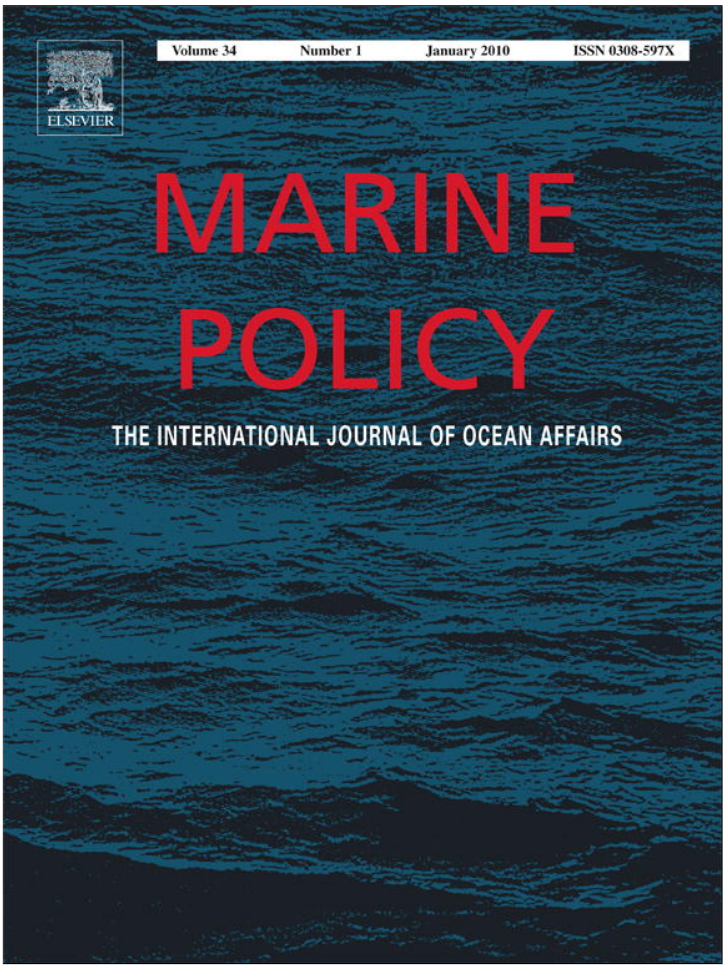
Provided for non-commercial research and education use.
Not for reproduction, distribution or commercial use.



This article appeared in a journal published by Elsevier. The attached copy is furnished to the author for internal non-commercial research and education use, including for instruction at the authors institution and sharing with colleagues.

Other uses, including reproduction and distribution, or selling or licensing copies, or posting to personal, institutional or third party websites are prohibited.

In most cases authors are permitted to post their version of the article (e.g. in Word or Tex form) to their personal website or institutional repository. Authors requiring further information regarding Elsevier's archiving and manuscript policies are encouraged to visit:

http://www.elsevier.com/copyright

Author's personal copy

Marine Policy 34 (2010) 114–119

Contents lists available at ScienceDirect

Marine Policy

journal homepage: www.elsevier.com/locate/marpol

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

Kate Richerson [a,b], Phillip S. Levin [c], Marc Mangel [b,*]

[a] MRAG Americas, P.O. Box 1490, Capitola, CA 95010, USA
[b] Center for Stock Assessment Research, Department of Applied Mathematics and Statistics, University of California Santa Cruz, CA 95064, USA
[c] Northwest Fisheries Science Center, NOAA Fisheries, 2725 Montlake Blvd E., Seattle, WA 98112, USA

ARTICLE INFO

Article history:
Received 9 January 2009
Received in revised form
25 April 2009
Accepted 3 May 2009

Keywords:
Fisheries
Seabirds
Indirect effects
Multiobjective decision-making
Non-commensurate values

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.

© 2009 Elsevier Ltd. All rights reserved.

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 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

*Corresponding author. Tel.: +1 831 459 5797.
E-mail address: msmangel@ucsc.edu (M. Mangel).

0308-597X/$ - see front matter © 2009 Elsevier Ltd. All rights reserved.
doi:10.1016/j.marpol.2009.05.001

Author's personal copy

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 trade-off 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 non-target 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

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

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

$$N(0, t+1) = \frac{a \cdot B_s(t)}{b + B_s(t)} \tag{2}$$

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

$$B_s(t) = \sum_a N(a,t) \cdot W(a) \cdot P_m(a) \tag{3}$$

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

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

where W_{50} is the mass at which an individual has a 50 percent chance of being removed by fishing and σ_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 $\bar{B}(F)$ that depends upon the rate of fishing mortality F

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

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

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

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

Author's personal copy

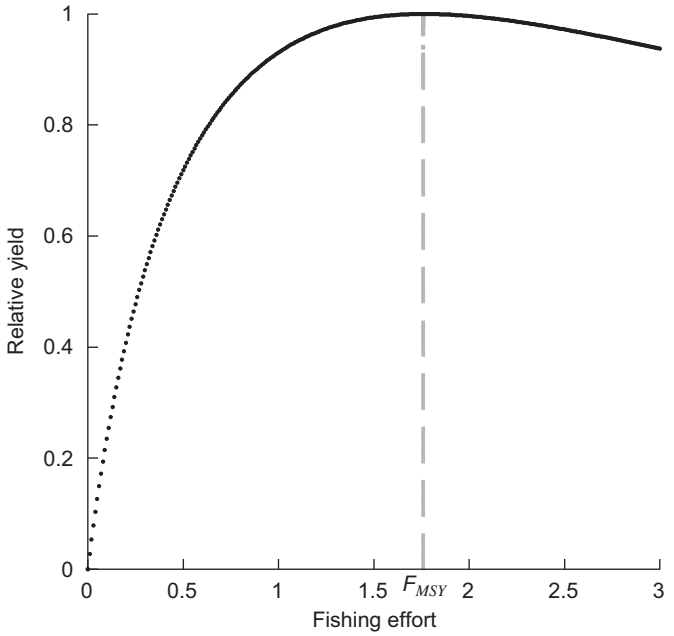


Fig. 1. Sandeel fishery yield as proportion of maximum sustainable yield in relation to fishing effort.

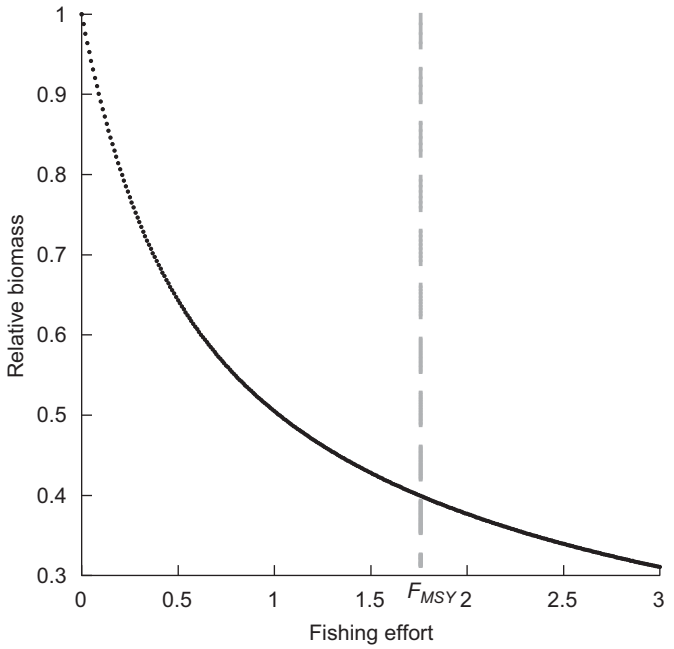


Fig. 2. Sandeel biomass as proportion of unfished biomass in relation to fishing effort.

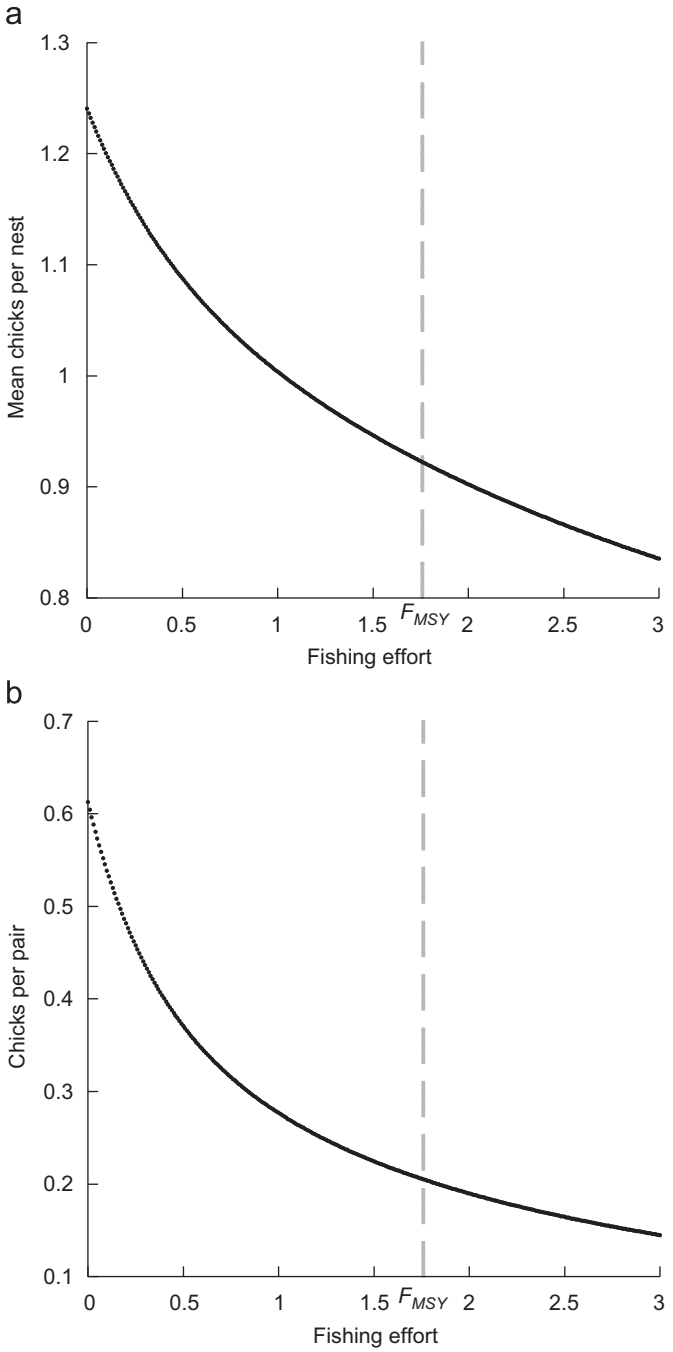


Fig. 3. (a) Breeding success of black-legged kittiwakes in relation to fishing effort. At F_{MSY}, breeding success is more than 75 percent of that in the absence of fishing. (b) Breeding success of Arctic terns in relation to fishing effort. At F_{MSY}, breeding success is about 25 percent of that in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

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_{MSY}; 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 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

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

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

Author's personal copy

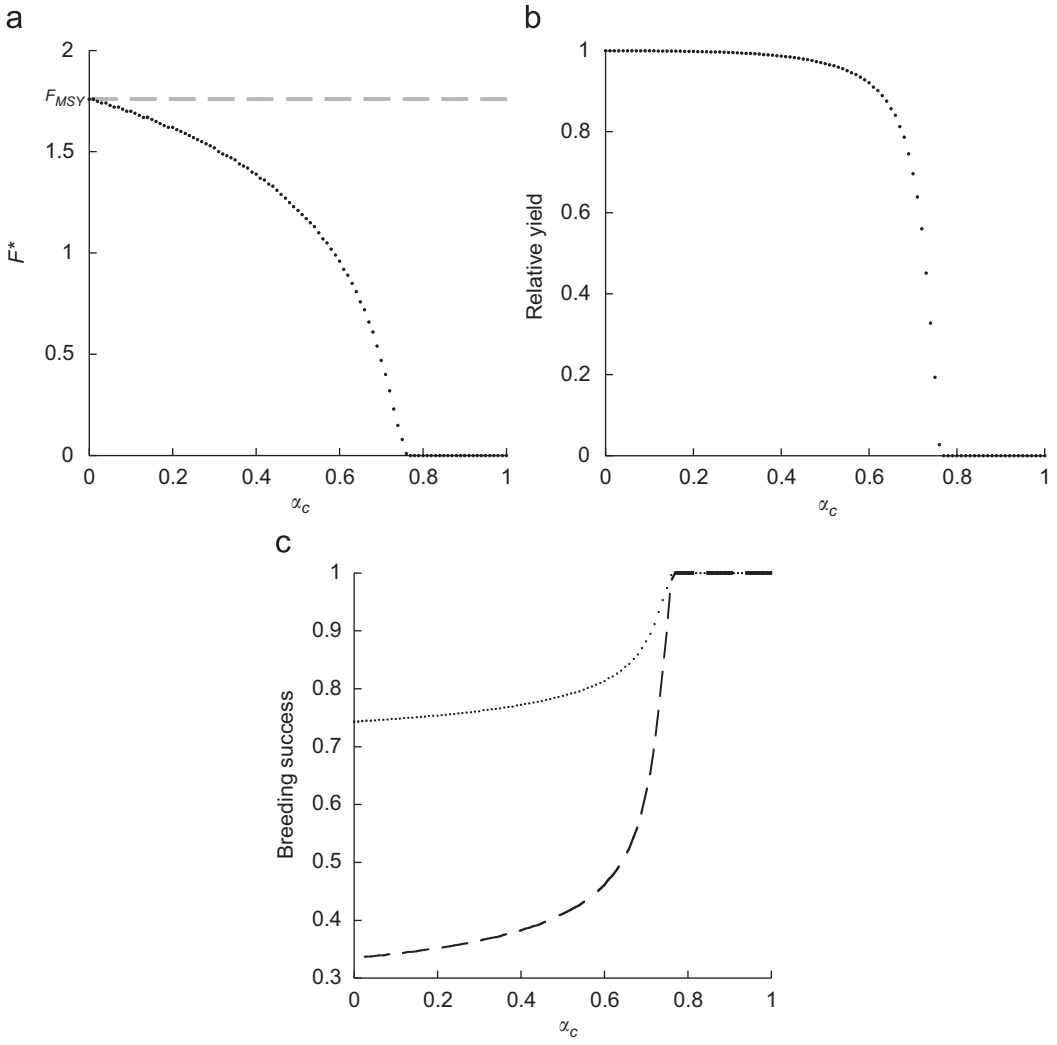


Fig. 4. (a) Fishing effort $F^*(\alpha_c)$ that provides the maximum total social value in relation to social value placed on conservation α_c. This can also be read as the implicit value placed on conservation at a given fishing level. (b) Fishery yield at $F^*(\alpha_c)$ relative to maximum sustainable 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 non-commensurate values through a non-dimensionalization and weighting. The left hand side of Eq. (10) is termed the social value function. Given a value for α_c (and thus α_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

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

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_{MSY} to F=1.0 is equivalent to setting α_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 α_c before dropping rapidly, as it must (Fig. 4b). As with Fig. 4a, this figure can also be used to impute a value of α_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 α_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 α_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

Author's personal copy

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 decision-making 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.

Acknowledgments

This work was partially supported by the Lenfest Ocean Program, the Center for Stock Assessment Research (UCSC), and NOAA Fisheries. We thank Keith Sainsbury for his comments.

References

[1] Food and Agriculture Organization of the United Nations. The state of world fisheries and aquaculture 2006. Rome: FAO Fisheries Department; 2007.
[2] Grafton RQ, Kompas T, Hilborn RW. Economics of overexploitation revisited. Science 2007;318:601.
[3] Clark CW. The worldwide crisis in fisheries: economic models and human behavior. Cambridge, UK: Cambridge University Press; 2007.
[4] Sissenwine M, Murawsi S. Moving beyond 'intelligent tinkering': advancing an ecosystem approach to fisheries. Marine Ecology Progress Series 2004;274:291–5.
[5] Wessells CR, Johnston RJ, Donath H. Assessing consumer preferences for ecolabeled seafood: the influence of species, certifier, and household attributes. Journal of Agricultural Economics 1999;81:1084–9.
[6] Kotler P, Zaltman G. Social marketing: an approach to planned social change. Journal of Marketing 1971;35:3–12.
[7] MSC. Marine Stewardship Council fisheries assessment methodology and guidance to certification bodies: default assessment tree, performance indicators, and scoring guideposts. MSC; 2008.
[8] Harvey CJ, Gross K, Simon VH, Hastie J. Trophic and fishery interactions between Pacific hake and rockfish: effect on rockfish population rebuilding times. Marine Ecology Progress Series 2008;365:165–76.
[9] Lewison RL, Crowder LB, Read AJ, Freeman SA. Understanding impacts of fisheries bycatch on marine megafauna. Trends in Ecology and Evolution 2004;19:598–604.
[10] Kaplan IC. A risk assessment for Pacific leatherback turtles (*Dermochelys coriacea*). Canadian Journal of Fisheries and Aquatic Sciences 2005 1710–19.
[11] Branch TA, Rutherford K, Hilborn R. Replacing trip limits with individual transferable quotas: implications for discarding. Marine Policy 2006;30:281–92.
[12] Furness RW. Impacts of fisheries on seabird communities. Scientia Marina 2003;67(Suppl. 2):33–45.
[13] Demaster DP, Trites AW, Clapham P, Mizroch S, Wade P, Small, RJ, et al. The sequential megafaunal collapse hypothesis: testing with existing data. Progress in Oceanography 2006;68:329–42.
[14] Jennings S, Kaiser MJ, Reynolds JD. Marine fisheries ecology. Malden, MA: Blackwell Publishing; 2001.
[15] Quinn TJ, Deriso RB. Quantitative fish dynamics. New York: Oxford University Press; 1999.
[16] Ralston S, O'Farrell MR. Spatial variation in fishing intensity and its effect on yield. Canadian Journal of Fisheries and Aquatic Science 2008;65:588–99.
[17] Furness RW. Management implications of interactions between fisheries and sandeel-dependent seabirds and seals in the North Sea. ICES Journal of Marine Science 2002;59:261–9.
[18] Poloczanska ES, Cook RM, Ruxton GD, Wright PJ. Fishing vs. natural recruitment variation in sandeels as a cause of seabird breeding failure at Shetland: a modelling approach. ICES Journal of Marine Science 2004 788–97.
[19] Furness RW. Responses of seabirds to depletion of food fish stocks. Journal of Ornithology 2007;148(Suppl. 2):S247–52.
[20] ICES Living Resources Committee. Report of the study group on recruitment variability in North Sea planktivorous fish (SGRECVAP). ICES SGRECVAP Report; 2007.

Author's personal copy

[21] Furness RW. Energy requirements of seabird communities: a bioenergetics model. Journal of Animal Ecology 1978;47:39–53.
[22] Cohon JL. Multiobjective programming and planning. San Diego, CA: Academic Press; 1978.
[23] Ludwig D, Mangel M, Haddad B. Ecology, conservation, and public policy. Annual Reviews in Ecology and Systematics 2001;32:481–17.
[24] Mangel M, Marinovic B, Pomeroy C, Croll D. Requiem for ricker: unpacking MSY. Bulletin of Marine Science 2002;70:763–81.
[25] Beddington JR, Agnew DJ, Clark CW. Current problems in the management of marine fisheries. Science 2007;316:1713–16.
[26] Hilborn R. Defining success in fisheries and conflicts in objectives. Marine Policy 2007;31:153–8.
[27] Grafton RQ, Arnason R, Bjorndal T, Campbell D, Campbell HF, Clark, CW, et al. Incentives-based approaches to sustainable fisheries. Canadian Journal of Fisheries and Aquatic Sciences 2006;63:699–710.
[28] Costello C. Can catch shares prevent fisheries collapse?. Science 2008 1678–81.
[29] Houston AI, McNamara JM. Adaptive models of behaviour. Cambridge, UK: Cambridge University Press; 1999.
[30] Clark CW, Mangel M. Dynamic state variable models in ecology: methods and applications. New York: Oxford University Press; 2000.
[31] Schwinghammer P, Guigné JY, Siu WC. Quantifying the impact of trawling on benthic habitat structure using high resolution acoustics and chaos theory. Canadian Journal of Fisheries and Aquatic Sciences 1996;53:288–96.
[32] Rozas LP, Minello TJ, Zimmerman RJ, Caldwell P. Nekton populations, long-term wetland loss, and the effect of recent habitat restoration in Galveston Bay, Texas, USA. Marine Ecology Progress Series 2007;344:119–30.
[33] Levin PS, Stunz GW. Habitat triage for exploited fishes: Can we identify essential ''Essential Fish Habitat?''. Estuarine, Coastal and Shelf Science 2005;64:70–8.
[34] North Pacific Fishery Management Council. Fishery management plan for groundfish of the Bering Sea and Aleutian Islands management area. Available at 〈alaskafisheries.noaa.gov/npfmc/fmp/bsai/BSAI.pdf〉; 2009.
[35] Grafton RQ, Kompas T, Hilborn RW. Economics of overexploitation revisited. Science 2007;318:601.
[36] Mangel M. The theoretical biologist's toolbox. Cambridge, UK: Cambridge University Press; 2006.