^{-Z(a)}) \frac{F \cdot s(a)}{M(a) + F \cdot s(a)} W(a)$$
(7)

 $F_{MSY}$  is the value of *F* that maximizes  $\bar{Y}(F)$ . Here it was found that  $F_{MSY}$ =1.76 (Fig. 1) at which the steady state biomass of the stock is about 40 percent of its unfished level (Fig. 2).

Once steady state population dynamics are determined, the challenge is to account for the indirect effects on the predator population. In general, this will require some kind of model [21] or extensive fieldwork. An example of the latter is found in [19] in which the breeding success of black-legged kittiwakes (*Rissa tridactyla*), Arctic terns (*Sterna paradisaea*), and two species of skuas (*Stercorarius parasiticus*, *S. skua*) are reported as a function of the estimated biomass *B* of sandeels in the Shetland area. The production of chicks by kittiwakes (chicks per adult breeding pair),  $c_k(B)$ , is a nonlinear function of biomass, given by

$$c_k(B) = 0.3467 \ln(B) - 2.9341 \tag{8}$$

while the chick production by terns (chicks per nest),  $c_t(B)$ , is a linear function of biomass, given by

$$c_t(B) = 4 \times 10^{-6}(B) - 0.0659 \tag{9}$$

Clearly, chick production will be greatest if F=0, because then biomass will be as large as possible. As F increases, chick production will decline, but fishing yield will increase and it is these two non-commensurate quantities that must be compared. At F= $F_{MSY}$ , although kittiwake breeding success is more than 75 percent of breeding success in the absence of fishing (Fig. 3a), tern



**Fig. 1.** Sandeel fishery yield as proportion of maximum sustainable yield in relation to fishing effort.



Fig. 2. Sandeel biomass as proportion of unfished biomass in relation to fishing effort.

breeding success is only about 25 percent of that in the absence of fishing (Fig. 3b).

Choosing the intensity of fishing mortality in this situation is a question of multiobjective decision theory when there are conflicting preferences [22]. A simple solution was found by noting that  $\bar{Y}(F)/\bar{Y}(F_{MSY})$  is a relative measure of yield and  $c_k\bar{B}(F)/c_k\bar{B}(0)$  and  $c_t\bar{B}(F)/c_t\bar{B}(0)$  are the relative production of chicks by kittiwakes and terns when the rate of fishing mortality is *F* and the steady state biomass is  $\bar{B}(F)$ . How these ratios (relative yield, compared to that at *F*=*F*<sub>MSY</sub>, and relative predator performance, compared to that at *F*=0) are to be valued is not a scientific question [23], but that does not mean that they are not amenable



**Fig. 3.** (a) Breeding success of black-legged kittiwakes in relation to fishing effort. At  $F_{MSY}$ , breeding success is more than 75 percent of that in the absence of fishing. (b) Breeding success of Arctic terns in relation to fishing effort. At  $F_{MSY}$  breeding success is about 25 percent of that in the absence of fishing.

to analysis. Because yield and population production have been converted to relative measures, they are now both non-dimensional and commensurate. Assume that the value  $0 \le \alpha_y \le 1$  is assigned to the social value placed on yield and the value  $\alpha_c=1-\alpha_y$  to the social value placed on production of kittiwake and tern chicks. The weighted combined value of (relative) fishery yield and predator success is

$$V(F,\alpha_c) = \alpha_y \left(\frac{\bar{Y}(F)}{\bar{Y}(F_{MSY})}\right) + \frac{\alpha_c}{2} \left(\frac{c_k(\bar{B}(F))}{c_k(\bar{B}(0))} + \frac{c_t(\bar{B}(F))}{c_t(\bar{B}(0))}\right)$$
(10)

Eq. (10) is the key result. It shows how to (i) account for the indirect effect of fishing by converting from absolute yield and



**Fig. 4.** (a) Fishing effort  $F^*(\alpha_c)$  that provides the maximum total social value in relation to social value placed on conservation  $\alpha_c$ . This can also be read as the implicit value placed on conservation at a given fishing level. (b) Fishery yield at  $F^*(\alpha_c)$  relative to maximum sustainable yield. (c) Breeding success when fishing effort is set to  $F^*(\alpha_c)$  for Arctic terns (chicks per pair, dashed line) and black-legged kittiwakes (chicks per nest, dotted line) relative to breeding success when F=0.

chick production to their relative values, normalized by the maximum values that they may take and (ii) to compare noncommensurate values through a non-dimensionalization and weighting. The left hand side of Eq. (10) is termed the social value function. Given a value for  $\alpha_c$  (and thus  $\alpha_y$ ) the level of fishing mortality  $F^*(\alpha_c)$  that optimizes the combination of previously non-commensurate values was determined. From that the individual components, such as relative yield and relative chick production, were computed.

Because an arithmetic average is used in Eq. (10), it cannot account for risk aversion. However, the geometric average does [24], so results for

$$V(F,\alpha_c) = \alpha_y \left(\frac{\bar{Y}(F)}{\bar{Y}(F_{MSY})}\right) + \alpha_c \sqrt{\frac{c_k(\bar{B}(F))}{c_k(\bar{B}(0))}} \frac{c_t(\bar{B}(F))}{c_t(\bar{B}(0))}$$
(11)

were also obtained.

### 3. Results

At  $F_{MSY}$  the steady state biomass of the fished stock was found to be about 40 percent of its unfished level and kittiwake breeding success drops about 25 percent, and tern breeding success drops by nearly 80 percent. One consequence of this is that although  $F^*(\alpha_c)$  monotonically declines as the value placed on conservation increases (Fig. 4a), if the relative importance of conservation is sufficiently high (about 0.75), the level of fishing mortality that maximizes the total value in Eq. (10) drops to 0. Alternatively, Fig. 4a can be read as an implicit value of conservation for a given level of fishing mortality. For example, dropping *F* from *F*<sub>MSY</sub> to *F*=1.0 is equivalent to setting  $\alpha_c$ =0.6 and maximizing the value function in Eq. (1), as can be seen by drawing a line from *F*=1 and seeing where it intersects the curve in Fig. 4a.

Although the level of fishing mortality that maximizes overall value drops steadily with increasing emphasis on conservation values, yield is nearly constant over a wide range of values of  $\alpha_c$  before dropping rapidly, as it must (Fig. 4b). As with Fig. 4a, this figure can also be used to impute a value of  $\alpha_c$  given a reduction of yield from MSY. For example, reducing yield from MSY to 80 percent of MSY imputes a value of conservation of about  $\alpha_c$ =0.7. That is, one can give a weight to conservation to yield of 7–3 and still have 80 percent of MSY.

Furthermore, by increasing  $\alpha_c$  from 0 (no value to conservation) to 0.7, the relative breeding success of kittiwakes increases from about 0.75 to 0.85, and that of terns (the more sensitive species in this example) from slightly more than 0.3 to about 0.55 (Fig. 4c). Thus, a modest reduction in catch can lead to a considerable increase in the breeding performance of the birds, but without the framework in Eq. (10) one could not characterize this trade-off.

When the risk-averse value function of Eq. (2) is used, the same qualitative patterns are obtained although the details change, as one would expect.

#### 4. Discussion

This work has immediate application to both FSCs, where it can be used to quantify the ecosystem impact of fisheries, and to more standard fisheries management systems such as rights-based or catch-share systems. Under such catch-share management, fishermen or groups of fishermen are allocated privileges to harvest a certain portion of the total allowable catch (TAC). These systems allow secure access for fishermen, which provides incentive to maintain larger stocks and allow rebuilding [25–27]. Recent data suggest that such management may halt and in some cases even reverse fishery collapse across a wide range of ecosystems and taxa [28]; however, this approach does not directly consider wider ecosystem implications of the fishery. Our method could be used to provide for ecosystem considerations when deciding the TAC.

There is a subtlety involved, however. Although Eq. (10) is a general framework, using it requires knowledge of reproductive success as a function of the biomass (and possibly the size distribution of biomass) of the target stock/prey species. Had they not been measured, they would have to be computed from a model. In general, fishing changes the age distribution, and thus the size distribution, of fished population. If a fishery is certified but the size distribution of the stock is far from the stable age distribution in the absence of fishing the consequence might be that although there is enough biomass available, it is the wrong kind for some predators. For example, many marine birds take single prey items. Thus the size difference between a 4 and a 10 cm prey item could represent the difference between one and a number of foraging trips for a land-based, central place foraging predator, with considerable consequence for the predator population.

Thus, one next step in this kind of work is to ask the question: does a certified fishery have both *sufficient biomass* and the *right kind* of biomass for predators that depend upon the stock? Those questions can be answered through state dependent life history theory, as implemented by stochastic dynamic programming [29,30]. Another application of this approach that can be developed connects to habitat. For example, bottom trawling reduced the complexity and internal structure of important fish habitat [31]. In the Gulf of Mexico, habitat specific demographic rates for red drum and shrimp are related to the patterns of water use and fisheries (reducing river flow alters sediment transport and this results in changes in marsh nursery habitat [32,33]). With suitable modification, our approach can address this kind of indirect effect as well.

While  $B_{MSY}$  and  $F_{MSY}$  have been successfully used to guide fishery management decisions (for example, see [34]), a further elaboration of this model could be based on the maximum economic yield (MEY) instead of MSY. Because economic gains may be maximized when the stock biomass is larger than that which provides MSY [35], using the MEY to inform decisionmaking will likely bring conservation goals and economic goals into closer alignment.

The model has been intentionally kept deterministic and the parameters certain. However, the fundamental message of this paper (Eqs. (10) and (11)) does not change with either process stochasticity, parameter uncertainty, or both. In the case of process stochasticity, the population dynamics (Eqs. (1) and (2))

will lead to stationary probability distributions for biomass and yield. One then modifies Eqs. (10) and (11) by taking the expectations of the social value function over the distribution of biomass [36]. If parameters in the population dynamics model are also uncertain, then a second average is required over those distributions. In these cases, Eqs. (10) and (11) are interpreted as the expected value of the social value function. If there is learning about parameters or one wishes to move from a stationary probability distribution, then Eqs. (10) and (11) can be evaluated by stochastic dynamic programming.

In conclusion, in the case of fishing on a prey species of a marine mammal or birds, there is no way around the trade-off since regardless of management details, prey are removed from the system. This paper has shown a consistent method to value those effects by putting yield and population production on a commensurate scale.

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