Ecosystem Simulations of Management Strategies for Data-Limited Seamount Fisheries

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Abstract

Traditional fisheries stock assessment requires large amounts of information, mainly from long-term data series, a requirement that is hard to apply to new or poorly documented fishing grounds. With the collapse of traditional shelf stocks and a general decline of global catches, the fishery industry has moved to alternative fishing grounds and species—a process of serial depletion. Seamounts are among those newly targeted ecosystems. In this paper we investigate if ecosystem simulations can help researchers understand the impact of fishing on pristine seamounts. Using ecosystem modeling tools, data gathered from elsewhere, and methods that search for optimal fishing policies, we explore what types of fisheries might be sustainable on seamounts. Although the analyses in this paper are not meant to describe actual fisheries for seamounts, some generalizations can be made. Simulations with policy objectives that maximize economic performance favor fleet configurations based on deepwater trawling, but entail a cost to biodiversity. Maximizing ecological performance favors fleets based on small pelagic and bottom longline fisheries, and maximizes biomass of long-lived species and biodiversity, but sacrifices total catches and jobs. The overall study suggested that sustainable seamount fisheries with tolerable ecosystem impacts appear to be closer to those found by maximizing an "ecological" objective function.

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Introduction

With global catches declining since the late 1980s (Watson and Pauly 2001), the world's fisheries resources have been characterized as seriously depleted or in danger of depletion (e.g., Jackson et al. 2001, Pauly et al. 2002, Baum et al. 2003, Christensen et al. 2003, Myers and Worm 2003), with very little evidence for any recovery (Hutchings 2000). What caused this phenomenon has been the subject of serious debate (Pitcher 2001). Indeed, poor management practices and increased fishing pressure (Ludwig et al. 1993), along with an excessive level of investment in fishing capacity, have resulted in serious stock depletion on continental shelves and have created serial depletion from new pressures on alternative fishing grounds (Pauly et al. 2002). Seamounts are among those "newly" targeted ecosystems that have been intensively fished since the second half of the twentieth century (Rogers 1994, Koslow et al. 2000).

Deepwater fisheries in general, and seamount fisheries in particular, are usually characterized by a boom and bust sequence (Koslow et al. 2000, Watson and Morato 2004), with the targeted fish stocks showing signs of overexploitation within a short period after the beginning of the fishery. For example, this has been the case with the orange roughy (*Hoplostethus atlanticus*) fishery off of New Zealand (Clark 1999, Clark et al. 2000), Australia (Koslow et al. 2000), Namibia (Boyer et al. 2001), and the North Atlantic (Branch 2001); the pelagic armourhead (*Pseudopentaceros wheeleri*) fishery over seamounts in international waters off Hawaii (Sasaki 1986); and the blue ling (*Molva dipterygia*) fishery in the North Atlantic (Bergstad et al. 2003). Seamount aggregating fish stocks are long-lived, slow growing species, with late maturity and low recruitment rates (Koslow 1997, Rico et al. 2001, Morato et al. 2004), often forming highly localized aggregations (Clark 1996). Thus, seamount fish stocks are rapidly depleted and maintenance of seamount fisheries depends on the discovery of unexploited seamounts. Moreover, many seamounts are located in international waters where no management is applied. Once depleted, seamount populations likely require decades to recover (Koslow 1997). Side effects caused by overfishing or extensive trawling on seamounts raise serious concerns: for example, damage to benthic communities dominated by corals and other fragile suspension feeders is common (Richer de Forges et. al. 2000, Koslow et al. 2001), as well as impacts on transient migratory species whose life histories rely on seamount food webs (Haney et. al. 1995, Holland et al. 1999, Weimerskirch et al. 2002). The prevention of further negative impacts on these sensitive ecosystems is now an important policy objective (Probert 1999, Roberts 2002).

There is rising concern about threats to seamount ecosystems in the Exclusive Economic Zones of coastal states and the high seas; several countries, such as Canada, Australia, New Zealand, and Portugal have begun to take action for the protection of such "fragile" ecosystems.

However, seamount ecosystems remain one of the worst cases of data-limited situations, comprising a true challenge for fishery scientists and managers who are urged to develop new fisheries under the precautionary approach of the Code of Conduct for Responsible Fisheries (FAO 1995). Little is known about seamount ecosystems in the Northeast Atlantic and elsewhere, or the impact of human activities upon them. A recent attempt to tackle this global lack of information has been made by the European Commission, which has funded the first European Seamount Project to integrate physical, biogeochemical, and biological research—the "OASIS project" (OceAnic Seamounts: an Integrated Study).

In this paper we investigate if ecosystem simulations can help in understanding the impact of fishing on pristine seamounts. By using ecosystem modeling loosely structured on North Atlantic case studies, data gathered from elsewhere, and optimization methods for policy search, we explore the types of fisheries that might be sustainable on seamount ecosystems.

Methods

Trophic model of seamount ecosystems

In this study, we used a general ecosystem model developed for North Atlantic seamounts (Morato and Pitcher 2002) based on the "Ecopath with Ecosim" approach (EwE; http://www.ecopath.org), a software for ecosystem trophic mass-balance analysis (Ecopath), with a dynamic modeling capability (Ecosim) (for details see Christensen and Walters 2004). This model was developed for a theoretical isolated seamount. Habitat covered by the model was defined by the summit, set at around 300 m, down to the base of the seamount at around 2,000 m. The model covered a small area of about 3,000 km^2 and included 37 functional groups: three marine mammal groups (i.e., toothed whales, baleen whales, and dolphins), seabirds, turtles, seven invertebrate groups (i.e., benthic filter feeders, such as corals or gorgonians, benthic scavengers, benthic crustaceans, pelagic crustaceans, seamount resident cephalopods, small and large drifting cephalopods), three zooplankton groups (i.e., gelatinous, shallow, and deepwater zooplankton), primary producers (i.e., phytoplankton), detritus, and twenty fish groups assembled according to their environmental preferences (i.e., depth and habitat: e.g., benthic, pelagic, or benthopelagic), body size, energetics, and life-history characteristics (see Appendix 1 and Morato and Pitcher 2002 for a complete description of the model).

The theoretical seamount was assumed to have a low initial level of exploitation and its fisheries were loosely based on those operating at the Azores/Mid-Atlantic Ridge comprising 6 fleets (see Appendix 1 and Morato et al. 2001): (a) demersal longline (targeting shallow water demersal and benthic fishes); (b) deepwater longline (targeting bathypelagic and bathybenthic fishes); (c) small pelagics fishery (for small pelagic fishes); (d) tuna fishery; (e) swordfish fishery; and (f) deepwater trawl (targeting seamount associated fishes, including orange roughy and alfonsinos, *Beryx splendens* and *B. decadactylus*). Landings, prices, and job estimates were loosely based on the Azores case study (Morato et al. 2001, Morato and Pitcher 2002).

Ecopath outputs are known to be very sensitive to the vulnerability parameters (see Walters et al. 1997, Pitcher and Cochrane 2002). In this study a standard value of 0.3 representing mixed predator/prey control was used. A brief sensitivity analysis of the policy simulations to different vulnerability settings was conducted by repeating simulations with vulnerabilities of 0.2, 0.3, and 0.5. Results from simulations were generally consistent between different vulnerabilities.

Model analyses

The impacts of alternative time patterns of fishing mortalities were explored using an optimization method (Walters et al. 2002, Christensen and Walters 2004), to search for patterns of relative fishing effort by fishing fleets, which would maximize one or more of the considered objectives:

1. "Economy," or net present economic value (i.e., total present value of the catch).
2. "Jobs," or employment (i.e., a social indicator, assumed proportional to gross landed value of catch for each fleet with a different jobs/landed value ratio for each fleet).
3. "Ecology," or ecological "stability" (i.e., measured by assigning a weighting factor to each group based on their longevity, and optimizing for the weighted sum).

Net present economic value of landed catches was calculated as the discounted sum over all fleets and times of catches multiplied by the prices of landed fish species. A discount rate of 0.04 was used. The ecological criterion component is based on Odum's (1971) definition of "maturity," with mature ecosystems being dominated by large, long-lived organisms. Thus, it is intended to identify the fleet structure that maximizes biomass of long-lived organisms, defined by the inverse of their production/biomass ratios (P/B).

The objective function can be thought of as a "multi-criterion objective function," represented as a weighted sum for the three above-mentioned criteria indicators:

$$OBJ = W_V \times \sum NPV_{ijt} + W_j \times \sum J_{ijt} + W_E \times \sum (B/P)_{ijt} + \varepsilon$$

where W = weighting factors; V = value; J = jobs; E = ecology; I = fleets; j = species caught; t = time in years; NPV = net present value; and B/P = biomass production ratio, assumed to be proportional to species longevity and thus ecological stability, with ε a normally distributed error term.

The Davidon-Fletcher-Powell (DFP) nonlinear optimization procedure was used to iteratively optimize the three above-mentioned objectives by changing relative fishing rates (F) (Walters et al. 2002). This search procedure results in what control systems analysts call an "open loop policy"; a recommendation for what to do at different future times without reference to what the system actually ends up doing along the way (Christensen and Walters 2004). The resulting "optimum" fishing rates by year/fleet served as input for the dynamic simulation, "Ecosim," where they replaced the baseline relative efforts by fleet/gear type. Ecosim was then run for a 50-year period to simulate the effect of the optimized fishing rates and to estimate the biomass, catch, and value variation. These two scenarios were compared with a "no fishing" scenario where all the fishing rates were set to 0.

Nonlinear optimization methods, such as DFP, can be difficult to use and can be misleading. In particular, the method can "hang up" on a local maximum and can give extreme answers due to an inappropriate objective function (Walters et al. 2002). To check for false convergence to local maxima, random starting Fs were used. To search for trade-offs among objective functions, optimal scenario solutions for a range of weightings of ecological and economic objectives were accessed. Additionally, at the end of each run, ecosystem indicators such as the mean trophic level of the catch (see equation in Pauly et al. 1998) and biodiversity index (modified from Kempton and Taylor 1976, Q75; Ainsworth and Pitcher 2004) which resulted from the suggested fishing effort in each range of weighting), were estimated.

Results

Optimal fishing scenarios

The optimized fishing rates (F) for the "economy" and "ecology" objectives, expressed as proportions of the base model fishing rates are summarized in Fig. 1. Maximizing economic value led to an increase in fishing rate in all fisheries (deepwater trawl, $F_{final}/F_{base} = 97.6$; swordfish fisheries ($F_{final}/F_{base} = 18.1$); small pelagic fisheries, $F_{final}/F_{base} = 16.2$), except for the deepwater longline, where the fishing rate was reduced to 0.86 of the base model value. In contrast, maximizing "ecosystem" stability led to a large decrease in all fishing rates (swordfish fisheries. ($F_{final}/F_{base} = 0.01$; demersal longline, $F_{final}/F_{base} = 0.16$; deepwater trawl, $F_{final}/F_{base} = 0.32$).

The effects of the optimized fishing rates on biomass (i.e., percentage of biomass change from base model) after a 50-year simulation are

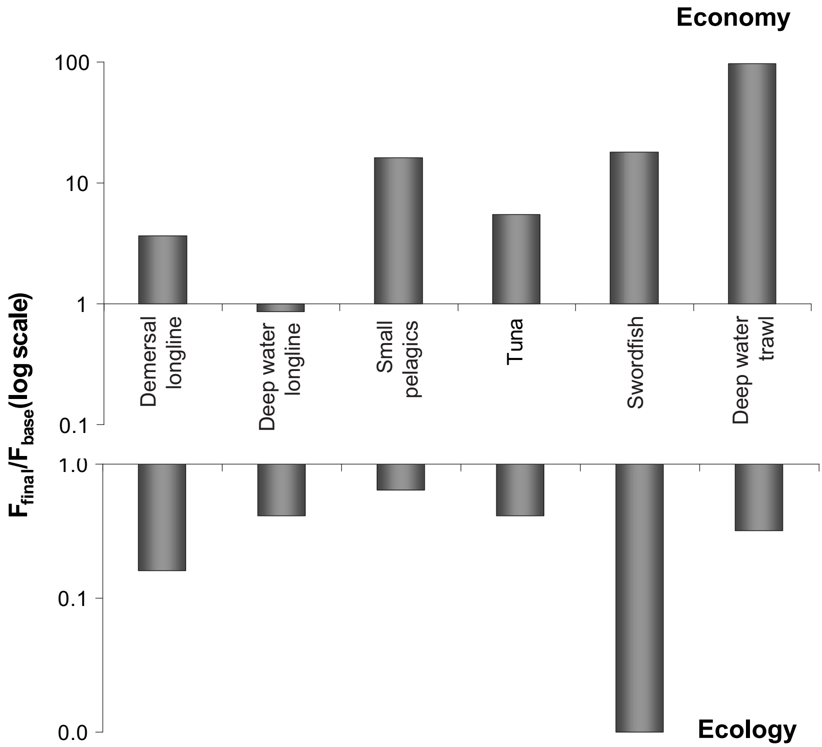

Figure 1. Optimized fishing rates (*F*), expressed as proportions of the base model, obtained by maximizing "economy" and "ecology" objectives. Note differences in scale.

presented, together with a "no fishing" scenario, in Fig. 2. Not surprisingly, the "no fishing" scenario produced a general increase in biomass for most of the groups, but particularly for sea turtles, rays and skates, and pelagic sharks. However, this was not the case for the most important prey groups of the system: mesopelagic fish and benthic invertebrates. The optimized fishing rates for the "ecology" objective function produced very similar results when compared with the "no fishing" scenario, producing a large increase in groups that have slow turnover and higher trophic levels. When economic value was maximized, a general decrease in biomass was observed associated with collapse of ten functional groups (pelagic sharks, tunas, benthopelagic sharks, seamount-associated fishes, bathypelagic fishes, sea turtles, rays and skates, alfonsinos,

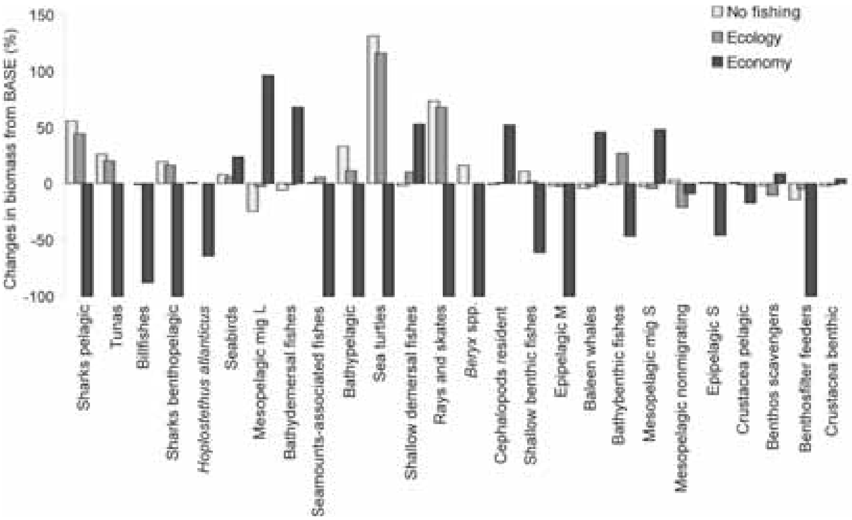

Figure 2. Changes in group biomasses (percent change of biomass from base model) under the three different fishing scenarios: no fishing, maximizing the "ecology" objective, and maximizing the "economy" objective. S = small; M = medium; L = large.

medium-sized epipelagic fish, and benthic invertebrate filter feeders, e.g., deep-sea corals).

The effects of the different fishing policies on the total landed catches are shown in Fig. 3. Maximizing the "economy" objective led to an increase in landings when compared to the base model. In this scenario, the deepwater trawl fishery was favored and had the highest contribution to the total catch. In contrast, maximizing the "ecology" objective required an overall decrease in catches and fishery operations conducted mostly by small pelagic and bottom longline fishing fleets.

Comparing the total value of the catches for the three scenarios (i.e., base model, maximizing "economy," and maximizing "ecology"; Fig. 4), maximizing "economy" generated more money than the base model and the "ecology" scenario. In all cases, deepwater trawl and bottom longline fishing fleets contributed the most to the total value.

Trade-offs

Surface plots of optimal scenario solutions for a range of weightings of ecological and economic objective functions are shown in Figs. 5-7. They show that it was not possible to maximize the performance of all three objectives (i.e., net economic value, number of jobs, and ecological

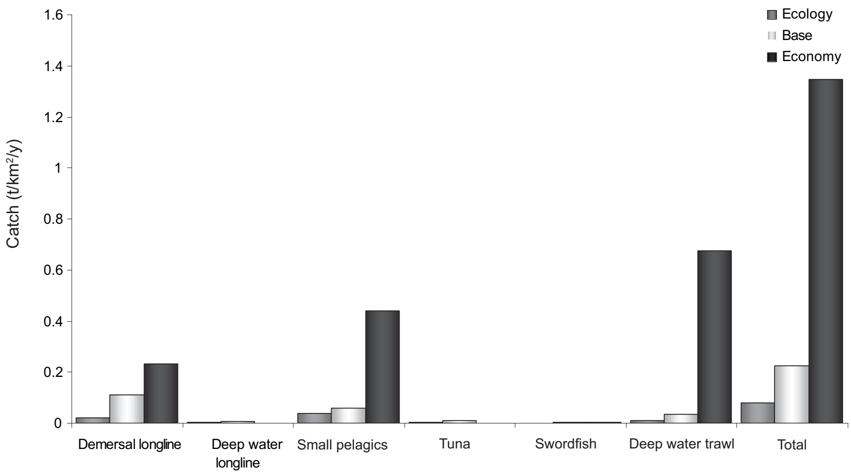

Figure 3. Catches (t km^{-2}y^{-1}) for the different fishing fleets under the base model and two fishing scenarios: maximizing the "ecology" objective function, and maximizing the "economy" objective function.

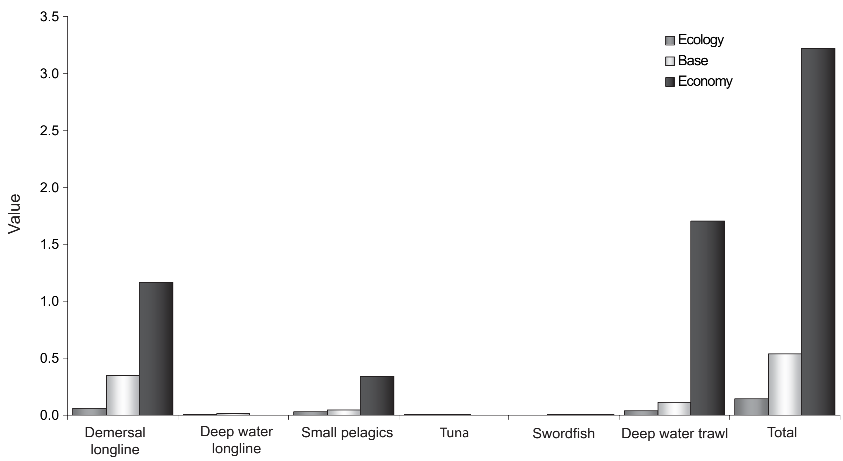

Figure 4. Value of the catches (relative value) for the different fishing fleets under the base model and two fishing scenarios: maximizing the "ecology" objective, and maximizing the "economy" objective.

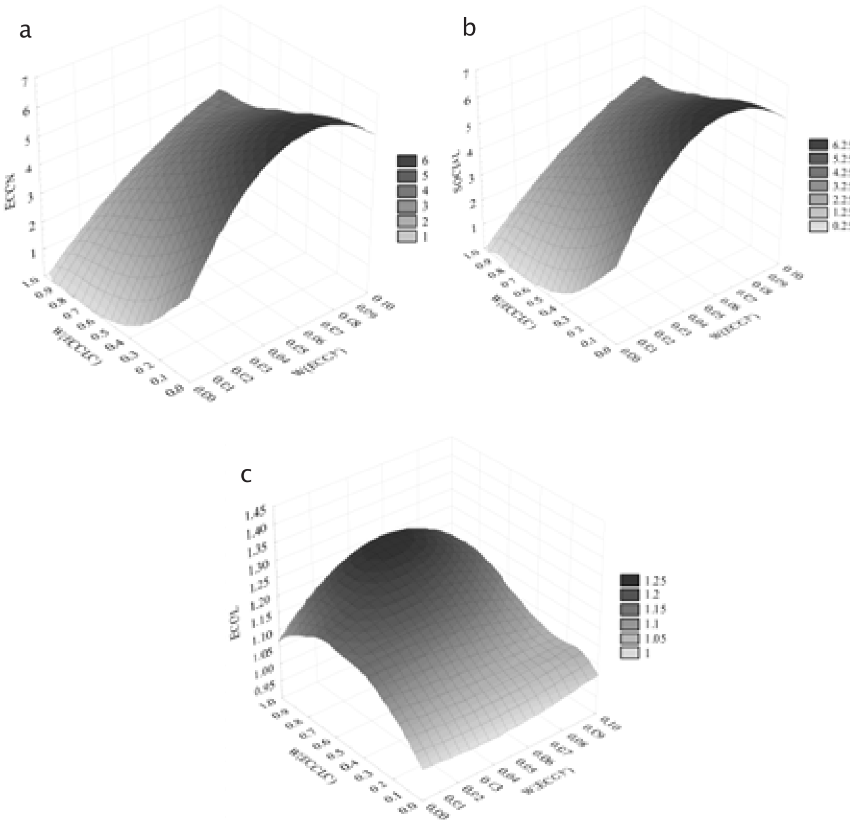

Figure 5. Surface plots showing optimal scenario solutions for a range of weightings of ecological and economic objective functions: (a) performance of "economy" objective, maximizing net economic value; (b) performance of "social" objective, maximizing number of jobs; (c) performance of "ecology" objective, maximizing ecological "stability" of the ecosystem.

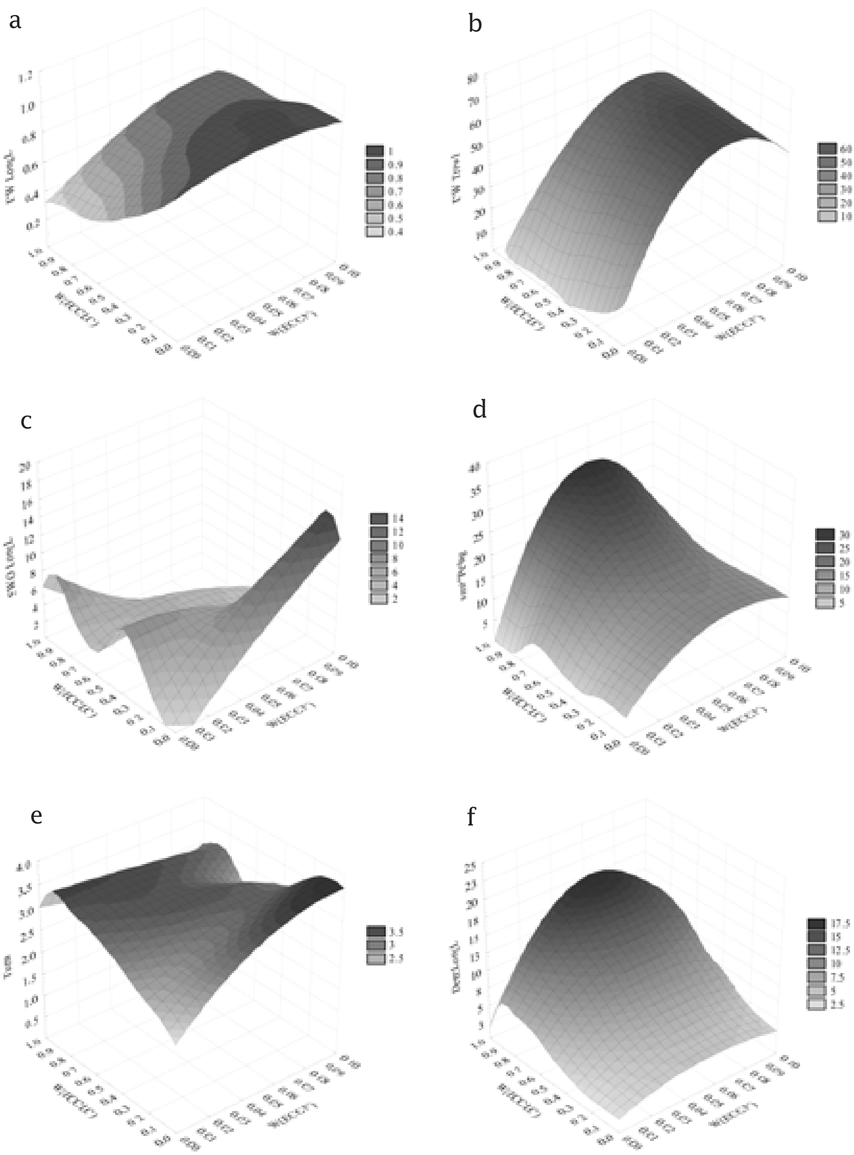

Figure 6. Surface plots showing the resulting fishing rates, as proportion of base model rates, of the optimal scenario solutions for a range of weightings of ecological and economic objective functions: (a) deepwater longline; (b) deepwater trawl; (c) pelagic longline; (d) small pelagic fishery; (e) tuna fishery; (f) demersal (bottom) longline.

"stability") simultaneously. This is true because net economic value (Fig. 5a) and number of jobs (Fig. 5b) reach a maximum with a high weighting factor on the economy objective and a small weight on ecology. This results in a decrease in the stability of the system. To the contrary, to maximize "ecosystem stability" a high weighting was assigned to the "ecology" objective (Fig. 5c). Assigning a low weighting factor to "economy" and a high weighting factor to "ecology" resulted in a decrease of net economic value and number of jobs, with a corresponding increase in the system's stability. Intermediate weightings produced, in general, intermediate performances for the three objective functions.

The fishing rates required to achieve different performances of the objective functions (i.e., fishing policies) are presented in Fig. 6. In order to maximize the net economic value of the system all fisheries, except deepwater longline (Fig. 6a), required an increase in their fishing rates. These increases were approximately 50 times the base model rate for the deepwater trawl fishery (Fig. 6b), 16 times for the swordfish fishery (Fig. 6c), 15 times for the small pelagic fish fishery (Fig. 6d), 3.5 times for the tuna fishery (Fig. 6e), and 2.5 times for the bottom longline fishery (Fig. 6f). To achieve ecological stability in the system, however, a decrease in the fishing rates of most of the fisheries was required with the exception of the tuna, swordfish, and to a lesser extent the bottom longline fisheries. The latter, along with the small pelagic fisheries, reached their highest fishing rates when a high weighting factor was assigned to the "ecology" objective and an intermediate weight to "economy."

Ecosystem indicators (i.e., mean trophic level of the catch and biodiversity) and total catches derived from the optimal fishing strategies for the overall range of weighting factors for "ecology" and "economy" are presented in Fig. 7. Total catches were maximized when weighting was high for "economy" and low for "ecology" objective functions (Fig. 7a). In contrast, the biodiversity index (Fig. 7b) was high only when a very small weight was assigned to "economy." The mean trophic level of the catch (Fig. 7c), in general, decreased with a corresponding increase in the weighting of "economy" and a decrease in the weighting of "ecology" objective functions. However, maximum trophic level was achieved with a high weighting of "ecology" and an intermediate weighting of "economy."

Discussion

The analyses in this paper are not meant to describe actual fisheries for seamounts, but rather as an exercise to explore the overall responses of seamount ecosystems to various multispecies management strategies.

The use of "open loop policy" search procedures can be unrealistic because it can entrust a fishery to fishing rates calculated at some time in the past and from data available from that time (Walters et al. 2002,

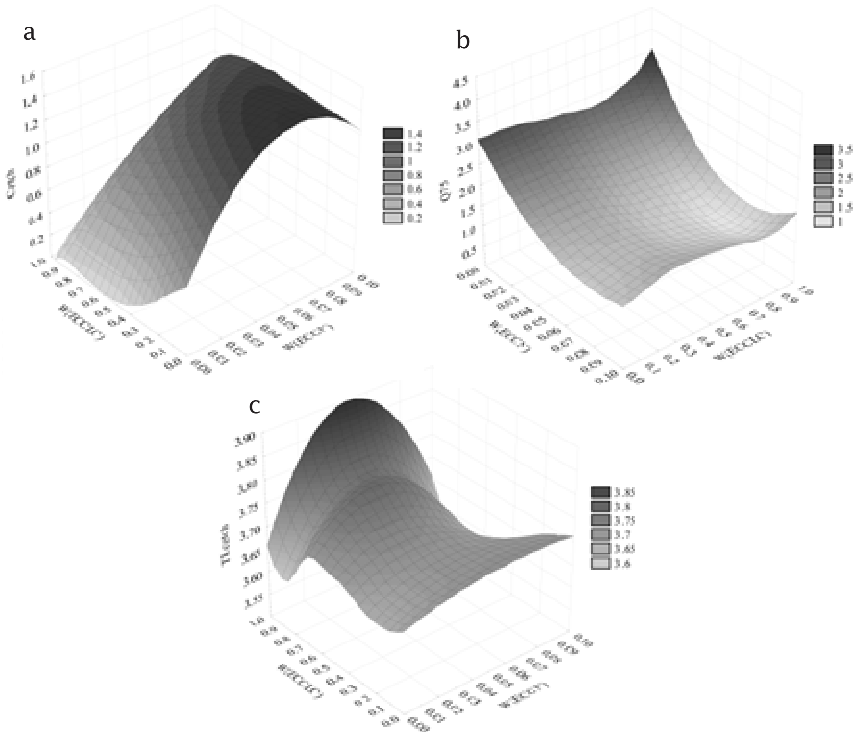

Figure 7. Surface plots showing the resulting ecosystem indicators and total fisheries catch for the optimal scenario solutions for a range of weightings of ecological and economic objective functions: (a) total catch; (b) biodiversity index (Q75) (note that this figure is shown from a different viewpoint); (c) mean trophic level of the catch.

Christensen and Walters 2004). Fisheries management needs to be implemented using "feedback policies" in which harvest goals are adjusted over time as new information becomes available and in response to unpredicted ecological changes (Christensen and Walters 2004). However, "open loop policy" calculations can give insights and directional guidance to where the system might be heading. In this study, this method appeared to be appropriate due to both the exploratory characteristic of the study and the data-limited situation of the North Atlantic seamounts.

Different extreme policy objectives for seamount fisheries may require different fleet configurations. Simulations that maximize economic performance favor deepwater trawling and require an increase in the fishing rates of all other fishing fleets, the only exception being the deepwater longline. On the other hand, maximization of ecological performance is achieved by favoring the operation of small pelagic and bottom longline fisheries. At the same time, a decrease in the fishing rates of all other fishing fleets is necessary.

Different fishing rates and fleet configurations produced different impacts on catches and consequently in the whole ecosystem. Optimizing for economics yielded six times the amount of landed catch and money than the base model scenario, and 17 and 23 times the amount of landed catches and money yielded by the scenario where ecology was maximized. This would, however, have implications to the whole ecosystem. While maximizing ecology produces an overall increase in biomass of most functional groups in the model, maximizing economics leads to a decrease and further collapse of some groups such as tuna, seamount-associated fishes, alfonsinos, as well as some charismatic species such as sea turtles and sharks. This point was well illustrated some time ago by Clark (1973) who pointed out that for populations that are economically valuable but possess low reproductive capacities (such as seamount-associated fishes, alfonsinos, and sharks), common property competitive exploitation and private property maximization of profits may lead to overexploitation and even to extinction of the population.

It is interesting to note that major collapses in deepwater fisheries, for example off New Zealand (Clark 1999, Clark et al. 2000), Tasmania (Koslow et al. 2000), and Namibia (Boyer et al. 2001), and habitat degradation (Probert et al. 1997, Koslow et al. 2001) were attributed to extensive deepwater trawling. In the Azores where no deepwater trawling is known to occur, seamount fisheries are mainly longline, handline, and pole-and-line, and are believed to be more sustainable. However, signs of stock decline are becoming apparent even in these systems (Santos et al. 1995, Menezes 2003). Thus, the question of whether deepwater (mainly trawl) fisheries are sustainable in the long term remains open (Clark 2001). Some authors (e.g., Probert 1999, Roberts 2002), agencies (World Wildlife Fund [WWF], International Union for Conservation of Nature and Natural Resources [IUCN]), and governments strongly advocate an urgent need for fishing regulations and/or the establishment of marine reserves in such areas.

It is apparent that major conflicts among stakeholders might emerge when different optimization scenarios result in completely different fishing policies (Figs. 5, 6). In addition, our results illustrate that maximizing "economy" affects biodiversity in the ecosystem and probably the trophic level of the catch, while maximizing the total landed catches (see Fig. 7). The opposite is true when "ecology" is favored; the total catch and the number of jobs are decreased in order to achieve high biomass of long-lived species and increased biodiversity in the ecosystem.

In conclusion, sustainable seamount fisheries with minimal ecosystem effects appear to be achieved when the "ecology" objective is maximized. However, more information for these fragile ecosystems and the long-term impacts of fishing and other human activities needs to be acquired. Meanwhile, the precautionary principle ought to be applied to seamount ecosystems, in order to ensure protection and sustainable management.

Acknowledgments

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Appendix 1
Modeling approach

The model parameters (Table A1), production to biomass ratios (P/B), and consumption to biomass ratios (Q/B), are calculated on a yearly basis. Biomass and catch are expressed in metric tons of wet weight per square kilometer. The Q/B ratios for fish groups were estimated using an empirical equation (Palomares and Pauly 1998). Temperature values were established as being 18ºC for the epipelagic region (0-200 m), 8ºC for the mesopelagic region (200-1,000 m), and 6ºC for the bathypelagic region (1,000-4,000 m). For some groups Q/B values were taken from other models. For most groups P/B ratios were extracted from previously constructed models, or were estimated assuming production and consumption ratio equal to 0.3 (Christensen 1996). Following Shannon and Jarre-Teichmann (1999), the proportion of food consumed and not assimilated was taken as 0.2. When no biomass estimate was available, this parameter was left to estimate by Ecopath using a value of 0.95 for ecological efficiency (EE). A preliminary diet matrix was assembled using published data, unpublished local information, or empirical knowledge. When unidentified categories were found in the literature, data were re-expressed out of 100% to exclude these groups.

The theoretical seamount was assumed to have a low initial level of exploitation. Landings (Table A2) varied from 218 t per year for shallow benthic fishes to 2.7 t per year for billfishes.

Table A1. Input parameters and estimates (in parentheses) from the theoretical non-migratory model of a seamount. P/B is production to biomass ratio, Q/B is consumption to biomass ratio, EE is ecotrophic efficiency, and TL is trophic level of the groups.

Group name	Biomass (t km^{-2})	P/B (year^{-1})	Q/B (year^{-1})	EE	TL	Landings (t km^{-2})
Toothed whales	0.000	0.020	10.270	(0.513)	5.17	
Baleen whales	0.123	0.060	5.563	(0.024)	3.56	
Dolphins	0.040	0.070	11.410	(0.050)	4.58	
Sea turtles	0.001	0.150	3.500	(0.899)	4.08	
Seabirds	0.000	0.040	84.390	(0.000)	4.36	
Tunas	0.032	0.742	16.291	(0.686)	4.58	0.011
Billfishes	0.020	0.500	4.200	(0.101)	4.54	0.001
Sharks pelagic	0.011	0.300	3.100	(0.868)	4.70	0.002
Sharks benthopelagic	0.030	0.510	6.900	(0.160)	4.40	0.002
Rays and skates	0.020	0.170	1.500	(0.678)	3.91	0.002
Large oceanic planktivores	(0.003)	0.112	2.066	0.100	3.56	
Epipelagic S	0.859	2.053	19.867	(0.567)	3.10	0.050
Epipelagic M	0.113	1.080	10.750	(0.902)	3.59	0.010
Epipelagic L	0.014	0.690	5.095	(0.487)	4.18	
Mesopelagic migrating S	2.000	1.980	8.000	(0.981)	3.37	
Mesopelagic migrating L	(0.003)	0.600	3.550	0.950	4.34	
Mesopelagic non-migrating	(1.421)	0.500	1.570	0.950	3.12	
Shallow benthic fishes	(0.820)	0.590	4.700	0.950	3.70	0.080
Shallow benthic fishes	(0.820)	0.590	4.700	0.950	3.70	
Shallow demersal fishes	(0.193)	0.660	5.200	0.950	4.04	0.020
Seamount-associated fishes	(0.890)	0.060	2.200	0.950	4.14	0.011
Hoplostethus atlanticus	(0.452)	0.048	2.000	0.900	4.39	0.010
Beryx spp.	(0.343)	0.060	2.000	0.950	3.90	0.010
Bathypelagic	(0.029)	0.500	1.477	0.950	4.12	0.006
Bathybenthic fishes	(1.143)	0.200	0.500	0.950	3.54	0.003
Bathydemersal fishes	(1.283)	0.200	0.600	0.950	4.19	0.002
Benthos filter feeders	(0.595)	0.800	9.000	0.950	2.00	
Benthos scavengers	(3.089)	1.830	13.567	0.950	2.35	
Crustacea benthic	(3.858)	1.600	10.000	0.950	2.00	
Crustacea pelagic	(5.161)	1.450	9.667	0.950	2.72	
Cephalopods resident	(0.189)	2.890	10.000	0.950	3.78	
Cephalopods drifting S	(0.175)	4.450	16.863	0.950	3.83	
Cephalopods drifting L	(0.001	2.500	10.000	(0.726)	4.33	
Gelatinous zooplankton	(8.895)	0.850	2.000	0.800	3.08	
Zooplankton shallow	16.684	(11.214)	37.379	(0.774)	2.11	
Zooplankton deep	6.849	(8.700)	29.000	(0.595)	2.23	
Phytoplankton	7.160	290.000	–	(0.358)	1.00	
Detritus	–	–	–	(0.045)	1.00	

S = small; M = medium; L = large.

Table A2. Average landings estimated for the different fisheries considered in the theoretical seamount.

Group name	Landings by fleet (t km^{-2}y^{-1})						Total
	DL	DWL	SP	T	SW	DWT	(t y^{-1})
Tunas				0.011			30.0
Billfishes					0.001		2.7
Sharks pelagic	0.001				0.001		5.5
Sharks benthopelagic	0.001					0.001	5.5
Rays and skates	0.002						5.5
Epipelagic S			0.05				136.4
Epipelagic M			0.01				27.3
Shallow benthic fishes	0.08						218.2
Shallow demersal fishes	0.02						54.6
Seamount-associated fishes		0.001				0.01	30.0
*Hoplostethus atlanticu*s						0.01	27.3
Beryx spp.	0.005					0.005	27.3
Bathypelagic		0.005				0.001	16.4
Bathybenthic fishes		0.002				0.001	8.2
Bathydemersal fishes	0.001	0				0.001	5.5
Total t km^{-2}y^{-1}	0.11	0.008	0.06	0.011	0.002	0.029	

DL = demersal longline; DWL = deepwater longline; SP = small pelagics fishery; T = tuna; SW = swordfish; DWT = deepwater trawl; S = small; M = medium.