26th Lowell Wakefield Fisheries Symposium

Global Progress in Ecosystem-Based Fisheries Management

EDITORS

GORDON H. KRUSE • HOWARD I. BROWMAN • KEVERN L. COCHRANE
DIANA EVANS • GLEN S. JAMIESON • PATRICIA A. LIVINGSTON
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About the Proceedings Book and the Symposium

Eighteen peer-reviewed research papers are included in this proceedings volume; all were presented at the symposium Ecosystems 2010: Global Progress on Ecosystem-Based Fisheries Management, November 8-11, 2010, in Anchorage, Alaska. A total of 61 oral presentations and 10 posters were presented at the symposium.

The goals of Ecosystems 2010 were to: (1) evaluate global progress toward EBFM by reviewing regional case studies, development of new analytical tools, and practical approaches toward future progress; and (2) offer explicit, practical advice for future progress in implementation of EBFM. To meet these goals, oral presentations and posters were organized along four main themes: (1) progress on regional applications; (2) new analytical tools and evaluation of ecosystem indicators; (3) human dimensions; and (4) case studies and practical solutions.

The symposium attracted broad international interest and was attended by 108 registered participants from 19 countries: Argentina, Australia, Brazil, Canada, Estonia, India, Italy, Japan, Korea, Malaysia, New Zealand, Norway, Pakistan, South Africa, Sweden, Taiwan, Thailand, United Kingdom, and USA. The size and diversity fostered a very collegial atmosphere to discuss and contrast approaches in many regions of the world. The keynote and seven invited speakers further emphasized the international focus of this symposium with presentations on contrasting marine ecosystems of Thailand, Korea, Japan, Australia, Namibia, Norway, and Atlantic Canada.

The symposium was concluded with a panel discussion, which appears as the last paper in this proceedings. Symposium participants agreed that this was a very successful collaboration among ICES, PICES, FAO, and regional partner organizations. Participants urged a follow-up symposium on this same topic in the not-too-distant future.

Details about the symposium Ecosystems 2010: Global Progress on Ecosystem-based Fisheries Management, including the program, presentation slides, and book of abstracts are available on the symposium website at http://seagrant.uaf.edu/conferences/2010/wakefield-ecosystems/index.php.

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The Lowell Wakefield Symposium Series and Endowment

Alaska Sea Grant has been sponsoring and coordinating the Lowell Wakefield Fisheries Symposium series since 1982. These meetings are a forum for information exchange in biology, management, and economics of various fish species and complexes, as well as an opportunity for scientists from high-latitude countries to meet informally and discuss their work.

Lowell Wakefield was the founder of the Alaska king crab industry. He recognized two major ingredients necessary for the king crab fishery to survive—ensuring that a quality product be made available to the consumer, and that a viable fishery can be maintained only through sound management practices based on the best scientific data available. Lowell Wakefield and Wakefield Seafoods played an important role in the development and implementation of quality control legislation, in the preparation of fishing regulations for Alaska waters, and in drafting international agreements for the high seas. In his later years, as an adjunct professor of fisheries at the University of Alaska, Lowell Wakefield influenced the early directions of Alaska Sea Grant. The Wakefield Symposium series is named in honor of Lowell Wakefield and his many contributions to Alaska's fisheries. In 2000, Frankie Wakefield (Lowell's wife) made a gift to the University of Alaska Foundation to establish an endowment to continue this series.

Premcharoen, S. 2012. Ecopath Model of the Mae Klong Estuary, Inner Gulf of Thailand.
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Ecopath Model of the Mae Klong Estuary, Inner Gulf of Thailand

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Abstract

The Mae Klong Estuary ecosystem is healthy based on output parameters from a mass-balanced model (Ecopath). The model constructed in this study included 21 compartments (state variables), representing 63 exploited fish and commercial invertebrate species as well as the energy (feeding) fluxes among them. The parameterization of the model reflects the ecological and multispecies interactions. Several higher order indices related to the ecosystem maturity indicators were computed and compared with other coastal ecosystems around the world. The results indicate that Mae Klong has a mixture of characteristics of both a mature system (high total system throughput, ascendency, and overhead) and immature system (high total primary production/total respiration ratio, low Finn's cycling index, and mean path length), in comparison with other estuarine systems. The results emphasize the need for management and conservation between the two sectors of fisheries and forestry, whose present trajectories tend toward further degradation of the Mae Klong ecosystem. The modeling approach presented in this study is innovative in Thailand and, indeed, for tropical mangrove ecosystems. It has also allowed Thai mangrove ecosystems to be placed within a broader, global context.

Introduction

An understanding of ecosystem structure and function, particularly relating to trophic composition, trophic efficiencies, and biomass transfers among species, is a priority for the sustainable management of aquatic resources (Ulanowicz 1986, Christensen and Pauly 1995, McCann 2000, Kitchell et al. 2004). Ecosystem complexity can be simulated through energy-based models (Jørgensen 1994, FAO 2007) that identify levels of production and evaluate functional responses to natural and/or anthropogenic impacts (Christensen and Pauly 1992).

Mangrove forests along the inner Gulf of Thailand provide productive nursery areas for offshore fishes (Menasveta 1976) but in recent decades have experienced drastic declines through a reduction in habitat quality (Sudara et al. 1994) and shrimp culture (Huitric et al. 2002, Barbier 2003). Fishery information from the Gulf of Thailand mangrove ecosystem has not been collected regularly, especially on food webs and trophic organization (Chong et al. 1990, Poovachiranon and Satapoomin 1994, Sasekumar et al. 1994, Hajisamae et al. 1999). A notable exception to this has been information from the Gulf of Thailand provided by Vibunpant et al. (2003), which, together with data from the present study collected from the Mae Klong Estuary in 2005-2006, were used to construct the model presented here.

The objective of this study was to construct a mass-balanced model of the Mae Klong Estuary ecosystem with special emphasis on the fish community, in particular the trophic interactions, and to calculate ecosystem maturity indicators.

Materials and methods

Study area

The Mae Klong River lies in the western part of the inner Gulf of Thailand (13.33º-14.00ºN, 99.50º-100.09ºE) (Fig. 1). The river, 138 km in length, starts from the confluence of the Khwai Yai and Khwai Noi rivers in Kanchanaburi Province and flows through Ratchaburi and Samut Songkhram provinces into the Gulf of Thailand.

The Mae Klong basin can be divided into two subbasins. The lower subbasin, under the influence of seawater intrusion, extends from the Mae Klong River mouth in Samut Songkhram Province to Ratchaburi Province. This subbasin is about 45 km in length, with a highly populated area near the coast. Patches of mangrove and broad mudflats occupy the coastline of Samut Songkhram Province, supporting mussel and clam cultivation. The main activities in the coastal area include aquaculture, salt ponds, and fisheries, particularly razor clam harvesting. Fish, shellfish, and jellyfish are important local fishery products in this area. The mouth of the Mae Klong River, with a vast area of intertidal mudflat, is an extremely productive location for the economically important razor clam Hoi Lot (*Solen regularis*). Mangroves are present along the shoreline on the eastern side of the river mouth. The encroachment of aquaculture and tourist infrastructure into the mangroves along with industrial, agricultural, and urban effluents into the Mae Klong River represent environmental concerns.

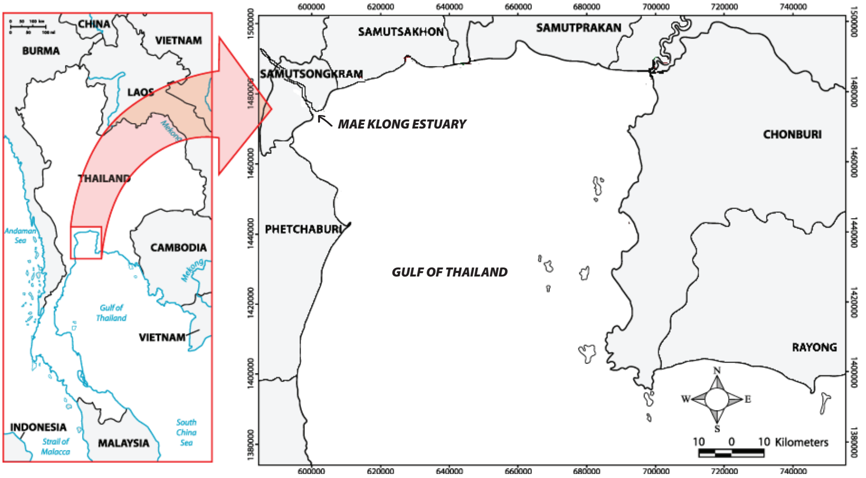

Figure 1. Location of Mae Klong Estuary, inner Gulf of Thailand (modified from Veeravaitaya 2007).

Ecopath modeling framework

The data requirements of an Ecopath model are expressed by its master equation. The basic condition is that input to each group is equal to the output from it (equilibrium conditions). Then, a series of biomass budget equations are determined for each group as:

Production = fisheries catch + predation mortality +
biomass accumulation + net migration + other mortality

In addition the groups in the system are linked through predators consuming prey. Such consumption can be described by

Consumption = production + non-assimilated food + respiration

The terms of this equation may be replaced by:

production by $i = B_i \times P/B_i$,

predator losses of $i = \sum_j (B_j \times Q/B_j \times DC_{ji})$, and

other losses of $i = (1 - EE_i) \times B_i \times P/B_i$

The equation developed by Polovina (1984) can be presented as follows:

$$P_i = Y_i + B_i M2_i + BA_i + EX_i + P_i(1 - EE_i) \qquad \text{eq. 1}$$

where:

i is the component (stock, species, group of species) of the model,

j is any predator of i,

B_i is the biomass for species or group (i),

P_i is the total production rate of i,

Y_i is the total fishery catch rate of i,

$M2_i$ is the total predation rate for i,

BA_i is the biomass accumulation,

EX_i is the export out of the system (migration or fisheries catches) for species or group i,

EE_i is the ecotrophic efficiency, i.e., the proportion of the ecological production that is consumed by predators and usually assumed to range from 0.7 to 0.99 (Polovina 1984), and

$P_i(1 - EE_i)$ is the other losses not elsewhere included.

To incorporate most of the production components in the form of predation or mortality, equation 1 can be re-expressed as:

$$B_i(P/B)_i - \sum_j(B_j \times Q/B_j \times DC_{ji}) - (P/B)_i(1 - EE_i) - EX_i - BA_i - Y_i = 0 \qquad \text{eq. 2}$$

where:

B_j is the biomass of predator (j),

P/B_i is the production/biomass ratio, usually assumed equal to the total mortality (Z_i),

Q/B_j is the consumption per unit of biomass for predator j,

DC_{ji} is the fraction of prey (i) in the average diet of predator (j),

Therefore, a system with n groups (boxes) will have n linear equations. Since Ecopath links the different groups, it allows the estimation of one unknown parameter for each group. Required inputs for creating an Ecopath model are three of four following parameters: B_i, $(P/B)_i$, $(Q/B)_j$, and EE_i, although it is recommended that B_i, $(P/B)_i$, and $(Q/B)_j$ are specified (Christensen and Walters 2004). Once three parameters are entered for each group, a diet composition matrix is constructed. The diet matrix is constructed by calculating the percent of each prey that occurs in each predator's diet. The Ecopath model then is checked for steady-state conditions. The element of the diet matrix or the values of the three input parameters are adjusted until the EE_i for each group is between zero and one. The data required for Ecopath are assembled and standardized to t per km^2 and t per km^2 per year.

Biomass estimation

The biomass of a fish species (or group of fish species) was assumed to be constant for the period covered by the model. This parameter is expressed in t wet weight per km^2.

CPUE (catch per unit effort) values were used to estimate biomass. The biomass of fish from the study was estimated using the swept area method (Sparre and Venema 1992) as follows:

$$B = \frac{CPUE}{aX_1} A$$

where:

A (total area) = 15.9 km^2

a (swept area) = 0.11112 km^2

X_1 (proportion of fishes in the path of the trawl retained by it) = 0.5

The swept area was estimated from the equation:

$$a = t \times v \times h \times X_2$$

where:

t (time spent trawling) = 6 hrs

v (trawling speed) = 1850 m per hr

h (length of trawl head rope) = 20 m

X_2 (effective width of the trawl relative to its head rope) = 0.5

Sources of the basic input parameters and diet

The data for each functional group were obtained from field data whenever possible. The value for the "detritus" group is in reality unknown but is set at a very large number in line with usual practice for Ecopath models. However, for a few species these data were not available and were obtained mostly from literature reports for species groups in a similar area (Table 1). Those data therefore have less of a pedigree than data actually recorded from the site. Imports were not included in the diet matrix due to the lack of net migration rate for most of the species.

Table 1. EwE model inputs and sources for groups in the Mae Klong Estuary.

Group	Input	Source	Group	Input	Source
Rays	B	From this study	Pelagics	B	From this study
	P/B	Vibunpant et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	N/A		Q/B	N/A
	Diet	From this study		Diet	From this study
Anchovies	B	From this study	Benthopelagics	B	From this study
	P/B	Garces et al. (2003)		P/B	Garces et al. (2003)
	Q/B	Garces et al. (2003)		Q/B	Garces et al. (2003)
	Diet	From this study		Diet	From this study
Sardines	B	From this study	Benthics	B	From this study
	P/B	Garces et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	Garces et al. (2003)		Q/B	N/A
	Diet	From this study		Diet	From this study
Catfishes	B	From this study	Nekton	B	N/A
	P/B	Vibunpant et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	N/A		Q/B	N/A
	Diet	From this study		Diet	Vibunpant et al. (2003)
Mullets	B	From this study	Sergestid shrimps	B	N/A
	P/B	Garces et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	Garces et al. (2003)		Q/B	N/A
	Diet	From this study		Diet	Vibunpant et al. (2003)
Perchlets	B	From this study	Shrimps	B	N/A
	P/B	Garces et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	Garces et al. (2003)		Q/B	N/A
	Diet	From this study		Diet	Vibunpant et al. (2003)
Ponyfishes	B	From this study	Crabs	B	N/A
	P/B	Vibunpant et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	N/A		Q/B	N/A
	Diet	From this study		Diet	Vibunpant et al. (2003)
Threadfin	B	From this study	Benthic invertebrates	B	N/A
	P/B	Garces et al. (2003)		P/B	Vibunpant et al. (2003)
	Q/B	Garces et al. (2003)		Q/B	N/A
	Diet	From this study		Diet	Vibunpant et al. (2003)
Croakers	B	From this study	Zooplankton	B	N/A
	P/B	Vibunpant et al. (2003)		P/B	Vibunpant et al. (2003)
				Q/B	Vibunpant et al. (2003)
				Diet	Vibunpant et al. (2003)
	Q/B	N/A	Phytoplankton	B	Vibunpant et al. (2003)
	Diet	From this study		P/B	Vibunpant et al. (2003)
Spotted scat	B	From this study	Detritus	B	Vibunpant et al. (2003)
	P/B	Garces et al. (2003)			
	Q/B	Garces et al. (2003)			
	Diet	From this study			

Defining functional groups (compartments)

The state variables selected for the food web in the present study were based on the following criteria: (1) ecological or taxonomic related species; (2) typical and abundant species; (3) species of economic and social importance; and (4) species for which there are historical data and information.

On the basis of the above criteria, 21 functional groups were selected (Table 2). Most groups represent the most important trophic links of this system. Only those of particular interest remained as an individual group, such as the commercially important shrimp, crab, sardine, anchovy, catfish, and threadfin. Nekton and sergestid shrimp are separated from zooplankton as a discrete group. Additionally, some fish groups were divided into pelagics (4 species), benthopelagics (9 species), and benthics (14 species).

Strategy for model balancing

The first Ecopath eq. 1 states that each group must be mass-balanced, i.e., catches, consumption, biomass accumulation, and export do not exceed production for a group. Therefore, balancing the model requires adjustment of the input parameters so that ecotrophic efficiencies (*EE*) do not exceed 1. If $EE > 1$, this indicates that predation on the compartment is greater than production by the compartment. If $EE < 1$ for a group, this indicates an excess of biomass at the end of the considered period (12 months in this study), which may accumulate in the system, migrate out the system, or is lost due to other mortality. I assumed no accumulation of biomass of any groups ($BA = 0$) and for fishery harvest it has been set at zero. This is because the harvest rate of sizes and categories of the fish sampled in the present study are not known (unreported). Although fluxes of water coming into the estuary are unknown, the water circulation is expected to export living or detrital matter out of the estuary. Therefore, a group with a low *EE* was expected to lose biomass through export via water fluxes passing through the estuary.

I also assume no significant interannual differences. This is a common and simplifying assumption done in order to allow the modeling of complex systems (Christensen and Walters 2004). I applied the following strategy to achieve mass balance for all groups. First, adjustments of diets were given priority since feeding habits of some groups are highly variable and mainly dependent on which food sources are available in the ecosystem. Second, I gave preference to the adjustments of parameters that were not estimated in the field.

Table 2. Composition of ecological groups used for EwE modeling of the Mae Klong Estuary.

Ecological groups	Taxa
Rays	Dasyatidae: *Dasyatis fluviorum, Himantura imbricata*
Anchovies	Engraulidae: *Setipinna taty, Stolephorus commersonii, Thryssa hamiltoni, T. setrirostris*
Sardines	Clupeidae: *Anodontostoma chacunda, Escualosa elongata,* ***E. thoracata, Herk-*** *lotsichthys dispilonotus, Hilsa kelee, Sardinella albella, S. fimbriata, S. gibbosa, S. lemuru*
Catfishes	Ariidae: *Arius macronotacanthus, A. sagor, Cryptarius truncatus, Ketengus typus, Osteogeneiosus militaris* Plotosidae: *Plotosus canius*
Mullets	Mugillidae: *Chelon tade, Liza subviridis, Moolgarda seheli*
Perchlets	Ambassidae: *Ambassis gymnocephalus, A. nalua*
Ponyfishes	Leiognathidae: *Leiognathus decorus, Secutor insidator, S. ruconius*
Threadfin	Polynemidae: *Eleuthronema tetradactylum*
Croakers	Sciaenidae: *Aspericorvina jubata, Chrysochir aureus, Dendrophysa russelli, Nibea albiflora, Panna microdon*
Spotted scat	Scatophagidae: *Scatophagus argus*
Pelagics	Atherinidae: *Hypoatherina valenciennei* Belonidae: *Strongylura strongylura* Hemirhamphidae: *Hyporhamphus (Hyporhamphus) limbatus, Rhynchorhamphus naga*
Benthopelagics	Elopidae: *Elops machnata* Sillaginidae: *Sillago sihama* Carangidae: *Alepes djedaba, Scomberoides commersonnianus, S. tol, Selaroides leptolepis* Gerreidae: *Gerres erythrourus* Drepanidae: *Drepane punctata* Teraponidae: *Terapon theraps*
Benthics	Ophichthidae: *Pisodonophis boro* Eleotridae: *Butis butis, Ophiocara porocephala* Gobiidae: *Acentrogobius caninus, A. chlorostigmatoides, Aulopareia cyanomos, Bathygobius fuscus, Favonigobius aliciae, Pseudapocryptes lanceolatus* Cynoglossidae: *Cynoglossus lingua, C. puncticeps* Triacanthidae: *Triacanthus biaculeatus* Tetraodontidae: *Lagocephalus lunaris* Uranoscopidae: *Uranoscopus bicinctus*
Nekton	Juvenile fishes
Sergestid shrimps	Sergestidae: *Acetes* spp.
Shrimps	Includes all juvenile and adult shrimp of *Alpheus* spp., *Penaeus* spp., and *Metapenaeus* spp.
Crabs	Portunidae, Scyllaridae, Ocypodidae
Benthic invertebrates	Polychaetes, bivalves, gastropods, barnacles, sipunculids, eunicids, bryozoans, nematodes, trematodes, nemerteans
Zooplankton	Zoea of crab, megalopa, mysids, amphipods, copepods, isopods, ostracods, cladocerans (*Daphnia*), cumaceans, euphausids, tintinnids, *Lucifer* larvae, bivalve larvae, Cirripedia larvae, stomatopod larvae, planktonic foraminiferans, fish eggs
Phytoplankton and benthic producers	Dominated by diatoms *Actinocyclus, Amphipleura, Amphora, Anabaena, Anomocnema, Asterionella, Asteronellopsis, Bacillaria, Ceratium, Closterium, Cocconeis, Coscinodiscus, Cyclotella, Cymbella, Diploneis, Ditylum, Epithemia, Eucampid, Eutonia, Fragilaria, Grammetophora, Gyrosigma, Lauderia, Mastogloia, Navicula, Nitzschia, Odontella, Oscillatoria, Phagus, Pinnularia, Pleurosigma, Pseudonitzschia, Rhizosolenia, Rhopaladia, Scenedesmus, Skeletonema, Surirella, Thalassiosira, Thalassiothrix, Thalassionema, Urotrix*, and dinoflagellates (*Dinophysis*) Marine algae
Detritus	Particulate and dissolved organic matter

Results

Trophic level and flow

The balanced parameter estimates of the Mae Klong Estuary food web are shown in Table 3, and the diet matrices are displayed in Table 4. Detritus and phytoplankton displayed the highest values for biomass and production. This has to be noted, since they are the main food supply for fish groups and show a strong relation with primary producers, including detritus.

Throughout the study period, most of the fish groups were characterized by small sizes and feeding at low trophic levels (*TL*). Functional groups were organized within three integer trophic levels (*TL*). The groups with *TL*s between 3.34 and 3.0 were rays, anchovies, catfishes, ponyfishes, threadfin, croakers, spotted catfish, pelagic fishes, and benthopelagic fishes. Invertebrates were classified between 2.0 and 2.6 and the lowest, by definition, were the primary producers and detritus groups (*TL* = 1). The average trophic level of each group revealed that pelagic fishes occupied the highest trophic level (3.24). There were no changes in the *TL*s of nekton, shrimps, sergestid shrimps, crabs, benthic invertebrates, zooplankton, phytoplankton, and detritus.

Table 5 shows the distribution of relative flows by trophic level. Relative flow was greatest at trophic level III. Most of the flows in trophic level II (detritivores and herbivores) are due to zooplankton (the dominant herbivores in this ecosystem) and shrimps and sergestid shrimps (the dominant detritivores). Flows in trophic level III are attributed to crabs and benthic invertebrates and an array of fish groups. At level IV, flows are dominated by pelagic fishes and at level *V* by top predators such as rays. Since the magnitude of flows at trophic levels greater than the fifth is very low, representing only a small fraction of the flows associated with the top predators, these levels were omitted from further consideration.

Mangrove plays an important role in detritus accumulation due to the large amount of leaf material that is incorporated within the soil. None of the species within the models feed directly on mangrove biomass. This detritus is utilized by several groups in the food web. Phytoplankton also contributes to the productivity of higher trophic levels that are dependent on detritus.

Summary statistics

Some whole system properties (Table 6) can be used to assess the status of the ecosystem in terms of maturity (sensu Odum 1969). The total system throughput estimated for the Mae Klong Estuary was 8,321 t per km^2 per year.

Table 3. Input and parameter estimates by Ecopath (in brackets) for the Mae Klong Estuary.

Group name	Trophic level	Biomass in habitat area (t/km^2)	Production/biomass (per year)	Consumption/biomass (per year)	Ecotrophic efficiency	Production/consumption
Rays	3.17	0.221	0.500	(2.500)	(0.357)	0.200
Anchovies	3.00	0.143	2.700	7.900	(0.902)	(0.342)
Sardines	2.93	0.036	2.700	7.900	(0.900)	(0.342)
Catfishes	3.09	1.357	2.000	10.000	(0.689)	0.200
Mullets	2.21	1.060	0.430	10.750	(0.793)	(0.040)
Perchets	2.73	0.041	2.150	10.750	(0.893)	(0.200)
Ponyfishes	3.06	0.070	3.500	(14.000)	(0.944)	0.250
Threadfin	2.98	0.239	1.740	8.700	(0.841)	(0.200)
Croakers	3.12	0.552	1.500	(7.500)	(0.950)	0.200
Spotted scat	3.00	0.035	2.150	10.750	(0.794)	(0.200)
Pelagics	3.24	0.769	3.000	(12.000)	(0.823)	0.250
Benthopelagics	3.14	0.076	2.150	10.750	(0.831)	(0.200)
Benthics	2.44	0.245	3.000	(12.000)	(0.925)	0.250
Nekton	2.50	(0.312)	4.000	(16.000)	0.950	0.250
Shrimps	2.00	(0.520)	10.000	(16.000)	0.950	0.250
Sergestid shrimps	2.00	(1.956)	5.000	(20.00)	0.950	0.250
Crabs	2.61	(1.129)	3.000	(12.000)	0.950	0.250
Benthic invertebrates	2.22	(14.751)	5.000	(25.000)	0.650	0.200
Zooplankton	2.00	(5.543)	40.000	280.000	0.200	(0.143)
Phytoplankton	1.00	(18.286)	200.000		0.440	
Detritus	1.00	10000.000			(0.111)	

Table 4. Diet matrix of the Mae Klong Estuary

Prey/ predator	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19
Rays																			
Anchovies																			
Sardines																			
Catfishes																			
Mullets																			
Perchets																			
Ponyfishes																			
Threadfin																			
Croakers																			
Spotted scat																			
Pelagics																			
Benthopelagics																			
Benthics											0.103								
Nekton		0.001		0.038					0.017		0.396	0.160	0.056						
Shrimps	0.05	0.023	0.289	0.483		0.250	0.060	0.160	0.295	1.000	0.082	0.029	0.141						
Sergestid shrimps	0.435	0.876			0.007	0.122		0.776	0.303		0.259	0.212	0.113						
Crabs	0.285			0.169					0.164		0.002		0.029						
Benthic invertebrates		0.017	0.125	0.129	0.141	0.004	0.273	0.007	0.084		0.020	0.254	0.009				0.500	0.100	
Zooplankton	0.230	0.082	0.486	0.122	0.030	0.356	0.667	0.032	0.130		0.077	0.346	0.046	0.500				0.100	
Phytoplankton			0.050		0.539				0.007					0.500				0.200	1.000
Detritus		0.001	0.050	0.059	0.283	0.268		0.025			0.061		0.606		1.000	1.000	0.500	0.600	
Import																			
Sum	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000

Table 5. Relative flows by trophic levels of the Mae Klong Estuary.

Group/trophic level	I	II	III	IV	V
Rays	0.000	0.000	0.858	0.127	0.016
Anchovies	0.000	0.001	0.997	0.002	0.000
Sardines	0.000	1.000	0.886	0.014	0.000
Catfishes	0.000	0.059	0.823	0.108	0.000
Mullets	0.000	0.822	0.162	0.160	0.000
Perchets	0.000	0.268	0.732	0.000	0.000
Ponyfishes	0.000	0.000	0.970	0.030	0.000
Threadfin	0.000	0.025	0.974	0.001	0.000
Croakers	0.000	0.007	0.893	0.091	0.009
Spotted scat	0.000	0.000	1.000	0.000	0.000
Pelagics	0.000	0.061	0.697	0.237	0.004
Benthopelagics	0.000	0.000	0.892	0.108	0.000
Benthics	0.000	0.060	0.351	0.042	0.002
Nekton	0.000	0.500	0.500	0.000	0.000
Shrimps	0.000	1.000	0.000	0.000	0.000
Sergestid shrimps	0.000	1.000	0.000	0.000	0.000
Crabs	0.000	0.500	0.444	0.056	0.000
Benthic invertebrates	0.000	0.889	0.111	0.000	0.000
Zooplankton	0.000	1.000	0.000	0.000	0.000
Phytoplankton	1.000	0.000	0.000	0.000	0.000
Detritus	1.000	0.000	0.000	0.000	0.000

The total primary production/total respiration (*PP/R*) ratio of 2.829 implied that Mae Klong is in a developing stage with this ratio being greater than 1. Ascendency (*A*) measures the structure of an ecosystem in terms of the amount and organization of biomass flow within the system. Based upon Odum's (1969) interpretation of the attributes of ecosystems, more speciation, finer specialization, longer retention, and more cycling within the system indicates that an ecosystem is more mature. Higher ascendancy values indicate relatively high values in one or more of these properties. The upper limit to ascendancy is the development capacity (*C*) of the ecosystem. System overhead is the difference between capacity and ascendancy. System overhead is the upper limit to how much ascendancy can increase to counteract unexpected perturbations. Higher overhead indicates that a system has a larger amount of energy reserves with which it can react to perturbations, so that the system should be more able to maintain stability when perturbed. An ascendency value of 44.3% was obtained from the Mae Klong Estuary, which is typical of values for coastal and estuarine ecosystems (see Table 7). The ecosystem has a large overhead, suggesting that all should be resilient, reflected in the high values for resilience.

Recycling of energy and matter are considered an important process in ecosystem functioning, as it is related to maturity and stability (Odum 1969, Christensen and Pauly 1993) and to recovery time (Vasconcellos et al. 1997), which is measured as Finn's cycling index (FCI). FCI is defined as the fraction of an ecosystem's throughput that is recycled. In Ecopath, it is expressed as a percentage of the total flows. This is similar to the predatory cycling index, which is calculated by excluding cycling through detritus. Disturbed systems are characterized by short and fast cycles, while complex trophic structures have long and slow ones (Odum 1969, Christensen 1995). A means of quantifying the length of each cycle is through the Finn's mean path length, which accounts for the number of groups involved in a flow. Path length will be affected by the diversity of flows and cycling. Since these increase with increasing maturity, it is assumed that long path lengths are associated with a mature ecosystem. The FCI value from the Mae Klong Estuary was 1.62% with a Finn's mean path length value at 2.278.

Discussion

The dynamic nature of estuaries in terms of biomass and species composition leads to questions about how food web structure and function are maintained under these constantly changing conditions (Livingston 2002). This study represents the first attempt to model the trophic components of the Mae Klong Estuary food web using an Ecopath model. As a first attempt, this required a considerable effort to gather information for an area that has never been studied. The Ecopath model presented here summarizes much of information that is available for the Mae Klong mangrove estuary ecosystem. The description of the Mae Klong ecosystem is based on estimations of the biomass and fish production, and on the components in the fish diet that gave an indication of the relationships between the 21 functional groups. The characteristics of the ecosystem model in this study are discussed here and compared with other coastal ecosystem models (Table 7). This is a preliminary study and several of the input variables and parameters include high uncertainty. For example, several of the estimated *P/B* ratios are higher than would be expected for the species groups that could be a result of underestimates of the biomass.

Table 6. System statistics of the Mae Klong Estuary.

Parameters	Value	Units
Sum of consumption	2013.439	t/km^2/year
Sum of all exports	2360.645	t/km^2/year
Sum of respiratory flows	1292.812	t/km^2/year
Sum of flows into detritus	2654.615	t/km^2/year
Total system throughput	8321.511	t/km^2/year
Sum of all production	3973.000	t/km^2/year
Calculated total net primary production	3657.000	t/km^2/year
Total primary production/total respiration	2.829	
Net system production	2364.459	t/km^2/year
Total primary production/total biomass	77.402	
Total biomass/total throughput	0.006	
Total biomass (excluding detritus)	47.250	t/km^2
Connectance index	0.195	
System Omnivory Index	0.117	
Throughput cycled (excluding detritus)	36.88	t/km^2/year
Throughput cycled (including detritus)	135.15	t/km^2/year
Ascendency (*A*)	10361.6 (44.3)	t/km^2/year (%)
Overhead (*O*)	13019.5 (57.7)	t/km^2/year (%)
Capacity (*C*)	23381.1 (100.0)	t/km^2/year (%)
Finn's cycling index	1.62	% of total throughput
Total no. pathways	62	
Finn's mean path length	2.278	

Table 7. Comparison of the Mae Klong Estuary with other coastal ecosystems.

Ecosystem	Throughput (t/km^2/yr)	*PP/R*	Relative ascendency (%)	Finn's index (%*TT*)	Finn's mean path length (food chain steps)	References
Quintana Roo, Yucatan, Mexico	4,815,000	3.17	–	–	–	Vidal and Basurto 2003
Southwestern Gulf of Mexico	7,712	–	–	11.5	6.8	Manickchand-Heileman et al. 1998
Tonamega Lagoon, Mexico	2,853	0.56	32.7	–	–	Avila Foucat 2006
Mont Saint Michel Bay, France	9400	6.1	–	0.64	2.1	Leloup et al. 2008
Eastern Scotian Shelf, Canada (1980-1985)	7,669	–	21.74	4.89	2.76	Bundy 2004
Eastern Scotian Shelf, Canada (1995-2000)	7,124	–	23.51	6.61	3.13	Bundy 2004
Pearl River Delta, China	15,244	2.86	–	–	–	Lijie et al. 2009
Gulf of Paria, Venezuela/Trinidad	2,285	3.8	41.7	7.2	6.2	Manickchand-Heileman et al. 2004
North coast of central Java, Indonesia	6,745	2.87	–	8.58	2.75	Nurhakim 2003
São Sebastião Channel, southeastern Brazil	11,442	0.7	25.4	30.1	–	Rocha et al. 2007
Caeté Estuary, Brazil	10,558	–	27.4	17.9	3.4	Wolff et al. 2000
Karnataka Arabian Sea, India	11,522	–	33.0	6.03	2.81	Mohamed et al. 2005
Somme Bay, France	2,312	–	35.0	12.2	–	Rybarczyk et al. 2003
Orbetello Lagoon, Italy (1995)	24,553	3.46	30.2	7.1	2.77	Brando et al. 2004
Ébrié Lagoon, West Africa	6,240	5.15	34.0	2.57	2.38	Villanueva et al. 2006
West coast of Sabah, Malaysia	3,152	2.07	–	–	–	Garces et al. 2003
West coast of Sarawak, Malaysia	1,414	2.08	–	–	–	Garces et al. 2003
Brunei, Southeast Asia	1,816	–	29.4	16.3	2.8	Silvestre et al. 1993
Mae Klong Estuary	**8,321**	**2.82**	**44.3**	**1.62**	**2.27**	**From this study**

Trophic level, energy flow, and pathways

Estimated ecotrophic efficiencies of the fish groups were generally within the range 0.7-0.9, as usually assumed for fish (Ricker 1969). The high ecotrophic efficiencies for most fish groups suggest that trophic relationships are tight and most of the system's secondary production is consumed by predators. The low ecotrophic efficiencies of detritus indicate that more detritus is entering this box than is leaving it, or that a significant quantity of detritus is being buried or exported to the seafloor (Manickchand-Heileman et al. 1998).

The predominance of fractional trophic levels <4.0 found in the present study has also been reported for other coastal areas in the Gulf of Mexico (Odum and Heald 1972, Vega-Cendejas and Arreguín-Sánchez 2001, Vidal and Basurto 2003); west coast of Sabah and Sarawak, Malaysia (Garces et al. 2003); the Swartkops Estuary, South Africa; the Ems Estuary, Germany; and Chesapeake Bay, USA (Baird et al. 1991). This may be attributed to the dependence of the food web on detritus and the abundance of juvenile fish that use the estuary as a nursery area (Yáñez-Arancibia et al. 1988), whose production depends directly and indirectly on primary producers (Arreguín-Sánchez 2001). In contrast, higher fractional trophic levels were found on the continental shelf in the southwestern Gulf of Mexico (Arreguín-Sáachez et al. 1993, Manickchand-Heileman et al. 1998) where adult fish are expected to be more abundant.

Mangroves play an important role in detritus accumulation due to the large amount of leaf material that is incorporated within the soil. Typically about half the detritus produced by fallen leaves is exported to adjacent aquatic regions mostly by tidal flush (Jacobi and Schaeffer-Novelli 1990). The other half is used by juvenile stages as a source of food by direct grazing on leaves and indirectly by detritus consumption (Lugo and Snedaker 1974, Thayer et al. 1987). The importance of these biological and energetic processes within these swamps is shown by the dependence on detritus of two-thirds of the world fisheries (Lai 1984). Increased cycling and storage both tend to increase the ratio of indirect to direct flows and contribute to network amplification and homogenization of available energy over all trophic levels (Patten et al. 1990). Detritus recycling or re-utilization involves the subsequent transformation of previously utilized but not dissipated energy-matter by consumers (Higashi et al. 1993). Mangroves are clearly very important for fueling the fish populations that are the subject of this paper, but they are under continuing pressure from human exploitation for fuel wood and timber, as well as being cleared for pond aquaculture.

The importance of detritus and primary production pathways in ecosystems, such as mangrove estuaries, was noted by Vega-Cendejas and Arreguín-Sánchez (2001). De Sylva (1985) indicated that estuarine nekton follow either a detritus-based or a phytoplankton-based food chain. Primary producers and detritus are energy sources that play differing roles and significance in the diet of fish of higher TLs in the Mae Klong Estuary. My results showed that phytoplankton and detritus are the key food sources that sustain mainly zooplankton secondary production, similar to observations in mangrove ecosystems in Sundarban, India (Ray et al. 2000), and the Yucatan Peninsula, Mexico (Vega-Cendejas and Arreguín-Sánchez 2001). Energy flow in the Mae Klong Estuary is also consistent with what is known about coastal lagoons and estuaries in general. The dominance of the detrital pathway as observed in this study has been reported for other shallow estuaries and coastal lagoons in the Gulf of Mexico (Odum and Heald 1972, Vega-Cendejas and Arreguín-Sánchez 2001); Caeté mangrove estuary, North Brazil (Wolff et al. 2000); the Swartkops Estuary of southeast South Africa; Chesapeake Bay in the eastern U.S. (Baird et al. 1991); Bay of Dublin, Ireland (Wilson and Parkes 1998); and the Kromme Estuary of southern South Africa (Heymans and Baird 1995). The high biomass of TL1 (detritus and primary producers) and its significant role in supporting the energy utilized indicate a bottom-up control in the Mae Klong Estuary.

Maturity of the Mae Klong Estuary: comparison with other coastal ecosystems

A comparative approach with other coastal ecosystems is helpful to characterize the structure and material flows in the Mae Klong Estuary. However, there are very limited quantitative descriptions of food webs for tropical/subtropical ecosystems (Lin et al. 2007).

The model estimate of total system throughput (T) of 8,321 t per km^2 per yr appears high when compared to other coastal systems (Table 7). The high biomass and production values for benthic producers, including mangroves (phytoplankton and detritus in this study), and the large organic nutrient loading from the upper reaches are probably the reasons for the high throughput values (Lin et al. 2007). These throughput values are still low, however, when compared to Quintana Roo, Yucatan, Mexico, which had T-values of 4,815,000 t per km^2 per yr.

Odum (1969) demonstrated that the primary production/respiration ratio (*PP/R*) reflects the maturity of an ecosystem. He suggested that the rate of primary production exceeds the rate of community respiration during early stages of ecosystem development, and hence *PP/R* is greater than one. However, in a mature system the ratio approaches 1 because the energy fixed tends to be balanced by the energy cost of maintenance. In their comparative study of 41 aquatic ecosystems, Christensen and Pauly (1993) found that the bulk of *PP/R* ratios were in the range between 0.8 and 3.2, although the extreme values were <0.8 and >6.4. *PP/R* values of 2.8 obtained from the Mae Klong Estuary are larger than 1, which is similar to other coastal ecosystems like Quintana

Roo, Yucatan, Mexico; Pearl River Delta, China; north coast of central Java, Indonesia; and west coast of Sabah and Sarawak, Malaysia (Table 7). This value implies that the Mae Klong Estuary and the other ecosystems are in an early developing stage and are prone to ecological perturbations, including anthropogenic impacts (Fetahi and Mengistou 2007). In contrast, the *PP/R* ratio of 0.56 in Tonameca Lagoon, Mexico, indicates that Tonameca Lagoon is probably mature and with a low level of organic matter (Avila Foucat 2006).

The *PP/R* ratio of the Mae Klong Estuary also indicates moderate eutrophication when compared with the value of 1.12 from Lake Nokoué, West Africa (Villanueva et al. 2006), which indicated a level close to "eutrophic status" as total system respiration approaches its production, a common feature in highly polluted systems. However, this may not be true if based on recent environmental domestic and industrial pollutions loads (Villanueva et al. 2006). In addition, system ascendency (*A*) and total system throughput (*T*) can also be used as indicators of eutrophication in ecosystems (Mann 1988). This is characterized by an increased value in *A*, as a function of elevated *T* parallel to a fall in information (*I*) (Ulanowicz 1986).

The estimated values of some properties, such as ascendency and development capacity, are tools to evaluate the organization, maturity, and tolerance to perturbations, as well as for ecosystem comparisons (Baird et al. 1991). According to Ulanowicz (1986), these properties tend to increase with maturity and decrease in systems under natural or anthropogenic stress. The ascendency value of 44.3% for the Mae Klong Estuary is relatively high when compared with many other coastal ecosystems (Table 7), but similar to the Gulf of Paria in Venezuela and Trinidad (41%); these values imply that the Mae Klong Estuary is more mature than other coastal ecosystems. However, Christensen (1995) and Aoki (1997) argue that ascendency is not the best indicator of the degree of eutrophication and maturation, and has a negative correlation with them, suggesting that relative ascendency should be called relative mutual information, which provides a measure of the distribution of flows in a system network in relation to the total flow.

The model identified the Mae Klong Estuary as a highly productive ecosystem and the Leontief matrix routine demonstrated that it is largely controlled from the bottom-up, which results from high nutrient inputs from river discharges draining mangroves and surrounding aquaculture ponds. However, when compared with other ecosystems (Table 7), global indicators (high primary production/respiration ratio, low Finn cycling index and mean path length) suggest that the Mae Klong Estuary ecosystem is immature, in line with Odum (1969), Finn (1976), and Ulanowicz (1986, 1995). Low maturity status is commom in megatidal coastal and estuarine systems, such as the bay of Mont Saint Michel (Leloup et al. 2008), due to the low rate of transfer of primary production (Le Pape and Menesguen 1997). Even if it is sometimes difficult to compare different systems that have different degrees of compartment aggregation, the very low values of the cycling index in the Mae Klong Estuary reflect an especially immature system.

The discrepancy in the Finn cycling index could change the interpretation of the developmental state of the ecosystem in the Mae Klong Estuary analysis. Odum (1969) found that cycling increases as systems mature (thus the FCI increases), although some discrepancies have been recorded in the interpretation of cycling with regard to ascendency and overhead. Baird et al. (1991) concluded that FCI shows the reverse rank-order correlation with ascendency, and FCI is not a measure of system maturity but of stress, while Ulanowicz (1986) defined FCI as a measure of maturity. Subsequently, Christensen (1995) has shown that not ascendency, but overhead, is related to a system's maturity, and thus an increase in FCI with an increase in overhead is an indication of system maturity. Vasconcellos et al. (1997) also found that recycling is the "chief positive feedback mechanism that contributes to stability in mature systems by preventing overshoots and destructive oscillations due to external impacts." Taking into consideration the controversy surrounding maturity and cycling of systems, it would be prudent to be careful when comparing the FCIs of systems. Futhermore, when comparing FCIs, consideration should be given to the currency used for comparison (Field et al. 1989).

The immature status of the Mae Klong Estuary trophic network may be explained partly by the intensive human exploitation of the estuary, through shellfish farming (blood cockle and horse mussel) (Alongi 2002). There may also be impacts in the estuary due to wider fishing activity offshore in the Gulf of Thailand (Christensen 1998, Christensen and Pauly 1998) because many commercial species breed in the estuary and use it as a nursery ground. These are large losses of primary production due to hydrodynamic exchanges (Le Pape et al. 1999).

It can be seen that the Mae Klong Estuary has a mixture of characteristics of a mature system (high total system throughput, ascendency, and overhead) as well as an immature system (high primary production/respiration ratio, low Finn's cycling index and mean path length). In addition, detritus-based food webs, and high fish and flow diversities are typically related to maturity. This is consistent with the system experiencing a moderate level of exploitation, driving its development back to earlier developmental stages.

It should be noted that this is a preliminary study. Further studies will be conducted in the near future. However, the results indicate the need of management and conservation between the two sectors of the fisheries and forestry, whose present trajectories tend toward further degradation of the Mae Klong ecosystem. Future management policies should consider resource conservation, as such systems serve as nursery grounds for many commercially exploited resources.

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FISKE2020: Toward Ecosystem-Based Fisheries Management in Sweden

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Abstract

The current state of fish stocks in Europe has resulted in a demand for new novel ideas and long-term strategies in fisheries management. In order to fulfill the challenges of future fisheries management, FISKE2020 was initiated by the Swedish government agency responsible for fisheries management.

FISKE2020 is a vision for Swedish fisheries that describes the aims and objectives of national fisheries management for the forthcoming 10 years. It portrays a clear path on how to reach a set of target levels in the year 2020 for all areas of Swedish fisheries management: recreational and professional fisheries, aquaculture, and fishing tourism. The aim of the study is to guide future policymakers and to contribute to the debate on sustainable fisheries. The study was conducted in consultation with 19 stakeholders including fisheries organizations, universities, and the European Commission. To achieve the targets for 2020 two key instruments have been identified: ecosystem plans and an increased gear selectivity to ensure that fish are caught after they have reached the optimal size (*Lopt*) where cohort biomass is maximized. In addition, incentive-based management and a socioeconomic allocation of resources between user groups are proposed.

Introduction

With 30% of global fish stocks (FAO 2010) and with 62% of European fish stocks in the northeast Atlantic being overfished or depleted (European Commission 2010), some to the point where it is doubtful that rebuilding schemes can be successful, and with the severe adverse impacts of fisheries on marine ecosystems and biological diversity (Palumbi et al. 2008), there is an urgent need for re-evaluating fisheries management and reconciling management with the ecosystem approach (Garcia and Cochrane 2005). To date, fisheries have been managed according to single species models based on single species assessments (Pauly et al. 2002), i.e., a target resource-oriented management, in spite of the obvious effects of fisheries on the whole ecosystem. Single species management has become increasingly questioned and criticized as part of overall management shortcomings.

Applying the ecosystem approach to fisheries management ultimately means that fisheries management and ecosystem management are joined together to form an ecosystem-based fisheries management (EBFM). An EBFM combines the ecological objectives, i.e., restoring and preserving the structure, diversity and function of ecosystems, with societal needs of fish as a source of food and the sea and coastal zone as a source of recreation (FAO 2003b).

The ecosystem approach in fisheries has been debated and criticized for being too complex to enable a practical approach and consequent implementation by managers (Rice 2008 and literature cited therein). In order to overcome this obstacle the Food and Agriculture Organization (FAO) has published guidelines for making the ecosystem approach operational. According to these, the ecosystem approach must be divided into several different components, each with overarching aims, which are further subdivided into prioritized issues under which operational goals are set. These operational goals must then be directly linked to measureable indicators and reference points. In addition, rules for decision-making must be established as well as how management measures are finally enacted. Management measures must then be monitored and evaluated for their performance. In addition, in an ecosystem approach to management it will be necessary to assess the risk that things may occur that will affect the possibility of the management system to perform in accordance with the set objectives. Risk assessment can also be used to set priorities at the planning stage that can help resolve some of the objections to the principles and guidelines proposed by FAO, i.e., that the components of the ecosystem approach are numerous and their internal hierarchy cannot be clearly established, and hence the overarching aim of the ecosystem approach is lost.

In the study FISKE2020, that among a range of topics covering fisheries management provides the framework for designing fishery ecosystem plans, we argue that a holistic management applied in an area on a proper and relevant scale with clearly defined goals, a predefined hierarchy between the goals and well-chosen indicators are components on the way to achieve an EBFM. To that end, we propose that managers draw up fishery ecosystem plans, tailored to meet the needs of a specific management area and that incorporate all parts of the ecosystem in the decision process. A fishery ecosystem plan covering a specific area will have the following objectives: (i) to rebuild fish stocks to a size and age structure that closely resembles an unfished stock by introducing optimal length at catch *(Lopt)* as a founding principle; (ii) to protect sensitive habitats; (iii) to protect threatened species; and (iv) to restore and safeguard the viability and resilience of the ecosystems. To reach the objectives under (ii-iv*)* it is proposed that the characteristics of the area covered by a fishery ecosystem plan, e.g., rate and extension of sensitive habitats and threatened species, depicts when and where fishing may take place, i.e., full-scale zoning of fisheries in the area.

The objectives described above refer to the direct impact of fisheries on the marine environment. However, fisheries are highly complex human-in-nature systems and several multidimensional, interconnected factors have to be addressed within a fishery ecosystem plan in order for the plan to be successful (Norse 2010). Thus, an ecosystem plan must add further dimensions in the form of social and economic objectives with clear incentive-based approaches and socioeconomic allocation of resources between user groups. However, “area” will be the overarching factor that decides the appropriateness of the measures chosen. Below we elaborate further on the set of measures that could be used in a fishery ecosystem plan to address the different dimensions of fishing and the ecosystem. Finally, we will discuss the nested, interconnected factors intrinsically linked with the area chosen for any fishery ecosystem plan. These factors have been identified as, for example, extent of sensitive habitats and state of fish resources, distribution of stocks, scales of stock components and fishing operations, social and economic patterns and networks, and governance structures (Crowder et al. 2006, Norse 2010, Lorenzen et al. 2010), and should be taken into consideration when deciding on meaningful boundaries for a fishery ecosystem plan. Lastly we make some conclusions on how we will reach the ultimate goal of fishing with an ecosystem-consciousness.

Method

During 2010, the Swedish government agency responsible for fisheries management carried out an evaluation of the need for continued management efforts to establish functioning ecosystems and ecosystem services. The conclusion reached was that a strategy with concrete decision strategies for fisheries management is crucial to reach long-term objectives. From this standpoint the project FISKE2020 (translated to FISHERIES2020) was initiated with the aim to provide advice to policy makers on fisheries management and to contribute to the debate on the current reform of the EU’s (European Union) Common Fisheries Policy. FISKE2020 portrays a clear path on how to reach target levels in the year 2020 for all areas of Swedish fisheries management: professional and recreational fisheries, aquaculture, and fishing tourism. In total 32 objectives divided into eight management areas were defined.

The strategy outlined can be applied to inland lakes as well as to territorial waters and the coastal zone (within 12 nautical miles), which are nationally regulated within the overall regulatory framework of the EU[1]. For the remaining part of Swedish waters, FISKE2020 refers to how Sweden, as a Member State of the EU, could influence the development, and how Sweden could apply and complement decisions taken within the EU. The evaluation utilized the knowledge of national experts from different fields of fisheries management, and also held a consultation process with stakeholders such as fishers, scientists, and NGOs.

A form of backcasting technique was used as a method in which the status in 1999 and today were compared to, based on current trends, how the situation in 2020 was perceived to be. In the case where the situation in 2020 deviated from the objective, concrete management measures were suggested as a means to reach the targets. To achieve the targets for 2020 two key instruments were identified: fishery ecosystem plans and increased gear (size) selectivity to ensure that fish are caught after they have reached the optimal size (*Lopt* strategy) where growth rate and cohort biomass are maximized. In addition, incentive-based management and a socioeconomic allocation of resources between user groups were proposed. These instruments are presented here.

Fishery ecosystem plans

Fisheries policy in the EU and lack of area considerations

Within the European Union, fish stocks are managed as a common resource through the fully fledged community policy "the Common Fisheries Policy," which was first adopted in 1982 but since revised. Scientists as well as managers and stakeholders have for the past years given voice to the shortcomings of this policy as regards securing resource conservation and sustainable utilization. With stocks in severe decline, 80% of the stocks in the EU being fished above MSY targets (Froese and Proelss 2010), and persistent problems of overcapitalization and overcapacity in the fishing fleet despite extensive scrapping programs, the time has come for a reform of this policy. Thus, the European Commission has launched a Green Paper on the reform of the Common Fisheries Policy (European Commission 2009). The Green Paper points to five major structural failings: fleet overcapacity, imprecise policy objectives, short-term decision-making, lack of responsibility by the industry, and a general lack of compliance. In our view, in addition to the above, the current legislative acts concerning fisheries (e.g., Council Regulations, Commission Regulations) act without clear link-

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[1] The jurisdiction for EU Member States to regulate national waters is given in article 9 in (European Commission) no. 2371/2002.

ages between their individual objectives and with little or no reference to each other. They contain elements of managing fisheries in specific areas and have, to some extent, taken considerations to the specific area needs but they are too fragmented and nonaligned to benefit management on an area level. However, all the necessary management instruments for achieving holistic area management of fisheries, i.e., fishery ecosystem plans with full consideration for the area, are present, e.g., the possibility for selectivity measures and zoning of areas (including zones where no fishing may take place), but as FAO has justly pointed out, the instruments are not used (FAO 2004).

In our view, it is necessary for future fisheries management to take the full step to an area-based management in fisheries, where the area dimension is explicitly and holistically considered, i.e., the adoption of fishery ecosystem plans as legislative acts on national or EU level, applied with full consideration for the area. We argue that area-based management in fisheries will lead to an ecosystem-based fisheries management that ultimately will mitigate at least some of the factors in fisheries that lead to unsustainability, defined by FAO as being inappropriate incentives, a high demand for a limited resource, poverty and lack of alternatives, complexity and lack of knowledge, lack of governance, and interactions of fisheries sector with other sectors and the environment (FAO 2004). The support for area-based management in fisheries is ever increasing and some scientists describe the current trends toward consideration of "place" or "area" in ocean management as a paradigm shift (Norse 2010, Lorenzen et al. 2010, Worm et al. 2009) and that "The time has come to consider a more holistic approach to place-based management of marine ecosystems: comprehensive ocean zoning" (Crowder et al. 2006).

Designing fishery ecosystem plans

In order to fully account for the range of impacts that fisheries can have on the ecosystem, such as targeting and depleting the oceans' top predators; altering food webs; depleting sensitive species like sharks, rays, and deep-sea species either through directed fisheries or through bycatches; and causing severe effects on sensitive habitats through destructive fishing practices, the perspective of fisheries management must broaden to encompass area-specific needs for conservation and protection. Thus, we believe that a natural step away from today's fragmented fisheries management within the EU would be to develop fishery ecosystem plans that take into account the cumulative impact of individual fisheries management decisions on the ecosystem. The fishery ecosystem plans are suggested as a management tool to shift away from single species management and to incorporate all parts of the ecosystem in the decision process.

The foundation for a fishery ecosystem plan should be a geographical area and should be based on the guidelines developed by FAO (2003a) as follows:

- **The overall objective,** relevant to the area should be identified. The overall objective must then be broken down into a subset of prioritized issues and other issues that can be met by well-chosen management measures;
- **operational** objectives should be set;
- **indicators** and **reference points** need to be used (or developed) as a direct measure of the operational objectives;
- **decision rules** for how management measures are to be applied should be developed, including risk assessment;
- the implementation should be **monitored** and **evaluated**.

The fishery ecosystem plans should be defined for different areas ranging from inland lakes to areas of the ocean. The foundation for an area-based ecosystem fishery plan should be zoning in space and time, tailored to fulfill the following objectives: (i) to rebuild fish stocks to a size and age structure that closely resemble the unfished stock by introducing optimal length at catch (*Lopt*); (ii) to protect sensitive habitats; (iii) to protect threatened species; and (iv) to restore and safeguard the viability and resilience of the ecosystems. In that regard the zoning is the depiction of where and when fishing activities may take place and what gears should be used and what selective measures should be taken in order to reach, and fish at, *Lopt,* in the respective zones. The zoning will rest on the following principles:

- The best available practice/technique shall be applied.
- The most restrictive habitat, stock, or species guide the tailoring of the measures.

In this regard, zoning refers to the separation of incompatible interests between the fishers on the one hand and the protection of fish stocks and conservation objectives for the marine environment, on the other. The zoning must, however, be on multiple scales and thus also account for the social and economic dimensions of fisheries, which is further described below. Zoning with reference to social and economic factors will for example enable the separation of incompatible interests in the coastal zone, such as small-scale versus large-scale fishing, and enable managers to use socioeconomic values as a basis for allocation of resources and reward fishers that honor rules and who fish with consideration for the wider marine environment.

Fishing and the ecosystem

Overfishing has worldwide been identified as one of the most severe threats to the oceans' ecosystems (Jackson et al. 2001, Pauly et al. 2002, Crowder et al. 2008). Ecosystems are suggested to react in different ways to fishing pressure depending on their inherent characteristics, where species-rich systems with a high degree of complexity of species and food strategies and where there is a greater connectivity between species, are proposed to better withstand fishery-driven collapse (McCann 2000, Bascompte et al. 2005, Worm et al. 2006). In addition, there is growing awareness of the importance of spatial dynamics and spatial heterogeneity of fish species. It has been proposed that despite wide geographic distribution many exploited species function demographically as much smaller local stocks (Steneck and Wilson 2010). In terms of fishing possibilities and quotas, the development over the past ten years has been dramatic for most Swedish stocks. The quota for cod in the Skagerrak has decreased by 78% from 1999 to 2009 (from 2660 t to 579 t) and the concomitant reduction in biomass is somewhere between 65 and 70% (ICES 2009). During the same period, the quota for cod in the Kattegat decreased by 93% (from 2590 t in 1999 to 187 t in 2009). The concomitant reduction in biomass has been somewhere between 80 and 90% (ICES 2009). There is an urgent need to change the current fishing pressure and pattern in order to safeguard the future of the targeted species and their ecosystems.

Lopt as a tool to rebuild fish stocks to a natural size and age structure: the Baltic example

An ecosystem-based management aiming to achieve well-functioning ecosystems, that provide the full range of ecosystem services, requires detailed knowledge about the structure and functioning of ecosystems. Large predatory fish have been identified as key components of ecosystems (Casini et al. 2009, Myers et al. 2007). Fishing, due to the present size selection, has led to the depletion of large individuals. Thus, one possible step toward ecosystem-based management could be to apply the principle that fish should be caught only after they reach their optimal length (*Lopt)* based on the stock's life history characteristics (Froese et al. 2008). The optimal length (*Lopt*) is defined as the body length when an unfished age group reaches its maximum biomass. This optimal length depends mainly on the growth rate of the fish and its natural mortality. As a rule, for most fish species the optimal length means that, by a wide margin, fish of that size have matured and been able to reproduce several times (Fig. 1).

We argue that managing fish stocks according to the *Lopt* principle, i.e., increased size selection targeting only fish after they reached their optimal length, could be an important component of an ecosystem-

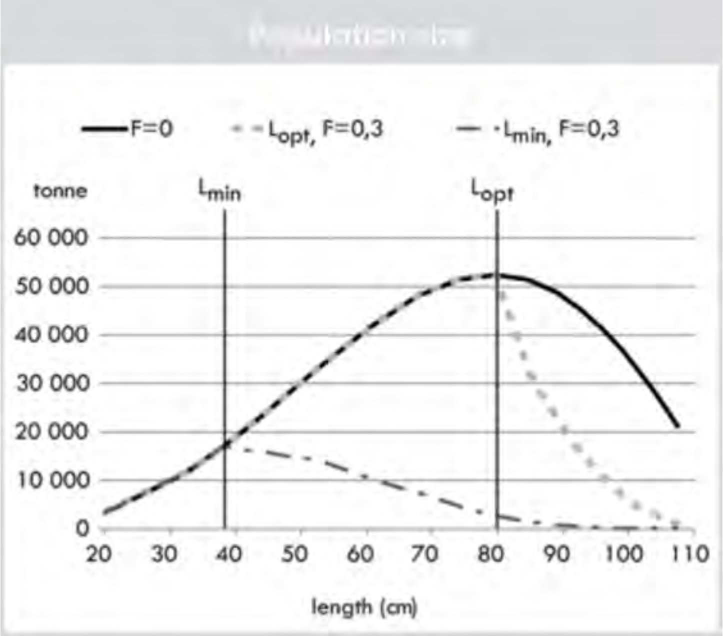

Figure 1. Population size in relation to length, based on knowledge of eastern Baltic cod, for two scenarios with different minimum sizes, and one scenario with no fishing (*F* = 0). *Lmin* = current minimum size limit; *Lopt* = optimal length according to the principle described. The solid curve represents the size structure of an unfished population (*F* = 0); the long and short dashed line represents the size structure when today's minimum size limit (*Lmin*) is applied in combination with current fishing mortality rate (*F* = 0.3); the dotted line represents the size structure if the optimal length (*Lopt*) is applied, combined with current fishing mortality rate (*F* = 0.3).

based management, even though the measure is aimed at individual stocks. The main benefit of *Lopt* is the rebuilding of stocks to resemble more closely the size and age structure of an unfished stock, which would allow opportunities for all species to play their role in the ecosystem.

Even though there are alternative harvesting strategies that could result in a size and age structure of an unfished stock, the *Lopt* principle may be a pragmatic harvesting scenario but only if the fishing mortality is kept at a low level to avoid depletion of older and lager individuals, i.e., the fishing mortality rate needs to be adjusted to an appropriate level. The biological advantage with the *Lopt* principle is that fishing takes advantage of the maximum production capacity of the fish stocks, which means that the impact on the population can be minimized. The *Lopt* principle is especially applicable in simple ecosystems and single species fisheries, as in the Baltic, but could also be valid for a mixed fishery where an increased size selection would allow better production of all targeted species.

Even though the *Lopt* principle is appropriate to most commercial stocks, it may not be suitable for sensitive species that are slow-growing and slow to mature and that rear a limited number of young for a prolonged period or breed only once, for example sharks, deep-sea species, and catadromous fish.

The *Lopt* principle requires a transition period for rebuilding the fish stocks, and a change in current regulations. During this period of transition, changes in management would mainly consist of reduced fishing pressure and gradually increased size selectivity in the fisheries.

In the Swedish fishery for Baltic cod it has been found that the increased size selectivity necessary to reach *Lopt* would result in an initial loss in economic revenue in the first years of implementing the system. However, due to the current size-dependent price relationship of cod the effect is offset and the management strategy results in an increase in economic revenues in the long run (Dr. J. Hjelm and Dr. M. Cardinale, Swedish Board of Fisheries, 2011, pers. comm.).

Most fisheries are multispecies fisheries with substantial bycatches, requiring additional solutions for encouraging fishing for individual species as separately as possible. This requires the continued development of species-selective gear, temporal regulations, such as closed seasons, and spatial regulations like closed areas and other types of zoning. The same effects can to a certain extent be achieved through having management measures aimed at the largest or most sensitive species, allowing the others a free ride, so to speak, which entails regulations based on the biologically most sensitive/important species.

The *Lopt* principle demonstrates how we, given the understanding and means at hand today, may take a significant step toward ecosystem-based management without drastically reducing future catches and revenues. If applied as a management measure, it is important that the *Lopt* principle is evaluated at regular intervals from a broader ecosystem perspective. Applying the *Lopt* principle and evaluating its performance as a management tool should result in consequently adapting management measures to further improve the result of the *Lopt* principle in an adaptive management cycle.

Incentive-based approaches to management in the context of a fishery ecosystem plan

Long-term fishery ecosystem plans in combination with management measures such as the *Lopt* strategy can steer fisheries in the desired direction. However, these management tools do not inherently take into account the impact that social and economic drivers and fishers' behavior have on the outcome of fisheries management. FAO has identified inappropriate incentives as the foremost factor leading to unsustainability in fisheries (FAO 2004). Thus, in order to promote sustainability, incentive-based approaches that clearly specify community and/or individual harvest or territorial rights must be included in the fishery ecosystem plans to complement other management approaches.

An essential management strategy, to create incentives for more sustainable fishing behavior, is to provide harvesters with secure harvesting or territorial fishing rights (Grafton et al. 2006). Provided that the system is set up appropriately, this enables fishers to fully benefit from long-term conservation as well as directly bearing the costs of overexploitation of the resource.

Transferable fishing rights

As an EU Member State, Sweden may take national nondiscriminatory measures for the conservation and management of fisheries resources on national waters (i.e., within 12 nautical miles of its baselines) as long as the EU has not adopted measures addressing conservation and management specifically for this area. That enabled the Swedish Parliament to issue a new law in 2009 allowing transferable fishing rights in the pelagic fishery. Quota trading began as soon as the implementing regulations came into force in November 2009.

The pelagic fishery is the first Swedish fishery for which tradable individual fishing rights have been introduced. One of the reasons for introducing the ITQ (individual transferable quota) system was to create incentives for reducing overcapacity. It is yet too early to evaluate the effects and consequences of the system but it is estimated that approximately 50% of the vessels left the pelagic fishery in the first year.

We argue that, in order to overcome current problems of overcapacity and to maximize the net economic return from the sector, fishing rights in the form of individual, community, group-based, or territorial rights should be introduced in the majority of Swedish fisheries. A comprehensive, fully fledged system of fishing rights is needed, and should be designed to be effective in the long-term and adaptable to each sea area and the fishery ecosystem plan in place. We suggest that the biological dimension of the zoning exercise, i.e., consideration of place and space of the fish resources and for the protection of the marine environment, should be considered when deciding which form of fishing right (individual, community etc.) should be introduced. With territorial rights, the scale of the fishing operation could be made more predictable, and thus effects on spatial heterogeneity of stocks may be diminished. The most common objection against the introduction of systems of transferable fishing rights in Sweden has been the fear of marginalization of small-scale fisheries. However, it has been found in a recent EU-wide study that schemes for small-scale fisheries, such as a separate quota allocation, and/or prevention of consolidation can be introduced alongside the system and result in the protection and continued participation in the small-scale fisheries (European Commission 2007).

Fishing contracts

A large part of the Swedish fishing activity is being granted through so-called special permits. We suggest that these be developed further to encompass a wider environmental concern. In order to clarify the conditions for exploiting the fishing resource, the fishing rights should be accompanied by a contract between the managing authority and the rights holder. The contract should clearly indicate the terms for fishing, such as the right to access all or certain fishing zones under a fishery ecosystem plan, when access is permitted and with which selectivity fishing can be conducted.

We argue that the fishing technique should be adapted to the ecological characteristics of the area, and the principle of best available technology should be applied. Furthermore, as a means to increase compliance the managing authority should have the legal right to withdraw the fishing rights, temporarily or permanently, if the obligations of the fishers are not fulfilled, despite the financial implications for the user.

Cost recovery scheme

In order to ensure functioning ecosystems and profitable fisheries a system of management and fisheries control needs to exist. We therefore suggest that part o the returns from fishers should be captured by the Swedish authorities to cover some costs of enforcement and management. Apart from recovering taxpayers' money invested in fisheries management, another benefit of cost recovery schemes is that it puts pressure on the managing authority to improve their cost effectiveness. Furthermore, it could lead to improving transparency and accountability in the delivery of fisheries management services (Harte 2007).

Cost recovery schemes are not a new concept in fisheries management outside Europe and in some countries the schemes play a significant rule in funding their management regimes. New Zealand, Iceland, and Australia, for example, have regained 50%, 37%, and 24% of the public costs of fisheries research, management, and enforcement from the industry (OECD 2002).

In Europe, most fisheries management programs are paid entirely by the state, which is also the case in Sweden. An implementation of a cost-recovery scheme for Swedish fisheries would require a change in the current EU or national legislation. As a matter of fact, in the proposal of a new Common Fisheries Policy, the European Commission mentions the possibility for Member States to introduce schemes to recover costs for implementing control systems and for management-related costs, for parts of the national fisheries (European Commission 2011).

There are various options on how to link the fee to the fishing activity. For example, in New Zealand the quota fee is linked directly to the quota share (Annala 1996). Another option could be to link the fee to the estimated catch value. For Swedish fisheries we suggest to follow the approach implemented in New Zealand, i.e., to link the fee directly to the size of the quota share. Furthermore, we believe that the fee should also be related to the gear being used, and hence serve as an incentive to encourage the use of environmentally friendly fishing methods.

Socioeconomic allocation

The Swedish fishing resource is a common national resource utilized by recreational and professional fishers, aquaculture, and fishing tourism companies. Currently, there is an evident focus on commercial fisheries in the legislations on the European as well as on the national level. The present lack of a strategy for allocation of the resources has resulted in tension between the user groups.

When the fisheries ecosystem plans are implemented as the basis for fisheries management, the effect that each category of fishers has on the ecosystem will be embedded in the management decision. To better utilize the available fishing resources we suggest that the management authority should allocate the resource between different categories of fishers on the basis of socioeconomic evaluation criteria. This would enable a prioritization of user groups not only based on revenues and track records but also on social and recreational values.

Allocation by the government, as opposed to allocation by the market, has the advantage that it builds on existing processes and has structural simplicity. On the downside it does not facilitate an optimal economic allocation of resources and it can worsen competitive lobbying (FAO 2006). As a means to reward fishing techniques with less impact on the ecosystem, we suggest that the allocation should also be considered within a fishing category. Furthermore, we suggest that the socioeconomic allocation of the resources should be applied within the fisheries ecosystem plans where zoning and spatial planning are important spatial tools.

Discussion

FISKE2020 presents a management model based to a large extent on spatial planning of fisheries and selective measures for fishing at optimal length (*Lopt*) and based on ecosystem requirements, as far as these requirements can be perceived by managers. The management instrument evolving from this model is a fishery ecosystem plan that hosts the full zoning, i.e., ecological and social, of a management area. The fishery ecosystem plans will rest firmly on the principles that best available practice and technique will be used; that the most restrictive stock, species, or habitat will guide the tailoring of the plans; and on incentive-based approaches. To that end, the fishery ecosystem plans provide a comprehensive and useful framework that may be applied to any area in need of a coherent, ecosystem-based management instrument.

However, the fishery ecosystem plans as outlined herein do not constitute a detailed prescription and thus here we elaborate further on some issues that need to be discussed in relation to drawing up fishery ecosystem plans. For example, we do not describe the boundaries of the plans, e.g., geographical, biological, economic, social, or in terms of governance structures. We strongly believe, however, that factors intrinsically contributing to or defining boundaries must be duly considered when drawing up a fishery ecosystem plan in order to identify meaningful boundaries for the plans. Addressing the issue of meaningful boundaries will benefit the performance of the fishery ecosystem plan. By necessity, these factors differ with the management area. It has been argued that correcting for the mismatch of governance structures (Crowder et al. 2006) and recognizing the need to fully integrate the different policy sectors with activities in the maritime space (Wakefield 2010) are crucial factors in ocean management, but these factors are beyond the scope of a fishery ecosystem plan.

Fisheries are complex socio-ecological systems as are all human used resources (Ostrom 2009). Addressing the social boundaries that determine the fishery ecosystem plan includes considering factors such as local social networks in fisheries communities, different reservoirs of knowledge, experience and understanding of local fisheries, and the possibility of generational transfer of local ecological knowledge (Symes and Philipson 2009). We suggest that the selection of important social factors, such as the ones described above, should form the basis for the subsequent decision on the socioeconomic allocation that will be an integral part of fishery ecosystem plans. We propose that considering and subsequent rewarding of appropriate social factors, that can aid in the course toward sustainable fisheries through a sense of stewardship and participatory management by implementing fishery ecosystem plans, are highly important for the success of the management measures.

The spatial heterogeneity of marine ecosystems has gained increased recognition but is presently not part of management considerations (Lorenzen et al. 2010). The failure of accounting for spatial heterogeneity of fish stocks, i.e., that regardless of geographic distribution, fish stocks function demographically as much smaller, local stocks (Steneck and Wilson 2010), has most probably been contributing to the severe decline of some very important fish stocks. This can to a large extent be attributed to the use of indiscriminate fishing methods in combination with increased fishing pressure, in itself the consequence of a historic increase in the demand for fish as a source of food. Here we argue that adopting the *Lopt* principle and subsequent selectivity measures will lead to increased consideration for spatial heterogeneity and promote the rebuilding of local fish populations. Spatial heterogeneity of fish stocks should, in addition, be accounted for in the designation of fishing zones in a fishery ecosystem plan. This could mean setting aside areas for the benefit of the viability and function of local stocks so they can contribute to overall stock increase.

In addition to the consideration and tailoring of measures to account for spatial heterogeneity of fish stocks, fishery ecosystem plans must fully recognize and select appropriate measures that meet the needs to manage areas designated for conservation purposes, e.g., preserving biodiversity hotspots such as sponge reefs, coldwater corals, and coral gardens as well as protecting threatened and declining species from fishing activities.

With FISKE2020 we present a course that is far from easy, but neither is it impossible; and we believe that, within a decade, it could lead to fishing being carried out within the framework provided by a sustainable ecosystem. The long-term optimum goal is to create fisheries that take place with an ecosystem consciousness, based on the collected knowledge base and reciprocal knowledge transfer of fishers, scientists, conservationists, and managers.

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Evidence for Shifts in Demersal Fish Populations on the West Coast of South Africa: 1986 to 2009

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Abstract

Long-term ecosystem changes of demersal fish assemblages on the west coast of South Africa were explored using research survey data collected annually over the past 24 years (1986-2009). Multivariate analyses of demersal fish assemblages show differences in spatial (latitude and depth) and annual factors, but no seasonal differences were detected. Temporal shifts in demersal fish populations over the past 24 years are further investigated in this study using the Sequential T-test Algorithm to test for Regime Shifts (STARS). Long-term changes were detected in 27% of demersal species with the majority of species shifts occurring in the early to mid 1990s or in more recent years (mid 2000s). The first shift period is generally associated with an increase in density of many species, while the second period of change reflects a decrease in density of many species. Shifts detected in the demersal fish assemblage in this study temporally follow spatial shifts observed in west coast rock lobster and small pelagic fish species, and regime shifts detected in the environment, specifically sea surface temperature and upwelling anomalies on the west coast. This study is the first to document shifts in demersal species of the southern Benguela ecosystem and thus supports the need for continued ecosystem-based management for the lucrative South African hake trawl fishery.

Introduction

Long-term ecosystem changes observed in the Benguela region have been classified as species alterations, species dominance shifts, or regime shifts (Cury and Shannon 2004, Jarre et al. 2006, van der Lingen et al. 2006). An ecological regime shift may be defined in several ways. Generally, a regime shift can be an abrupt change from a stable ecosystem state, acting over a large spatial scale and resulting in trophic restructuring that persists to allow an alternative stable ecosystem state to exist (de Young et al. 2004, Jarre et al. 2006). There are several agreed criteria that define regime shifts in the ocean. These include the duration of the shift being relatively short in comparison to the length of the actual regimes, and changes being recorded across a wide range of trophic levels and species that reflect the state of the ecosystem (de Young et al. 2004). Quantifying the state of an ecosystem is not a simple task and requires a combination of analyses, frequently including models and indicators. An indicator can be defined as a variable, pointer, or index, whose position or trend in relation to reference points reflects the present state and dynamics of the system (Jarre et al. 2006). Much research has recently been conducted on the applicability and types of quantitative ecosystem indicators that can be used to clearly identify the state of marine ecosystems (Rochet and Trenkel 2003, Daan et al. 2005, de Juan et al. 2009, Shin et al. 2010). Community-based indicators, rather than single-species measures, can be considered likely to best reflect ecosystem status (Fulton et al. 2005) and a variety of indicators, used simultaneously, most effectively captures several key functional groups (Jarre et al. 2006, Shin et al. 2010).

Dramatic shifts in fish community structure have frequently occurred in highly productive clupeoid populations in upwelling ecosystems (Jarre-Teichmann et al. 1998, Jennings and Kaiser 1998, de Young et al. 2004, Roy et al. 2007, Coetzee et al. 2008). External pressures on species, anthropogenic or natural, are likely to be of great significance, particularly over the period at which a regime shift is actually taking place and can magnify and accelerate shifts (Jennings and Kaiser 1998, Rothschild and Shannon 2004, Shannon et al. 2004). While communities are undergoing a regime shift, particular species may be more sensitive to pressures such as fishing and/or environmental effects, and the outcome of the imminent regime shift may be influenced by such pressures (Rothschild and Shannon 2004). Fishing pressure may alter the speed, magnitude, or nature of the shift (Jennings and Kaiser 1998). Cury and Shannon (2004) investigated the possibility of regime shifts having occurred in the northern and southern Benguela regions. They concluded that the fluctuations between sardine and anchovy dominance observed in the southern Benguela between the mid 1980s and 2000 did not affect several ecosystem levels and hence these changes were considered species dominance shifts rather than regime shifts. Persistent low levels of exploited fish catches and altered trophic level energy transfers observed in the northern Benguela ecosystem, however, indicate a clear regime shift to have occurred in the late 1980s and early 1990s in this ecosystem (Cury and Shannon 2004). More recently, Howard et al. (2007) investigated several biological, environmental, and anthropogenic forcing variables in the southern Benguela ecosystem using a Sequential T-test Algorithm for Regime Shift (STARS) detection (Rodionov 2004). This study detected two major long-term regime shifts in the southern Benguela since the 1950s. The first change occurred in the late 1950s and is believed to be largely a result of intense fishing pressure along with some environmental changes. The second long-term change spanned the late 1990s and the early 2000s and appears to be linked to environmental rather than anthropogenic forcing (Howard et al. 2007). Robust environmental shifts were detected by Howard et al. (2007) in 1991 and 2004/2005 in response to the environmental forcing variables of sea surface temperature (SST) and upwelling anomalies.

Regime shifts are likely to have profound effects on fisheries targeting those species influenced by the shift. In the southern Benguela region, a persistent change in relative abundance of small pelagic species (anchovy, *Engraulus encrasicolis,* and sardine, *Sardinops sagax*) on the west versus the south coasts occurred between 1996 and 2001 (van der Lingen et al. 2006). There is little convincing evidence for observed shifts in pelagic fisheries being a result of fishing effects alone (Shannon et al. 2004, 2008, 2009). The abrupt change in spatial distribution in anchovy in 1996 has been linked to changes in the cross-shelf sea surface temperature gradient between coastal and mid-shelf waters on the south coast (Roy et al. 2007). Reasons for the gradual spatial shift of sardine are not clear, but it is hypothesized that favorable environmental conditions on the south coast combined with increasing fishing pressure on the west coast are likely to have driven this change in relative spatial distribution (Coetzee et al. 2008). As a result, the resources, and thus the purse seine fishery, shifted concentration onto the south coast of South Africa in the early and mid 2000s (Coetzee et al. 2008).

Similarly, a major shift in the concentration of west coast rock lobster (*Jasus lalandii*) resource from the traditional fishing grounds on the west coast of South Africa to more southern grounds was observed between the late 1980s, early 1990s, and at the turn of the century (Cockcroft et al. 2008). This spatial change in relative abundance remains persistent to date. Despite studies focused on the variability of the physical environment, and other changes in this valuable resource (reduced somatic growth and increased lobster mass strandings), the causes for the eastward shift in rock lobster abundance remain poorly understood. It is currently not known whether there are any common causes between the observed spatial changes in small pelagic and benthic rock lobster resources in the southern Benguela (Cockcroft et al. 2008).

Changes in pelagic species abundance may interact with pelagic stages (larvae or juveniles) of certain demersal species in terms of predation, recruitment success, and competition for food (Hislop 1996) and with pelagically feeding demersal fish such as hake. Thus, shifts in dominance, abundance, and/or spatial distribution of pelagic fish species off South Africa may well be related to or linked to changes in the demersal community.

South Africa's deep-sea hake fishery primarily targets *Merluccius paradoxus* in waters of 200 m to 600 m depth off the west coast of southern Africa (Wilkinson and Japp 2005) and forms the focus of this study. The South African fishing grounds for this species extend in a continuous band from off Port Nolloth on the west coast to the southern tip of the Agulhas Bank (Fig. 1). Overviews of this fishery are provided by Griffiths et al. (2004) and Atkinson et al. (2011).

Annual demersal research surveys have been conducted since 1986 by the former Department of Environmental Affairs and Tourism, now Department of Agriculture, Forestry and Fisheries (DAFF), primarily to monitor and manage the status of the hake stock and collect biological information for several other key species. Atkinson et al. (2011) used multivariate techniques to analyze the demersal data series from 1986 to 2009 and detected a significant change in the demersal fish assemblage occurring between 1992 and 1993 and a possible second shift in the mid 2000s. Through interrogation of the same long-term (24 years) demersal fish species data, this study aims to detect changes in density at the species level, compare these to the changes detected in demersal assemblage composition by Atkinson et al. (2011), and link these to previously identified periods of ecosystem change or resource shifts in the same region. Furthermore, as a consequence of fishing a decrease in the abundance (measured through biomass) of late-maturing, slow-growing species is hypothesized, with either no change or an increase in faster-growing, early maturing species. Species showing population shifts are further categorized accordingly to assess whether this hypothesis is supported.

Methods

Demersal fish density data analyzed in this study were collected from annual research trawl surveys on the west coast of South Africa (west of 20°E, Fig. 1) during the austral summer (January/February) spanning 1986 to 2009. All data analyzed in this study were collected using South Africa's research vessel FRS *Africana*; however, the trawl gear configuration was altered prior to the 2004 survey. From 1986 until 2003 and in 2006 the "old" trawl gear consisted of a two-panel 54.8 m (180 ft)

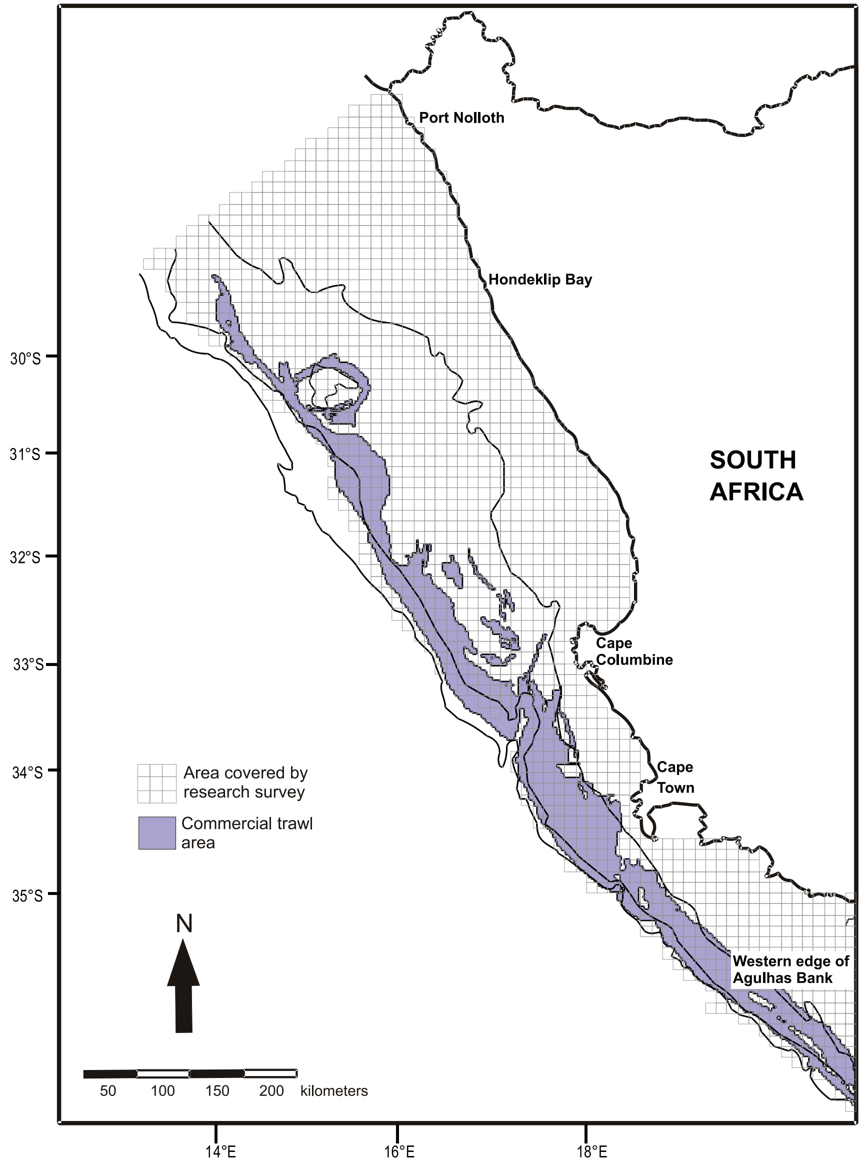

Figure 1. **West coast area covered by annual demersal research trawl survey and commercial trawl area. Data source: Department of Agriculture, Forestry and Fisheries; Wilkinson and Japp (2005).**

German otter trawl, 50 m sweeps and 1.5 t "W" Vertical otter boards. Door spread was approximately 120 m, mouth opening 2 m vertical and 26 m horizontal with a rope-wrapped footrope chain (Yemane et al. 2008). From 2004 until 2009, excluding 2006, the "new" trawl gear used was a four-panel 180 ft German otter trawl, 9 m sweeps and 1.5 t Morgere multipurpose otter boards. Door spread was 60-75 m, mouth opening, 3-4 m vertical and 20-29 m horizontal. The footrope was constructed from rubber discs that slightly raise the footrope from the seabed (Atkinson et al. 2011). The new trawl gear sampled a greater portion of the water column, and has reduced herding and reduced sampling of flatfish and batoids as a result of the modified footropes. Currently, DAFF is in the process of acquiring sufficient comparable annual data sets from the new trawl gear to allow calculation of calibration factors for all fish species against the old gear; however, these data are not yet available for inclusion in this study. The FRS *Africana* did not conduct surveys during 1989, 2000, and 2001 due to technical problems. The Norwegian RV *Dr. Fridtjof Nansen* conducted the 2000 and 2001 surveys, but data from those years were excluded from the analysis due to the incompatibility with other years.

The research surveys cover an area of approximately 32,000 square nautical miles on the west coast of South Africa's continental shelf up to the 500 m isobath (Fig. 1). For survey purposes, the continental shelf is stratified into five 100 m depth zones with the random stratified survey design aiming to obtain homogeneous station density and sampling across the shelf. The entire catch of almost all trawls (99%) was sorted to the lowest practical taxon. Trawls yielding >4 t (1%) were subsampled with about 50% of the catch fully processed and the remainder screened for large individual fish or rare species and then weighed. A target of at least 100 trawls was set for each survey.

In all cases, catch biomass data (in kilograms) were standardized to 30 minute tows. An assumed constant towing speed of 3.5 knots and average mouth width of 26 m are used to calculate the swept area of a 30 minute tow as 0.0246 square nautical miles using the following equation:

$$\text{Swept area} = (\text{speed} \times \text{duration}/60) \times (\text{mouth width}/1852).$$

The catch biomass (in kilograms) was standardized by dividing by the swept area to obtain a mean density of fish with units of kilograms per square nautical mile (kg/nm^2). For each species, standardized data in kg/nm^2, referred to hereafter as fish density, were analyzed using the method developed by Rodionov (2004), the Sequential T-test Algorithm for Regime Shifts (STARS). This uses sequential analysis over long time-series specifically to detect regime shifts.

Demersal trawls do not adequately sample small pelagic, mesopelagic, and mid-water fish species; therefore those species were omitted from this study. Subsequently, all remaining demersal fish species occurring in more than 10% of years surveyed (i.e., species occurring in more than two annual surveys) were selected for STARS analysis. This resulted in 97 (56%) demersal species (variables) being analyzed for regime shifts.

The STARS algorithm tests each new observation (data point) for a significant difference from the mean of the current regime (as calculated from previous observations) under the statistical criteria of a Student's *t*-test. If the current value is found to be greater or less than the critical value of the current regime mean, then the value (year) is marked as a possible change point. Subsequent observations are similarly tested to confirm this change point as a new regime or simply an outlier. The algorithm relies on sequential testing of each subsequent data point against the previous data; thus shifts detected at the end of a time series require subsequent data to test their robustness as a true shift. Similarly, the algorithm is unlikely to detect shifts during the early years of a time series due to the limited previous observations. The testing procedure calculates a Regime Shift Index (RSI) which represents a cumulative sum of normalized anomalies relative to the critical level (Rodionov 2004). A detailed description of the STARS algorithm is provided by Rodionov (2004) and summarized by Howard et al. (2007).

The cut-off length (l, years) determines the minimum length of regimes for which the magnitude of the shifts remains intact (Rodionov 2004). Regimes that are longer than the cut-off length will be detected, but the probability of detecting regimes shorter than the cut-off length decreases proportionally to their length. Regimes shorter than the cut-off length may still be detected if the shift is sufficient in magnitude (Rodionov 2004). Cut-off lengths of 5 and 7 years, as indicated by cluster analysis and multidimensional scaling (MDS) results (Atkinson et al. 2011), were considered appropriate for testing of the demersal fish data in this study.

The significance level is the maximum level at which a new regime is recognized from shifts in the mean (Rodionov 2004). The lower the significance level, the larger the magnitude of the shift required in order to be detected. Howard et al. (2007) investigated shifts at significance levels of 5% ($p = 0.05$) and 10% ($p = 0.1$) and concluded that the 10% level was more appropriate to compensate for the large interannual variability displayed by the noisy time-series. Similarly, significance levels of 5% and 10% were investigated in this study.

Outliers in the data may prevent the average from representing the mean value of a regime and may significantly affect the results of the regime shift detection (Huber 2005). To account for outliers, a Huber's parameter is applied; this controls the weight assigned to outliers and thus the magnitude of the average values of each regime (Rodionov 2004). The influence of Huber's parameter values of $H = 1$, 3, and 6 tested in this study yielded negligible difference among results at the cut-off length of 5 years and 10% significance. Results are reported for a Huber's parameter value of 1 only.

Time series data frequently show serial correlation (red noise); however, STARS assumes that there is no autocorrelation (Howard et al. 2007). Prior to STARS analyses, serial correlation can be removed using a "prewhitening" method. The inverse proportionality with four corrections (IP4) prewhitening method involves subsampling and bias correction of the least-squares estimate of the serial correlation (Rodionov 2006) and was selected for use in this study. Shifts that are detected in a prewhitened time series are smaller in magnitude than those detected without prewhitening, under "straight" analysis (Rodionov 2006). Following Howard et al. (2007), all variables (species) in this study were analyzed both straight (without prewhitening) and with prewhitening under all model parameters described above (i.e., cut-off length, significance level, and Huber's parameter). Prewhitened results can be considered to be more robust in representing a regime shift.

If STARS analysis detected a shift (either straight or prewhitened) in more than 50% of model settings (cut-off lengths, significance levels, Huber's parameter, straight, and prewhitened) for a species, the shift was considered robust and likely to reflect a true long-term shift for the purposes of this study.

Results

Species population shifts were detected in a total of 37 (40%) demersal fish species examined for regime shifts, of which 25 (27%) occurred in ≥50% of model settings and were considered true population shifts in this study (Table 1, Fig. 2). More than half (52%) of these species show population shifts occurring toward the latter part of the time series (i.e., from 2002 to 2009); however, many of these species' shifts (46%) occur at the very end of the time series (in 2008/2009, shaded in gray in Table 1). Successive annual data are considered necessary to validate shifts detected at the end of a time series. Ten species (40%) show shifts in both early (1992 to 1997) and later (2002 to 2009) years, whereas a negative shift in early years only was detected in just one species (monkfish, *Lophius vomerinus*). One species (west coast sole, *Austroglossus microlepis*) showed population shifts in 1999 and 2009, thus not fitting into any of the previous categories described (Table 1). Twenty-four percent of species show population shifts that correlate to an increase in density in the early years (1992 to 1997) followed by a negative shift in more recent years (2002 to 2009).

Table 1. Classification of demersal fish species in which true population shifts (≥50% of model settings) were detected using STARS analysis. Species shaded in gray show shifts detected at the end of the time series only. Symbols in parentheses indicate the year in which either positive (+) or negative (–) shifts were detected.

Population shifts detected in 1992-1997 and 2002-2009	**Population shifts detected between 2002 and 2009**	**Population shift detected in early years only (1997)**	**Population shifts detected in 1999 and 2009**
Lepidopus caudatus, ribbonfish (–1992) (+2007, 2008)	*Notacanthus sexspinis*, spiny eel (+2002, 2004, 2006) (–2008)	*Lophius vomerinus*, monkfish (+1997)	*Austroglossus microlepis*, west coast sole (–1999) (+2009)
Chelidonichthys queketti, lesser gurnard (+1993) (–2003)	*Sufflogobius bibarbatus*, pelagic goby (+2005, 2007)		
Congiopodus spinifer, smooth horsefish (+1994) (–2003)	*Paracallionymus costatus*, ladder dragonet (+2004, 2007)		
Holohalaelurus regani, izak shyshark (+1994, 1995) (–2007)	*Galeorhinus galeus*, soupfin shark (–2007, 2008, 2009)		
Squalus megalops, bluntnose spiny dogshark (+1995) (–2007, 2008)	*Scombrops boops*, gnomefish (+2006)		
Raja sp., skates (+1995) (–2004, 2007)	*Brama brama*, angelfish (+2006, 2007)		
Zeus capensis, John dory (+1997) (–2005) (+2009)	*Torpedo nobiliana*, Atlantic electric ray (+2007, 2009)		
Genypterus capensis, kingklip (+1992, 2006, 2009)	*Hoplostethus mediterraneus*, Mediterranean roughy (+2008)		
Scyliorhinus capensis, yellow spotted catshark (+1997, 2009)	*Hexanchus griseus*, bluntnosed six-gill shark (+2009)		
Hydrolagus africanus, African chimaera (+1995, 1996, 2009)	*Tripterophycis gilchristi*, Gilchrist's triplefin (+2009)		
	Scomberesox saurus, Atlantic saury (+2009)		
	Synogrops japonicus, Japanese splitfin (+2009)		
	Metelectrona ventralis, lanternfish (+2008)		

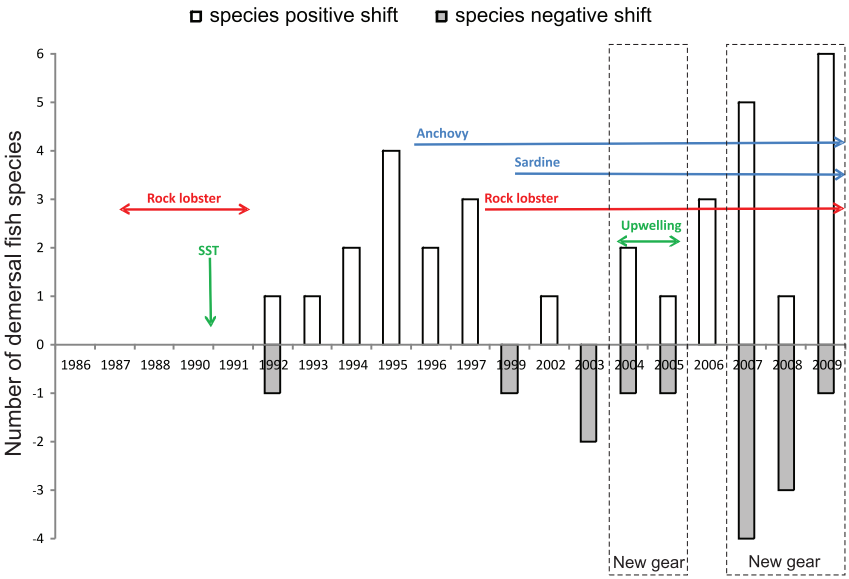

Figure 2. Number of species detected to show true population shifts (>50% of model settings) between 1986 and 2009 using STARS analysis. Shifts detected in two environmental forcing variables, namely sea surface temperature (SST) and upwelling anomalies (from Howard et al. 2007), and anchovy and sardine small pelagic species (from van der Lingen et al. 2006) and rock lobster (from Cockcroft et al. 2008) eastward shifts are temporally represented. Data enclosed in dashed boxes represent years when "new" trawl gear was used.

All but two species (monkfish and west coast sole) in which true population shifts were detected can be temporally classified into two types of shifts. The first temporal shift occurred in the mid 1990s, represented by an average increase in population size of some species. The second temporal change occurred during the mid 2000s as a result of increases in some species but also declines in several others. An overview of the number of species showing positive and negative shifts during the time period analyzed is presented in Fig. 2.

Long-term shifts of representative species from the first two types of shifts described above and listed in Table 1 are illustrated in Fig. 3. The fast growing spiny eel, *Notacanthus sexspinis*, shows an initial population increase in 2002 but in 2008 a decline in the population was detected (Fig. 3A). The long-lived soupfin shark, *Galeorhinus galeus*, shows a robust decline toward the end of the time series (Fig. 3D, 2007 to 2009), whereas the Japanese splitfin, *Synogrops japonicus*, shows an increase at the end of the time series (Fig. 3B). A fast-growing ribbonfish species, *Lepidopus caudatus*, indicates suppressed population densities until apparent increases in 2007 (Fig. 3C), whereas the slow-growing, longer-lived izak shyshark, *Holohalalurus regani*, shows a decline in abundance in 2007 (Fig. 3F). The commercially harvested kingklip, *Genypterus capensis*, shows a sustained increase in population density on the west coast of South Africa since 1992 (Fig. 3E). These species are used to broadly represent the types of shifts detected for other species in this analysis.

Discussion

Investigating the demersal fish research database for regime shifts over the past 24 years using STARS analyses indicates the occurrence of two periods during which population shifts of many species were detected. The first shift, occurring in the mid 1990s, is associated with an overall increase in the density of many species, whereas the second shift detected during the mid 2000s reflects decreases of many species (Fig. 2). Changes in the demersal fish populations observed in the southern Benguela region in this study temporally follow the regime shifts detected in environmental forcing variables on the west coast (SST = 1991 and upwelling anomalies = 2004/2005, Howard et al. 2007) and the eastward spatial shifts observed in rock lobster populations in the late 1980s, early 1990s, and at the turn of the century (Cockcroft et al. 2008, Fig. 2). The delayed response in the demersal community may be a result of a lag effect subsequent to shifts in forcing environmental variables and rock lobster populations. Persistent change in relative abundance of small pelagic species in the mid and late 1990s (Roy et al. 2007, Coetzee et al. 2008) may have contributed to the subsequent changes detected in the demersal fish species on the west coast during the mid 2000s (Fig. 2). Many demersal species prey on small pelagic fish (Hislop 1996); their reduced availability on the west coast of South Africa, as a consequence of their change in relative abundance, may have contributed to the negative population densities in some demersal species during the mid 2000s.

Six species reflecting population shifts during the early to mid 1990s and in the mid 2000s are likely to have *K*-selected life-history strategies (based on information available at the genus or family level only), these being yellowspotted catshark, *Scyliorhinus capensis*; African chimaera, *Hydrolagus africanus*; spiny horsefish, *Congiopodus spinifer*; izak shyshark, *Holohalalurus regani*; bluntnose spiny dogshark, *Squalus megalops*; and skates, *Raja* sp. (Jennings et al. 1999, Stevens et al. 2000). The latter four (spiny horsefish, izak shyshark, bluntnose spiny

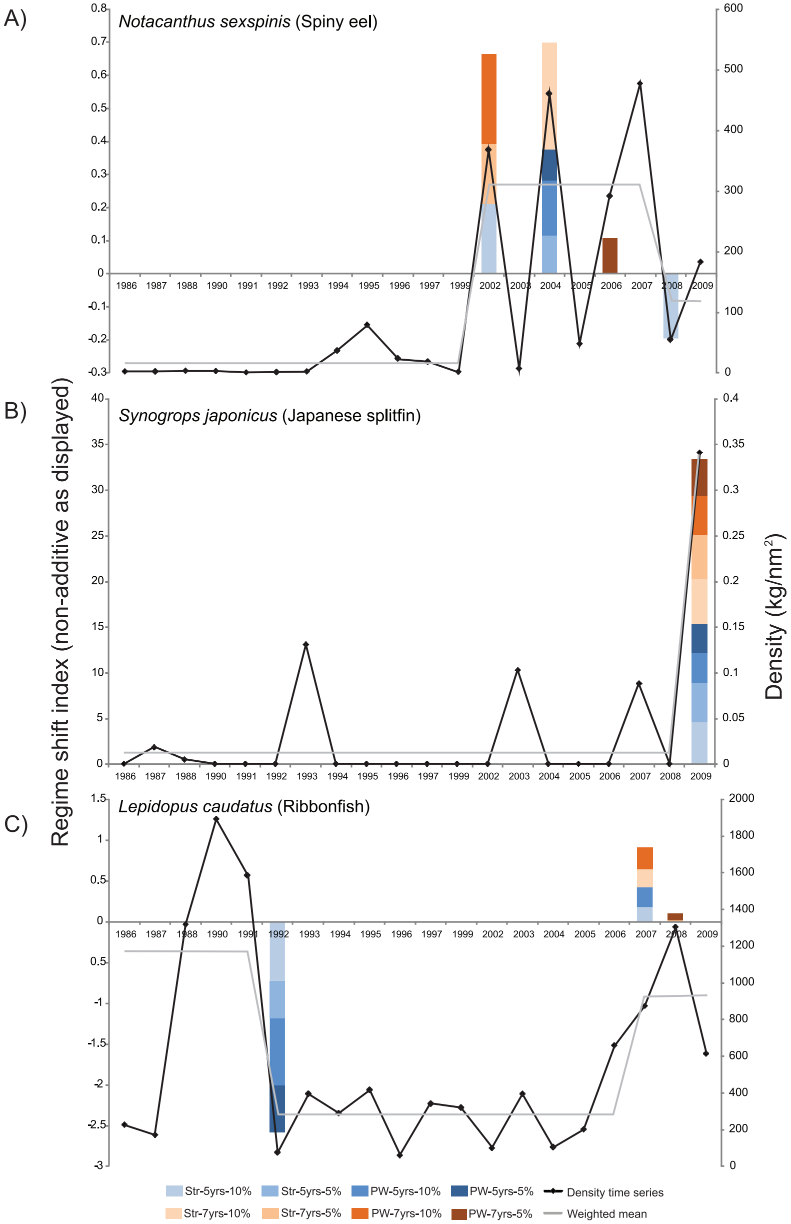

Figure 3. **Results of STARS analysis showing magnitudes of regime shift indices for different model runs (non-additive stacked vertical bars, left axis) occurring in representative species. (A) *Notacanthus sexspinis* (spiny eel) shifts in later years (2002-2009). (B) *Synogrops japonicus* (Japanese splitfin) shifts at the end of the time series. (C) *Lepidopus caudatus* (ribbonfish) shifts in early (1992-1997) and later (2002-2009) years.**

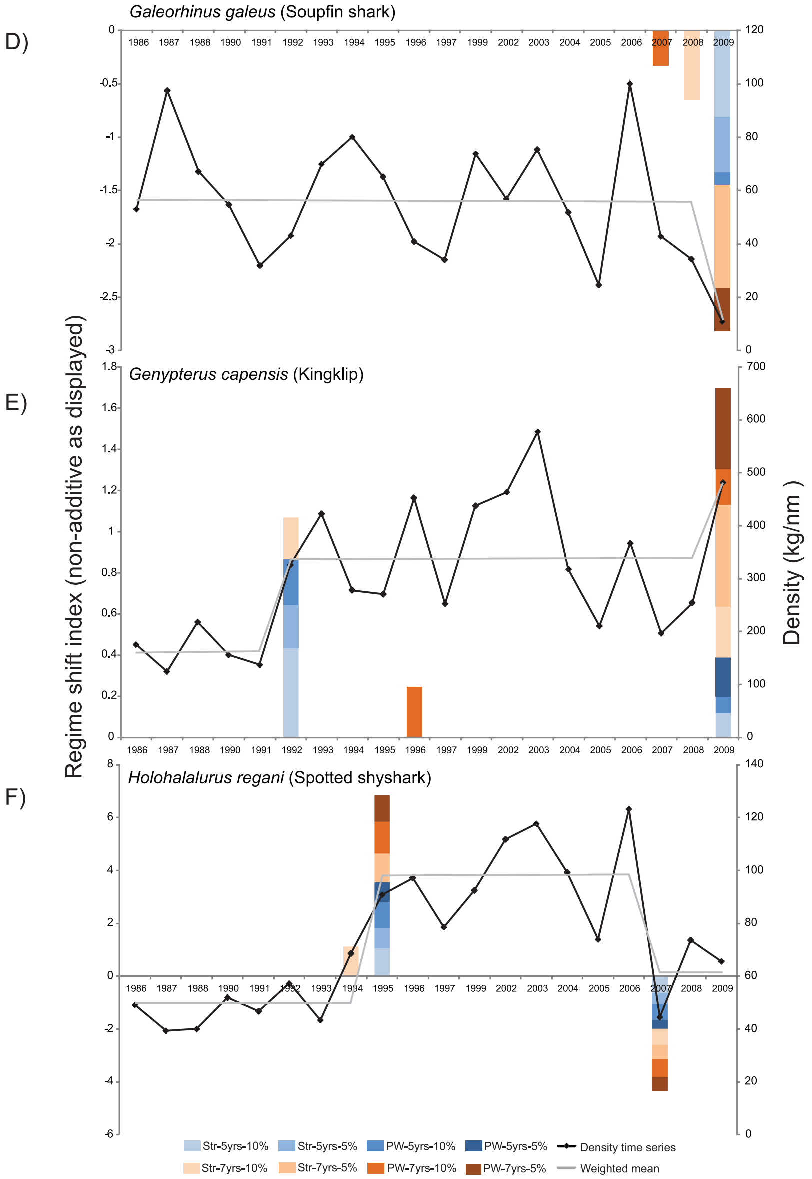

Figure 3 (continued). (D) *Galeorhinus galeus* (soupfin shark) shifts in later years (2007-2009). (E) *Genypterus capensis* (kingklip) increasing shifts in 1992, 1996, and 2009. (F) *Holohalalurus regani* (spotted shyshark) shifts in early (1992-1997) and later (2002-2009) years. Str = straight, PW = pre-whitened, 10% = 0.1 significance, 5% = 0.05 significance, 5 yrs and 7 yrs refer to respective cut-off length model settings. Right axis = fish density and weighted mean.

dogshark, and skates, Table 1) show an initial positive shift, followed by a negative, overall decreasing shift in more recent years, a pattern predicted for *K*-selected species in a fished system (Pianka 1970). However, four long-lived, slow-growing species (Walmsley et al. 2005, Stevens et al. 2000) increased in recent years: monkfish, *Lophius vomerinus*; Atlantic electric ray, *Torpedo nobiliana*; African chimaera, *Hydrolagus africanus*; and yellowspotted catshark, *Scyliorhinus capensis*. Alternative analyses and further research of life histories is considered necessary to explore the hypothesis of a decrease in late-maturing, slow-growing species within a fished system.

Kingklip, *Genypterus capensis*, and soupfin shark, *Galeorhinus galeus*, two commercially valuable species, show population shifts that are likely to be directly influenced to some extent by targeted fishing activities. A kingklip-directed longline fishery was terminated in 1991 in South Africa as a result of stock collapse and this species is now only landed as bycatch with a controlled maximum bycatch allowance (Griffiths et al. 2004). Furthermore, an area along the south coast, known to be a kingklip aggregating and spawning area, has been seasonally closed to fishing since 2005 to protect spawning stocks (Anon. 2006). Relative abundance shifts for this species show a positive increase in 1992 and again in 2009 (Fig. 3E), which could reflect results of implementation of effective management measures.

Soupfin shark are targeted by the demersal shark longline fishery and declining catch rates have been reported since 2001 (Da Silva and Burgener 2007). The STARS analysis detects negative abundance shifts in 2007, 2008, and 2009 for this species (Fig. 3D), suggesting a decrease in the population, possibly influenced by cumulative fishing pressure, which has remained fairly constant during this period (Atkinson et al. 2011). Large, slow-growing, late-maturing sharks and rays are known to be particularly vulnerable to overfishing due to their *K*-selected life-history characteristics (Stevens et al. 2000). Stevens et al. (2000) suggest that fluctuation in shark populations may indicate community changes induced by fishing pressure and fishery managers should be attentive to such fluctuations.

Confounding the interpretation of changes detected in demersal fish populations in this study are the effects of the change in trawl gear configuration, implemented in 2004. Comprehensive calibration studies are being conducted, but these require sufficient data from both old and new gear annual surveys to calculate calibration factors for each species. Results from these studies are, however, not yet available for application (Dr. R. Leslie, DAFF, pers. comm.). The effect of the new trawl gear can be predicted for many species, e.g., increased water column sampling, less herding, and reduced catches of flatfish and batoids. Taking these predicted effects into consideration, several species show changes inconsistent with those predicted as a result of gear change.

As examples, gear changes are not predicted to affect catches of ribbonfish, *Lepidopus caudatus*, or angelfish, *Brama brama*; however, STARS analyses detect positive shifts in both of these species during the mid 2000s. Furthermore, catches of lesser gurnard, *Chelidonichtys queketti*, are predicted to decrease as a result of the new gear; but no shifts (or substantial change in catch rates) are detected after 2004 (Table 1).

Other species show shifts that can almost certainly be linked to the effects of gear change. Population shifts detected in the ladder dragonet, *Paracallionymus costatus*, correlate closely with the introduction of the new trawl gear configuration in 2004 (Table 1). This species appears to reflect inflated densities when the new trawl gear configuration is used and reduced densities when the old configuration was used (prior to 2004 and in 2006), suggesting that the shifts detected for this species are likely an effect of gear differences. The pelagic goby, *Sufflogobius bibarbatus*, is reportedly increasing in abundance in southern Namibian waters (central Benguela region) possibly as a result of proposed regime shifts detected in this region in the late 1980s/early 1990s (Boyer and Hampton 2001). Pelagic goby occur only in the far north of the study area, adjacent to the area studied, by Boyer and Hampton (2001), and it is possible that the positive STARS population shifts detected for the pelagic goby in 2005 and 2007 (Table 1) are a reflection of a southward spread in distribution range due to its increasing abundance. However, the change in trawl gear configuration in 2004 may also have influenced the catch rates of this species, which would be expected to increase as a result of the gear configuration change.

Aside from these species, there is currently limited evidence that the change in trawl gear configuration influences catch rates of demersal species sufficiently to reflect population shifts when applying STARS analyses. Nonetheless changes detected in demersal species should be interpreted taking cognizance of the change in trawl gear during the mid 2000s. Further calibration surveys are planned to facilitate separating gear effects from true demersal population changes (Dr. R. Leslie, DAFF, pers. comm.).

Ecosystem regime shifts have profound implications for marine ecosystems and need to be incorporated into management strategies (e.g., single species stocks managed as part of dynamic ecosystems of interacting species, Rothschild and Shannon 2004). The possibility of regime shifts occurring where important economic marine resources (e.g., small pelagic, rock lobster and demersal species) are affected, implies that management of fishing pressures on these ecosystems should aim toward enhancing stability and resilience in the system and not attempt to prevent natural fluctuations (de Young et al. 2008). Resilience of a system implies the capacity to absorb disturbance and reorganize itself while undergoing change so as to retain similar overall functioning and structure (Folke et al. 2004). The diversity within species and populations and the diversity of functional groups appear to be critical in maintaining ecosystem resilience (Folke et al. 2004). Ecosystem resilience is reduced when key functional groups of species, age groups, or trophic levels are removed from the system. Fishing, pollution, and habitat destruction can result in species reduction or removal. Reduced resilience of ecosystems renders them more vulnerable to changes that could previously be absorbed, thus increasing the likelihood of anthropogenically driven regime shifts (Folke et al. 2004).

Shin et al. (2010) stated that practical implementation of an ecosystem approach to fisheries (EAF) remains a challenge that is yet to be achieved. Development and monitoring of ecological and socioeconomic indicators have been identified as practical ways to support the implementation of EAF by assessing ecosystem status, the impacts of human activities, and the effectiveness of management measures (Cury and Christensen 2005, Jennings 2005). Jarre et al. (2006) propose the use of indicators, derived from observations and models, to be synthesized in a rule-based decision support system that captures and organizes disparate information to interpret and assess the risk of long-term ecosystem change occurring. The global IndiSeas scientific Working Group (created in 2005 under the EurOceans Network of Excellence, and endorsed by UNESCO-IOC) uses ecological indicators (mostly survey-based) to analyze and compare, inter alia, the impacts of fishing on the structure and functioning of marine ecosystems across the world (Shin et al. 2010). Analyses conducted by this working group suggest that combining and comparing sets of indicators from survey data and catch data help in diagnosing the status of exploited populations and ecosystems (Shin et al. 2010). Jouffre et al. (2010) assessed the feasibility of long-term scientific research survey data, originally designed for stock assessment and management purposes, for use in estimating EAF indicators. Jouffre et al. (2010) concluded that such long-term series are inescapable for EAF management since the past cannot be resampled, and that EAF objectives enhance the need for continuing such monitoring with new insights. South African long-term demersal trawl survey data analyzed in this study show good potential for further development of indicators using STARS analyses, specific to the southern Benguela region, for use in contributing toward an EAF management.

Ecosystem regime shifts present a challenge for fisheries management. Successful management under potential regime shift conditions requires a flexible strategy to rapidly adapt to changing conditions. Maintaining (or rebuilding) resilient ecosystem states in conjunction with adaptive, flexible management protocols is considered to be an effective way of implementing an ecosystem approach to fisheries management (Cury and Shannon 2004, de Young et al. 2008). This study has shown the applicability of the STARS analysis technique as a simple, yet efficient, tool with which to identify temporal changes in demersal fish populations within the southern Benguela ecosystem. Furthermore, the results show that changes detected using STARS analysis correspond temporally with assemblage changes detected in the same demersal community using multivariate analyses (Atkinson et al. 2011), thereby strengthening confidence in these results. This study is the first to show changes in demersal fish populations that may be effects of earlier regime shifts detected in environmental forcing variables in the southern Benguela ecosystem (Howard et al. 2007).

Acknowledgments

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Exploring Climate and Fishing Impacts in an Ecosystem Model of the Strait of Georgia, British Columbia

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Abstract

Building an ecosystem-based approach to fisheries management requires knowledge of how climate and fishing induce changes in fish community structure over short and long time periods. It is recognized that investigating the internal structure of marine production systems, particularly in the form of species interactions, is as important as taking into account external factors such as environmental conditions and fishing activity. In this study, we used an individual-based spatially and temporally explicit multispecies model (OSMOSE) to explore the potential impacts of climate change and fishing on the dynamics of fish populations in the Strait of Georgia, Canada. In the OSMOSE model, the fate of all individuals of multiple fish species was modeled through their life cycles including changes in their spatial distribution, natural mortality, predation, starvation, growth, fishing mortality, and reproduction. Our simulations suggested research should consider the pathways through which environmental disturbances enter the ecosystem and interact with predator-prey dynamics and species life history in order to understand species' responses to environmental changes and management actions. As an example, in the simulations Pacific herring was more sensitive to changes in copepod biomass than changes in phytoplankton biomass, and intensive fishing on Pacific herring decreased the overall fish production from the ecosystem. This study demonstrates the importance of using a model such as OSMOSE to explore scenarios that combine species interactions, fisheries management, and climate change.

Introduction

Commercial fisheries directly affect the dynamics of targeted species, their predator and prey species, and all other species in the ecosystem that have indirect connections through the food web. Interannual and interdecadal climatic variations also strongly affect the dynamics of each species as well as the structure and function of marine ecosystems (McGowan et al. 1998). Thus, it becomes imperative to develop tools to understand how harvested populations and entire ecosystems respond to climate variations (McGowan et al. 1998). In all, the combined effects of ecological processes through trophic interactions, environmental disturbances, and fishing need to be understood to help move toward ecosystem-based approaches to fisheries management. This understanding has been hampered by the lack of modeling tools that can include ecological considerations (Gamble and Link 2009). By developing models that take account of these ecological processes explicitly, we may be able to move progressively toward ecosystem-based fisheries management (Link 2002, Pikitch et al. 2004).

In this paper we present an individual-based ecosystem model (OSMOSE: Object-oriented Simulator of Marine Ecosystems Exploitation) that was developed to explicitly model species dynamics, species interactions, and the properties of multispecies assemblages (Shin and Cury 2001, 2004). Since its original development, new features have been added that allow OSMOSE to better address various ecological processes temporally and spatially. We applied the new OSMOSE to the Strait of Georgia (SoG) ecosystem in British Columbia, Canada (Fig. 1). The SoG is a semi-enclosed sea covering an area of approximately 6,900 km^2 (Thomson 1981). The SoG ecosystem supports a number of species of current and past commercial interest, including Pacific herring (*Clupea pallasi*), Pacific hake (*Merluccius productus*), spiny dogfish (*Squalus acanthias*), walleye pollock (*Theragra chalcogramma*), Pacific cod (*Gadus macrocephalus*), and lingcod (*Ophiodon elongates*), and

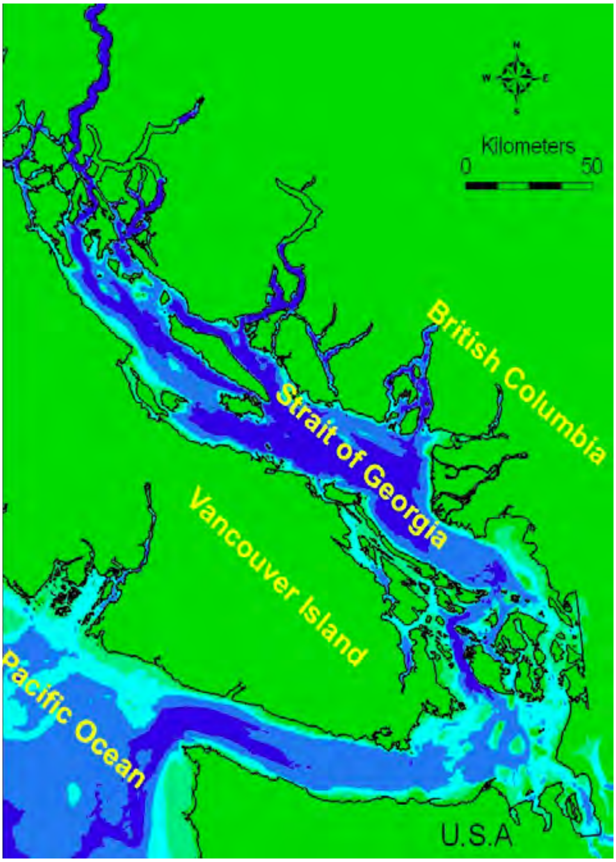

Figure 1. **Map of the Strait of Georgia.**

important non-harvested species such as harbor seal (*Phoca vitulina*). During the last century, changes in the fish community have occurred in response to fishery practices and to climate regime shifts observed in the North Pacific (Hare and Mantua 2000, Beamish et al. 2004).

The objective of our study is to examine how climate change and fishing may affect species in this ecosystem that are connected through the food web. We use OSMOSE as our modeling platform to simulate ecological interactions produced by fishing, climate change, and their interactions. The model does not assume additive interactions between fishing and climate change; rather, through explicit modeling of these two processes in an individual-based setting, the model may result in either synergistic (the combined impact of these two stressors is greater than the simple sum of their individual impacts) or antagonistic (the combined impact of these two stressors is less than the sum of individual impacts) effects (Folt et al. 1999, Breitburg and Riedel 2005). We approximated the climate effect indirectly by imposing changes in phytoplankton or copepod biomass, because changes in phytoplankton abundance have been related to changes in water temperatures (e.g., in the northeast Atlantic; Richardson and Schoeman 2004) and because phytoplankton biomass has been correlated with fish production along the B.C. coast (Ware and Thomson 2005, Perry and Schweigert 2008). We included copepods because they are directly consumed by many fish species. We modeled changes in phytoplankton or copepod biomass over time along with various fishing scenarios in order to understand the potential combined effects of ecological processes acting through trophic interactions, environmental disturbances, and fishing.

Materials and methods

The OSMOSE model

In the individual-based OSMOSE model, all fish within the same year class (or cohort) of a certain species are divided into a number of schools and these fish schools are treated as super-individuals (Shin and Cury 2001, 2004). OSMOSE keeps track of the fate of all fish schools of each species through their life cycle and simulates species interactions through predation in a dynamic and spatially explicit way. The biological processes simulated in OSMOSE include reproduction, growth, spatial movement (including migration), and death due to predation, starvation, and fishing, as well as death due to other uncounted causes represented by additional mortality D. Each species has a relative fecundity (φ, number of eggs spawned per gram of mature female or number of young produced by each female) and age at maturity (A_{mat} in years) as input attributes. The egg production of each species is the product of ϕ and spawning biomass. Recruitment emerges from the annual survival of eggs and juveniles. The average growth of each fish school follows the von Bertalanffy growth model, but for each individual fish school, its growth is adjusted based on the quantity of food ingested during a time step (Shin and Cury 2004). Predation in OSMOSE is a size-based opportunistic process so that trophic interactions are fundamentally dynamic with no pre-established links between species. This assumption may be unrealistic in some cases, e.g., when species do not coexist in the water column, or when for any reason certain species are never encountered in predators' stomach contents. In order to allow deliberate selection/exclusion of certain food items, a simple binary diet suitability matrix is imposed based on crude trophic knowledge, which is still compatible with opportunistic predation. On top of the diet suitability matrix, predation is constrained by a minimal and a maximal predator to prey size ratio, a maximum ingestion rate, and the spatial overlap between predator and prey. A fish school is subject to starvation mortality if the food ration is too low to provide the basic fish maintenance requirements (Shin and Cury 2004). Fishing mortality is assumed to be knife-edged, i.e., all fish schools become vulnerable to fishing when they reach the age of recruitment to the fishery (A_{rec}). The movement of fish schools within the modeled area as well as migration out of the modeled area is represented explicitly in OSMOSE. Spatial distribution of each species at the initial time step is determined by distribution maps provided as input to the model. At each of the subsequent time steps, the spatial distribution is updated in a random fashion but within the grid cells defined by the distribution maps. If fish schools of a migratory species are out of the area at a certain time step, they are subject to average growth and natural mortality (M) only. All processes are updated at each time step, which can be set as one year, half year, or a finer scale of every two weeks, depending on the purpose of the study.

OSMOSE configuration for the Strait of Georgia ecosystem

The number of species that can be included in OSMOSE is conceptually unlimited. Each species follows its own dynamics and can be independent of each other species given its own food and lack of predation interactions with other modeled species. In the SoG OSMOSE model configuration, six currently or historically important commercial fish species are included: Pacific herring, Pacific hake, walleye pollock, spiny dogfish, Pacific cod, and lingcod, which have made up an average of 91% of total commercial catches since 1954. Among these six species, Pacific herring has been dominant in catches (Fig. 2). Pacific herring is not only commercially important but also ecologically significant by serving as a major food source for many species in the SoG including Pacific hake (McFarlane and Beamish 1985), Pacific cod (Westrheim and Foucher 1987), lingcod (Cass et al. 1990), spiny dogfish (Ware and McFarlane 1995), and harbor seal (Olesiuk 1993). In addition to the six commercial fish species, harbor seal is included in the model as a major predator. The harbor seal is the most important marine mammal species in the SoG, consuming an annual average of 27,324 tons of prey since 1999, four times more than what California sea lions (*Zalophus californianus*) and Steller sea lions (*Eumetopias jubatus*) consumed annually in the SoG (Peter Olesiuk, Pacific Biological Station, Nanaimo, 2009, pers. comm.). The diet of the harbor seal is dominated by Pacific herring and hake, composing 75% of its annual consumption (Olesiuk 1993). Spotted ratfish (*Hydrolagus colliei*), a noncommercial species, is included in the SoG model because of its high abundance, at 58-68% of the total

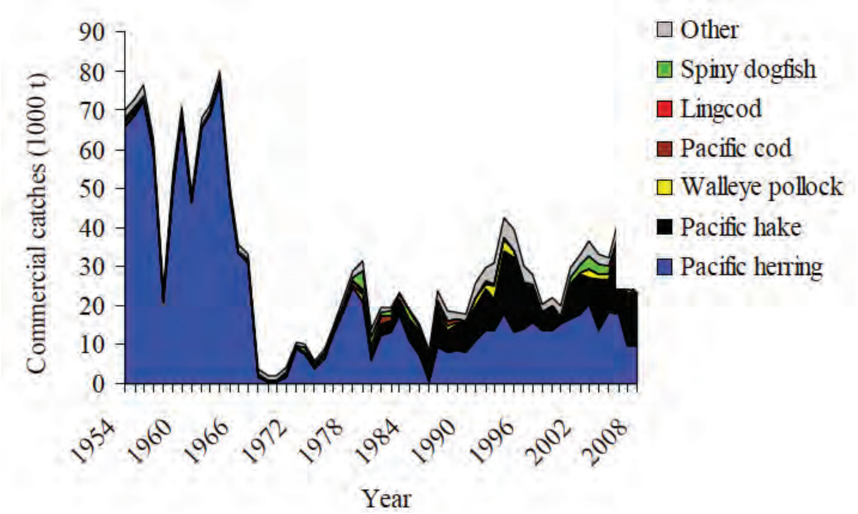

Figure 2. Commercial fish catches in the Strait of Georgia.

biomass obtained during bottom trawl surveys in May and June 2001 in the southern SoG (Palsson et al. 2003). The SoG model also simulates the dynamics of euphausiids, which are a major food source for Pacific herring (Stout et al. 2001), Pacific hake (McFarlane and Beamish 1985), walleye pollock (Shaw and McFarlane 1986), and spiny dogfish (Ware and McFarlane 1995). For benthos, the SoG OSMOSE model simulates the dynamics of pandalid shrimp, which is an important prey for lingcod (Cass et al. 1990), Pacific cod, and walleye pollock (Yang 1993). Except for euphausiids, pandalid shrimp, spiny dogfish, and harbor seal, all species are divided into juvenile and mature categories based on predefined sizes (Pacific herring: 15 cm; Pacific hake and walleye pollock: 30 cm; Pacific cod: 60 cm; lingcod: 65 cm; spotted ratfish: 30 cm) in order to construct the diet suitability matrix. Pacific cod and lingcod juveniles are further separated into two groups because their young-of-the-year (<25 cm) are in locations different from other age classes (e.g., in eelgrass beds, Cass et al. 1990). The dividing size of spiny dogfish is set at 60 cm corresponding to 15 years old to reflect the fact that spiny dogfish younger than 15 form pelagic groups (Beamish and Sweeting 2009). A diet suitability matrix was subsequently developed for the different size groups of the ten species (Table 1).

Table 1. Diet suitability matrix. The Roman numerals assigned to each prey species correspond to the predator species on the first column. Numbers in the predator column and prey row are fish sizes in cm. 1 indicates the predator feeds on the particular prey; 0 means that it does not.

		Prey																			
	Predator	I	II	III <15	III >15	IV <30	IV >30	V <40	V >40	VI <25	VI <60	VI >60	VII <25	VII <65	VII >65	VIII <30	VIII >30	IX <60	IX >60	X	Cope-pods
I	Euphausiid	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
II	Pandalid shrimp	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
III	Pacific herring<15	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
III	Pacific herring>15	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
IV	Pacific hake<30	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
IV	Pacific hake>30	1	0	1	1	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
V	Walleye pollock<40	1	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
V	Walleye pollock>40	1	0	1	1	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0
VI	Pacific cod<25	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
VI	Pacific cod<60	0	1	1	1	0	0	1	1	1	1	0	0	0	0	0	1	0	0	0	0
VI	Pacific cod>60	0	1	1	1	0	0	1	1	1	1	0	0	0	0	0	1	0	0	0	0
VII	Lingcod<25	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
VII	Lingcod<65	0	1	1	1	1	1	1	0	1	1	0	0	0	0	0	1	0	0	0	0
VII	Lingcod>65	0	1	1	1	1	1	1	1	1	1	0	0	0	0	0	1	1	0	0	0
VIII	Spotted ratfish<30	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
VIII	Spotted ratfish>30	1	1	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
IX	Spiny dogfish<60	1	0	1	1	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	1
IX	Spiny dogfish>60	1	1	1	1	1	1	1	1	1	1	0	0	0	0	0	1	1	0	0	1
X	Harbor seal	0	0	1	1	1	1	1	1	1	1	1	0	0	0	0	1	1	1	0	0

The SoG OSMOSE model was initialized for each of the ten simulated species at the biomass levels in 2005 (Table 2) as in Li et al. (2010), who reasoned that more data had become available for the SoG ecosystem since 2005 and the harbor seal population had been stabilized for about one decade by that year. Phytoplankton and copepod biomasses were given as input and set at 15 and 40 ($t \times km^{-2}$) respectively, according to Li et al. (2010). In this OSMOSE model, phytoplankton served as food for euphausiids only. Copepods provided food for juveniles of several fish species as well as Pacific herring adults (Table 1). Biological parameters including those for reproduction, growth, and survival are shown in Table 2. All reproduction and growth parameter values were either taken directly from the literature or calculated indirectly based on information provided in the literature. In a single-species fish model, natural mortality M encompasses death due to all causes other than fishing mortality (F). The SoG OSMOSE model was initialized with no trophic interactions. Each species was subjected to M as calculated based on its maximum age (A_{max}), according to Hoenig (1983):

$$M = -\frac{\ln(\alpha)}{A_{\max}}$$

The parameter α was arbitrarily set at 0.05 in our study such that the resulting M values were similar to the literature for species such as Pacific herring (e.g., Schweigert et al. 2009). The SoG OSMOSE model was then run for 100 years with Pacific herring and hake being fished at $F = 0.2$ and 0.1 $year^{-1}$ respectively, while other species were under no fishing. The SoG OSMOSE model was tuned by adjusting larval mortality and the amount of food until each species reached the equilibrium condition of 2005. Then the species' trophic interactions were activated and the model was tuned again to reach the same equilibrium condition by reducing the initial M (now termed as D) to exclude predation mortality, and by reducing the amount of food to account for the fact that some food is provided by the modeled species.

Except for Pacific herring, all the modeled species are year-round residents in the SoG. In the SoG model, Pacific herring of age 2 or older migrate out of the system in summer and fall. While inside the SoG, spatial distribution maps for each species are required. To obtain these maps, the SoG was divided into 1,300 spatial grid cells each with dimensions of 4×4 km^2. The geo-referenced occurrence data from both commercial fisheries and research surveys were translated into distribution maps. The time step is three months so that reproduction, fishing seasonality, and Pacific herring migration are represented, yet the amount of computation is kept relatively small compared to finer time

Table 2. Biological parameters and initial biomasses for the ten modeled species: growth parameter values for the von Bertalanffy curve including L_∞, k, and t_0, as well as parameters a and b for the weight-at-length function. Relative fecundity φ is the number of eggs spawned per gram of mature female per year, or number of young produced per each female. A_{mat}, A_{max}, A_{rec} are the ages at maturity, maximum longevity, and recruitment to fishery. M is natural mortality when species trophic interactions are not taken into consideration.

Species	Growth					Reproduction		Survivorship			Biomass
	L_∞ (cm)	k (year^{-1})	t_0 (year)	a (g·cm^{-3})	b	φ (eggs·g^{-1})	A_{mat} (year)	A_{max} (year)	A_{rec} (year)	M (year^{-1})	(t·km^{-2})
Euphausiids	1.84	1.68	–0.2	0.009	2.920	24469	1	2	1	1.151	20.0
Pandalid shrimp	2.37	0.92	0.23	0.001	3.069	345	2	4	2	0.576	1.0
Pacific herring	26.3	0.36	–0.03	0.007	3.000	200	3	9	2	0.435	16.597
Pacific hake	44.50	0.46	–0.173	0.001	2.997	108	3	11	2	0.356	4.326
Walleye pollock	44.50	0.92	0.57	0.005	2.441	938	3	8	3	0.489	1.183
Pacific cod	87.50	0.39	–0.01	0.007	3.096	564	3	8	3	0.489	0.070
Lingcod	90.50	0.22	–1.15	0.000	3.217	26	6	17	5	0.186	0.556
Spotted ratfish	79.0	0.20	0.00	0.007	3.013	2 (per ♀)	5	15	3	0.261	2.0
Spiny dogfish	105.52	0.06	–4.279	0.003	3.060	7.2 (per ♀)	35	90	15	0.043	5.710
Harbor seal	150.22	0.63	–1.523	0.011	3.116	1 (per ♀)	4	25	1	0.156	0.287

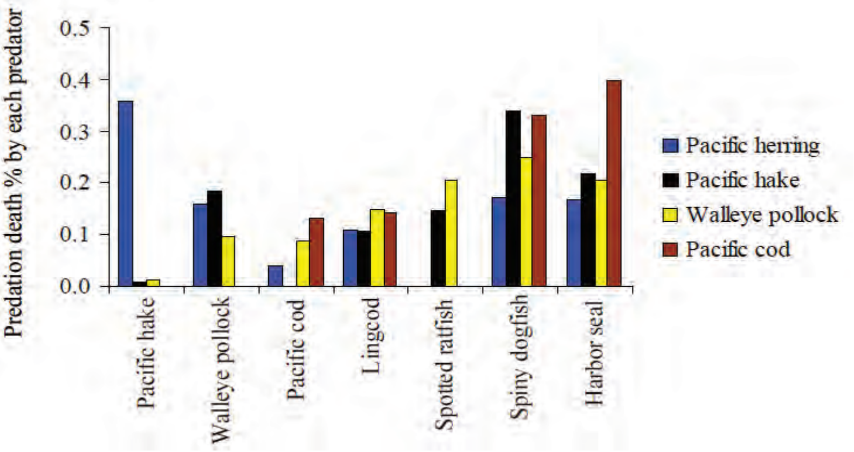

Figure 3. Percentage of abundance loss caused by each of the seven predators. Predators are listed along the x-axis.

steps. Seasonal reproduction was derived from the literature and fishing seasonality was calculated from records of commercial fishing effort.

Simulation scenarios

Fishing directly affects the dynamics of the targeted species, but also indirectly affects other species via trophic interactions. During the last 30 years of the 100-year run, we modified the fishing mortalities while keeping phytoplankton and copepod biomass at the constant level of 2005. Six contrasted fishing scenarios were simulated: $F = 0$ for Pacific herring and hake, and intensive removals of Pacific herring, Pacific hake, spiny dogfish, and harbor seal. The annual F was arbitrarily set at 1.0 to achieve the fishing down effects.

We then allowed phytoplankton and copepod levels to vary during the last 30 years to explore how the plankton changes impact the fish species. Because phytoplankton and zooplankton production changes correlate with water temperature elsewhere (Richardson and Schoeman 2004), their biomass in the SoG may also oscillate in a decadal fashion as has been observed in sea surface temperature (Masson and Cummins 2007). We therefore simulated temporal changes in phytoplankton and copepod biomass in a decadal fashion while assuming spatial homogeneity across all grid cells. Beamish et al. (2004) observed dramatic increases in primary productivity accompanied by doubled euphausiid biomass during cool periods in the early 2000s. The increase of euphau-

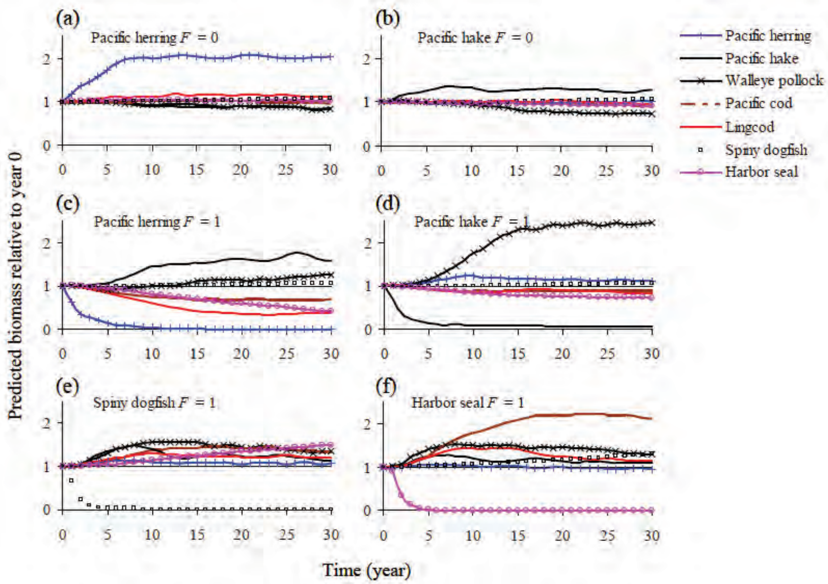

Figure 4. Predicted biomass of main species under constant plankton conditions relative to year 0 prior to the 30-year period: (a) without fishing on Pacific herring; (b) without fishing on Pacific hake; (c) fishing down Pacific herring; (d) fishing down Pacific hake; (e) fishing down spiny dogfish; (f) fishing down harbor seal.

siids was expected to have subsequent impacts on their predators, with the degree of impact varying depending on the trophic levels of these predators. We simulated “high-low-high” scenarios in which phytoplankton, on which euphausiids feed, increased instantly from their original values to higher levels (adding 300 t per grid cell) for ten years followed by an immediate reduction to low levels (removing 300 t per grid cell from the reference level) for a decade before being increased again to higher levels for the last ten years. Under these phytoplankton “high-low-high” scenarios, we tested concomitant fishing scenarios in which $F = M$ (M value for each species is in Table 2) in addition to a scenario with the initial F levels (Pacific herring $F = 0.2$, and Pacific hake $F = 0.1$). Similarly, we simulated copepod “high-low-high” scenarios to see how changes in copepod biomass may affect the fish species. Again, the same fishing scenarios were investigated to examine the combined effects of copepod variations and fishing.

Another series of scenarios consisted of allowing phytoplankton and copepods to change together by gradually increasing or gradually

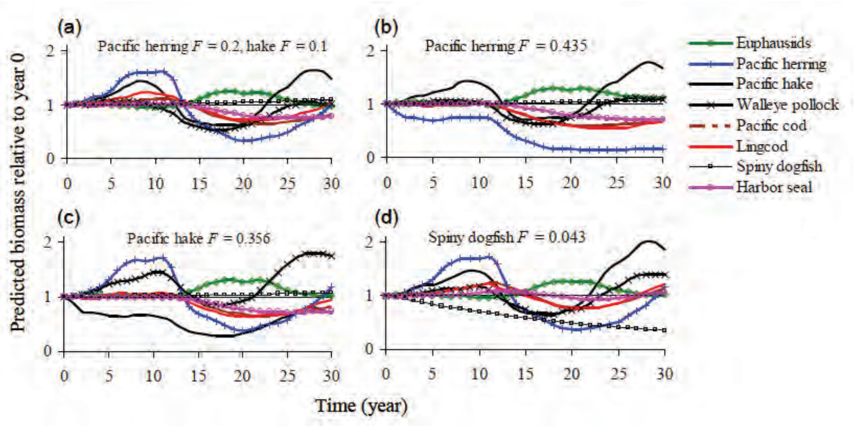

Figure 5. Predicted biomass of main species under decadal changes in phytoplankton biomass (high-low-high) relative to year 0 prior to the 30-year period: (a) at equilibrium condition of 2005; (b) increasing Pacific herring fishing mortality *F* to 0.435; (c) increasing Pacific hake *F* to 0.356; (d) fishing spiny dogfish at *F* = 0.043.

decreasing their biomass over the last 30-year period until they reached the same high or low levels as used in the previous scenarios. We tested the initial F levels (Pacific herring $F = 0.2$, Pacific hake $F = 0.1$) and higher F levels ($F = M$) when phytoplankton and copepods gradually increased. As phytoplankton and copepods gradually decreased, we removed the Pacific herring and hake fisheries, and additionally imposed a fishing mortality on spiny dogfish of $F = M = 0.043$ to explore the effect of releasing predation pressure from spiny dogfish.

Results

Under constant phytoplankton and copepods, all populations remained at equilibrium levels during the last 30 years of the 100-year run. Under this equilibrium condition, Pacific herring was mostly subjected to predation by Pacific hake, while Pacific hake was mostly subjected to predation by spiny dogfish. Both spiny dogfish and harbor seals were important predators of walleye pollock and Pacific cod (Fig. 3). This equilibrium condition was altered when fishing practices changed for any species. When Pacific herring F was reduced from 0.2 to 0 and remained unfished for 30 years, its biomass reached an equilibrium level higher than the reference level (Fig. 4a). Pacific hake and walleye

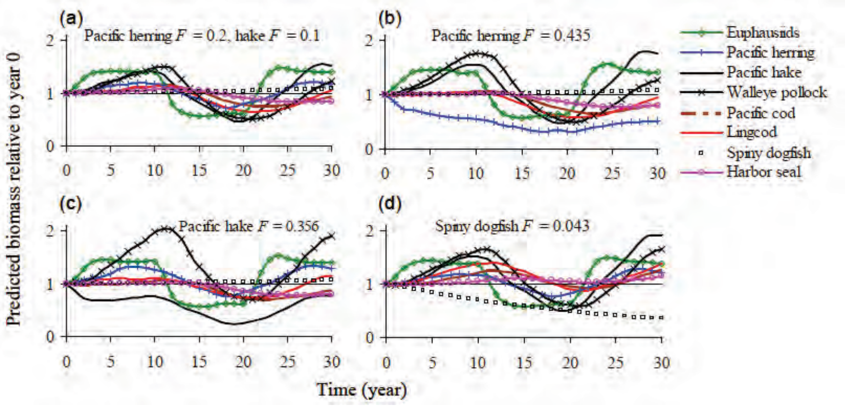

Figure 6. Predicted biomass of main species under decadal changes in copepod biomass (high-low-high) relative to year 0 prior to the 30-year period: (a) at equilibrium condition of 2005; (b) increasing Pacific herring fishing mortality *F* to 0.435; (c) increasing Pacific hake *F* to 0.356; (d) fishing spiny dogfish at *F* = 0.043.

pollock population biomasses decreased due to their competition with Pacific herring for euphausiids. Lingcod appeared to benefit from the Pacific herring increase. When Pacific hake was unfished, its biomass also reached a higher equilibrium, whereas walleye pollock declined slightly because of its competition with Pacific hake for euphausiids (Fig. 4b). When Pacific herring was fished intensively, more euphausiids became available and, as a result, the species that feed on euphausiids such as Pacific hake, walleye pollock, and spiny dogfish increased while other species declined (Fig. 4c). For euphausiids, although predation pressure by Pacific herring was diminished, predation by other predators such as Pacific hake increased. As a result, euphausiid biomass displayed no obvious change (not shown). As Pacific hake was fished down, Pacific herring biomass did not increase dramatically as expected (Fig. 4d). On the contrary, walleye pollock, which has a similar body size and high prey overlap with Pacific hake, increased dramatically to replace Pacific hake. When spiny dogfish was fished down, harbor seal biomass increased gradually due to reduced food competition with spiny dogfish (Fig. 4e); spotted ratfish biomass also increased due to reduced predation (not shown). Other prey species of spiny dogfish increased in biomass to different degrees; however, the change in Pacific herring biomass was minor. When harbor seal was removed from the system, in addition to the biomass increases for Pacific hake, walleye

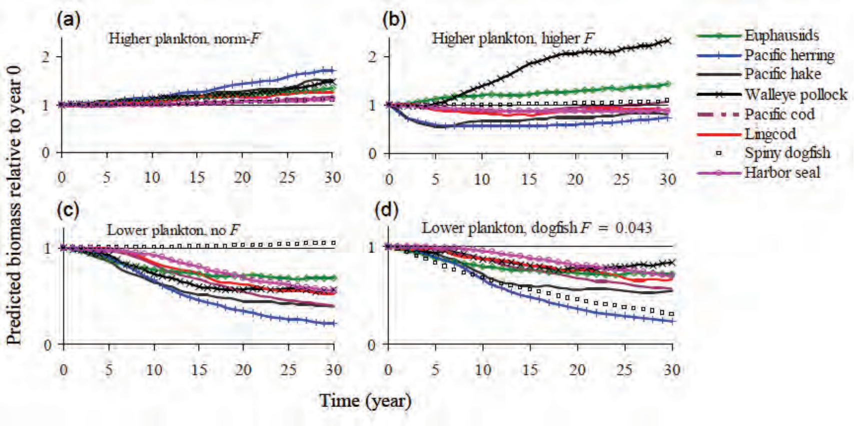

Figure 7. Predicted biomass of main species under gradual increases in phytoplankton and copepod biomass relative to year 0 prior to the 30-year period: (a) Pacific herring *F* = 0.2 and hake *F* = 0.1; (b) Pacific herring *F* = 0.435 and hake *F* = 0.356, predicted biomass of main species under gradual decreases in phytoplankton and copepod biomass; (c) without fishing on any species; (d) only fishing spiny dogfish at *F* = 0.043

pollock, and lingcod, the biomass increase for Pacific cod was most dramatic (Fig. 4f). Again, release from harbor seal predation did not increase the Pacific herring population. When both spiny dogfish and harbor seal were fished down (results not shown), biomasses for Pacific hake, walleye pollock, cod, and lingcod were further increased while Pacific herring biomass was not affected. Although Pacific herring was released from predation by spiny dogfish and harbor seal, predation loss by other predators increased either in number or in weight. The ratio of total biomass (sum of the ten species) in the last year of the 30-year period (year 30) to year 0 (called the system biomass ratio) was 1.0, 0.998, and 1.11, respectively, when harbor seals, spiny dogfish, and both of them were fished intensively. This implies that removing only one of the top predators does not help increase total system biomass, but removing both spiny dogfish and harbor seal appeared to increase system biomass. The system biomass ratios with Pacific hake $F = 0$ and $F = 1$ were similar at 1.029 and 1.006, respectively, indicating some species redundancy between Pacific hake and walleye pollock. On the other hand, the system biomass ratio increased to 1.239 at Pacific herring $F = 0$, but decreased to 0.891 at $F = 1$. This implies that removing forage species such as Pacific herring decreased the overall fish production from this model system.

With changes in phytoplankton biomass in the "high-low-high" scenarios, euphausiids responded quickly by either increasing or decreasing their biomass (Fig. 5a). The changes in euphausiid biomass tended to have greater impact on walleye pollock and Pacific hake than on Pacific herring likely because Pacific herring of age 2 and above spent half the year outside the SoG and thus consumed fewer euphausiids in the system than Pacific hake and walleye pollock. With the rise and fall of euphausiids, each species peaked and dipped at different times as a result of different longevity and predation pressures. Pacific herring biomass peaked the earliest in the ninth year and decreased afterward likely due to increased predation by its predators whose biomasses rose with the euphausiids. On the other hand, Pacific herring biomass started to increase at the end of the low euphausiid regime in the 18th year as Pacific herring predator biomass continued to decline. Spiny dogfish, because of its long life span, did not exhibit biomass oscillations like the relatively short-lived species. Fishing Pacific herring at $F = M = 0.435$ caused its population to decline even during the first decade of high euphausiid biomass and resulted in higher biomasses for Pacific hake and walleye pollock (Fig. 5b). Pacific herring only showed slight increases in biomass during the last decade with high euphausiid biomass. The reduction of Pacific herring biomass had negative impacts on Pacific cod, lingcod, and harbor seal populations (Fig. 5b). With higher F on Pacific hake, walleye pollock became more prevalent (Fig. 5c). Pacific hake was able to sustain a lower biomass level and recover during the last decade of high euphausiid biomass. When spiny dogfish was fished at $F = 0.043$, its biomass steadily declined, which caused harbor seals (Fig. 5d) and spotted ratfish (not shown) to increase. In addition, the biomasses of walleye pollock, Pacific hake, Pacific cod, and lingcod also increased.

In the copepod "high-low-high" scenario, Pacific herring biomass increased more than any other species when copepod biomass rose to high levels, and Pacific herring biomass was reduced more than any other species when copepods were reduced to low levels of biomass, because of its heavier dependence on copepods as food (Fig. 6a). When Pacific herring, Pacific hake, walleye pollock, Pacific cod, and lingcod were at low biomass during the regime of low copepod biomass, euphausiid biomass increased due to reduced predation pressure. During the last decade of high copepod biomass, Pacific hake was able to increase its biomass quickly. When Pacific herring was fished at $F = 0.435$, its population failed to recover even at high copepod biomass during the last decade (Fig. 6b). Pacific cod, lingcod, and harbor seal populations also showed little signs of recovering during the third decade. In contrast, Pacific hake became more abundant as more euphausiids became available to them. Increasing Pacific hake F to 0.356 still allowed Pacific hake biomass to increase when copepods returned to higher levels during the last decade (Fig. 6c). Again, spiny dogfish declined at $F = 0.043$ while other species benefited to different degrees (Fig. 6d).

As the biomasses of phytoplankton and copepods gradually increased, all species gradually increased and Pacific herring appeared to benefit the most (Fig. 7a). Increasing F on Pacific herring and hake caused these two species to be reduced to the same degree during the first few years but Pacific hake was able to gradually increase its biomass likely due to the increased euphausiid biomass (Fig. 7b). Walleye pollock biomass was again able to increase rapidly. When both phytoplankton and copepod biomasses were decreased gradually, even without fishing, all species except spiny dogfish declined, and Pacific herring appeared to suffer the most (Fig. 7c). Imposing F on spiny dogfish was able to improve the populations slightly, but the steady declining trend of Pacific herring biomass was not changed (Fig. 7d).

Discussion

There is no doubt that marine ecosystems during the last century have been under multiple stresses from human exploitation and global climate changes. The SoG ecosystem is no exception. Fisheries have also had an impact on the Strait of Georgia ecosystem; for example, commercial fishing for lingcod in the 1960s and 1970s likely reduced its abundance (Fargo and Tyler 1989). In order to manage marine resources in a sustainable fashion, it is important to understand and predict the effects of fishing and climate change on ecosystems and fish production. Within an ecosystem with various species interacting through predator-prey relationships, the response of one species to the changes in another population can be nonlinear and nonintuitive (Gamble and Link 2009). It has been speculated that the decline of the Pacific herring population during the late 2000s was related to the increase in the harbor seal population in the SoG in addition to declines of recruitment (Schweigert et al. 2009). Assuming constant phytoplankton and copepod biomass, we found that removing harbor seals from the SoG did not increase the Pacific herring population, but caused the biomasses of Pacific herring predators to increase, particularly those of walleye pollock and Pacific cod (Fig. 4f). Even when both spiny dogfish and harbor seals were removed from the system, predation pressure on Pacific herring was not relaxed. At the equilibrium condition, Pacific hake was the most important predator for herring; however, intensive fishing on Pacific hake did not cause Pacific herring to reach higher population levels (Fig. 4d) because Pacific hake was replaced by another predator (walleye pollock). In contrast, removing forage species such as Pacific herring decreased the overall fish production from this model system. These simulations support the use of conservative management strategies to maintain a healthy ecosystem even with the variability imposed by climate change.

Aside from fishing impacts, the SoG has also experienced strong variability and changes in environmental conditions. It has warmed by 0.024°C per year since 1970 (Masson and Cummins 2007). The significant potential rise of water temperature during the next few decades predicted by the Intergovernmental Panel on Climate Change will inevitably have an impact on phytoplankton and zooplankton abundance (Richardson and Schoeman 2004). It is important to note, however, that responses of fish populations to temperature may differ among ecosystems because of different species interactions (McFarlane et al. 2001). In our simulations, euphausiids responded quickly with the increase or decrease of phytoplankton biomass, while other species such as Pacific hake and walleye pollock responded more gradually. The harbor seal at the top of the food chain responded even slower, and the long-lived spiny dogfish was not affected by changes in phytoplankton biomass within the 30-year simulation. On the other hand, spiny dogfish was sensitive to even small levels of fishing mortality (0.043), indicating that very conservative management strategies for these long-lived species are appropriate. Compared with Pacific hake and walleye pollock, the impact of phytoplankton changes on Pacific herring was less pronounced. Instead, Pacific herring was more sensitive to changes in copepod biomass because both Pacific herring juveniles and adults feed on copepods. Thus, the effect of temperature on a particular species depended on the ecological pathways. This implies that research should consider the pathways through which environmental disturbances enter the ecosystem and interact with predator-prey dynamics and species life history in order to understand species responses to environmental changes and management actions. Our simulations indicated that imposing higher fishing mortalities on Pacific herring during periods of lower prey (copepod) biomass could cause the population to fail to recover even after a period of high prey biomass. When anticipating adverse environmental conditions, fishery managers should apply a precautionary approach. When facing declines of forage populations such as Pacific herring, conservative harvest strategies should be applied; manipulation of populations of top predators in our simulations was not able to help forage populations to recover.

In addition to the production of phytoplankton and zooplankton modeled in this paper, climate changes can have impacts on other aspects including physiology (e.g., productivity), geographic range, and phenology at population, species, community, and ecosystem levels (Hughes 2000; McCarty 2001; Vasseur and McCann 2005; Cheung et al. 2008, 2011). In this initial study on investigating climate change effects using OSMOSE, we have ignored all potential aspects except phytoplankton and copepod production. We acknowledge that while this sort of simplistic expression of possible climate change effects is common in modeling studies, ignoring other climate change effects can reach conclusions different from the ones portrayed in the paper and that future studies should consider analyses of sensitivity to the exclusion of other mechanisms.

One important change to the original OSMOSE of Shin and Cury (2001, 2004) is that each modeled species except euphausiids is given a certain amount of implicit prey in addition to the prey species in the model. Modeled species feed on modeled prey species first, up to their maximum ingestion rate. When modeled prey species become insufficient, predators shift to their implicit prey item. The implicit prey item helps stabilize the model simulation. We tuned the amount of implicit prey for each species based on empirical results and arbitrary assumptions. For instance, we set the implicit prey for harbor seal at 3.5 t per grid cell, so that the proportion of the implicit prey is within a reasonable range (around 13%).

In conclusion, ecosystem models can help to better understand and quantify how the future may unfold under different scenarios of fishing management combined with scenarios of climate change. The use of simple species interaction rules makes OSMOSE a generic model that can be applied to different ecosystems given biological parameters and distribution maps of each species. The results from OSMOSE simulations can convey informative and practical implications for fisheries management, in particular with the new features of the model that allow the combined effects of species interactions and temporal climate changes to be investigated.

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Ecosystem-Based Fishery Management for the New England Fishery Management Council

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Abstract

National and international legislation provide impetus for implementation of ecosystem-based fisheries management (EBFM) in the northeast region of the United States. EBFM provides a coherent means to reconcile conflicts among fisheries as well as ultimately simplifying the management system. The Scientific and Statistical Committee of the New England Fishery Management Council has proposed an implementation strategy in the council's mandate area, which is focused on the operational aspects of EBFM. Three approaches are considered. The first "incremental approach" outlines how existing fisheries management plans can be modified to address the needs of EBFM. The second "holistic approach" provides a broader "ecosystem basis" for management through employing constraints imposed by overall ecosystem productivity to guide the allocation of species-specific catches. The third "blended" approach employs multispecies models to inform current stock assessment and management. The proposed EBFM implementation strategy starts with the incremental approach, moves through the "blended approach," and is to achieve full implementation of the holistic approach within three to five years. The transition from incremental through blended to holistic approach at first would increase complexity of the management system, but later simplifies it by focusing on whole ecosystem productivity. The current collection of nine fishery management plans would be replaced by two EBFM plans, one for the Gulf of Maine and the other for Georges Bank. The challenges and opportunities afforded by EBFM are discussed as well as potential institutional changes and the next steps of implementation.

Introduction

Over the past two decades, it has become increasingly evident that fisheries management focused on single stocks in isolation of the broader ecosystem has been a factor leading to resource declines and damaged ecosystems with negative repercussions for fishing participants and communities (Gislason et al. 2000). While significant efforts have been made to include ecosystem considerations within single species management (e.g., consideration of bycatch impacts and implications), these have generally been grafted onto existing management plans to address specific issues, which is leading to increasingly complex and often unwieldy management systems. This has motivated efforts to implement ecosystem-based fisheries management (EBFM) to both address the ecosystem implications of fisheries and to be more flexible and adaptive to the ongoing needs of management (Link 2010).

Since the 1990s, the United States has undertaken a number of initiatives and developed legislation in support of EBFM. In its 2004 report (USC 2004), the Commission on Ocean Policy identified the need to understand important relationships among parts of the ecosystem and how these respond to environmental factors. It also highlighted the need to understand the ways in which humans both influence and are affected by changes in the ecosystem. In 2009, the Joint Ocean Commission identified priorities (JOC 2009) including (1) enacting legislation to create incentives for EBFM and (2) reauthorizing an improved Magnuson-Stevens Fishery Conservation and Management Act (MSRA) to rely more strongly on science to guide management actions for the long-term sustainability of U.S. fisheries. The MSRA, the

National Environmental Policy Act (NEPA), the Endangered Species Act (ESA), the Marine Mammal Protection Act (MMPA), and the Coastal Zone Management Act (CZMA) all require resource managers to take account of the impacts of human activities on the ecosystem, as well as the impact of management on fishing participants and communities. Finally, in July 2010, national policy on the stewardship of the ocean, coasts, and Great Lakes was established by a White House Executive Order. This policy identifies nine national priority objectives, the first being the adoption of ecosystem-based management (EBM) and the second the use of coastal and marine spatial planning as a primary tool for implementing EBM. EBM considers the many uses of the ocean beyond fishing (e.g., recreation, telecommunications, oil and mineral exploration) and will require legislative and institutional changes beyond the mandate and ability of the regional fishery councils. Ultimately, EBFM initiatives by the fishery councils will become a key component of regional EBM.

In addition to recognizing the above legislation, an EBFM plan has significant benefits for the New England Fishery Management Council (NEFMC). The NEFMC has authority for nine fishery management plans. Of these, six are single-species plans and the remaining three include multiple species (although interactions among the species are not directly considered). The Northeast Multispecies (Groundfish) Plan covers 13 species (total of 20 stocks) while the Small Mesh Fishery Management Plan includes three hake species. The Skate Fishery Management Plan covers seven species. Adopting EBFM opens the possibility of substantially consolidating the number of fishery management plans administered by the NEFMC while facilitating consideration of important interactions among species and fisheries now under separate management plans. To the extent that fishery interactions and climate change effects are important but not directly taken into account in current management, issues such as the simultaneous rebuilding of stocks and the choice of long term target levels remain in question. Adoption of EBFM would allow these issues to be addressed within an integrated framework.

The NEFMC recognizes the need to engage the Mid-Atlantic Fishery Management Council (MAFMC), the Atlantic States Marine Fisheries Commission (ASMFC), and Canada in their EBFM efforts, which have implications for the management of shared stocks. As well, a number of the states have or are in the process of drafting EBM plans for their coastal waters, and the Northeast Regional Ocean Council has drafted a 2010 work plan to develop measures of ecosystem health and facilitate marine spatial planning. An EBFM planning framework will allow the NEFMC to better engage with these related EBM activities.

In 2008, the New England Fishery Management Council tasked its Scientific and Statistical Committee (SSC) with outlining a strategy to implement EBFM over the next three to five years. This paper discusses the implementation plan proposed by the SSC. The scientific framework for the overall EBFM strategy is outlined by M. Fogarty (pers. comm.). The transition strategy is outlined including the consequences for council institutions and next steps of implementation. The strategy drew extensively from discussion at a 2009 stakeholder workshop, which considered an EBFM implementation framework, international and national examples of EBFM, and the steps toward EBFM (NEFMC 2010). It represents a significant collaborative effort by the NEFMC.

The human dimension of EBFM

A change from single species management to EBFM necessitates a change in how the human dimension to fishing would be considered by the New England Fishery Management Council. Management would continue to manage only human activity, but under EBFM, this would be done against a backdrop of ecological conditions rather than just biotic and abiotic effects on single species or groups of species (Link 2010). Ecosystem-based management acknowledges humans as fundamental parts of the ecosystem. Although social sciences have studied the human dimension of ecosystems, its contribution has generally been limited to understanding stakeholder involvement in decision-making, often neglecting analysis of trade-offs within ecological choices (Endter-Wada et al. 1998, Link 2010).

Socioeconomic studies range from macro level analyses of social, cultural, political, and economic values; behaviors; and trends to micro-level analyses of individual and group attitudes, values, and behaviors. The importance of these analyses lies not only in gauging the acceptance of management or governance approaches but also in accounting for variability in human resource use, projecting future needs or changes, and assessing the vulnerability and resilience of coastal communities.

The shift to EBFM will also require wider participation of stakeholders in decisions involving trade-offs between fisheries. Given that the biological inputs to EBFM will focus on the total production constraints on fishing, the basis for determining the mix of species fished will require broader analysis of social and economic impacts on fishing communities (Link 2010). Likewise, EBFM will widen the scope of community-based co-management (fishers and managers jointly responsible for management) and require a more adaptive and flexible connection between management and fishing activities (Gutierrez et al. 2011, Clay and Olson 2008).

Implementation plan for EBFM

Making the transition to a different management system is understandably daunting given the demands and responsibilities that the NEFMC faces on an ongoing basis (exemplified by 2010 discussions on the Northeast Region Coordinating Council on new operational procedures for acceptable biological catches and annual catch limits). Each element of the management system must be considered with the need to identify:

- Geographic management areas based upon ecosystem processes (herein termed ecosystem production units or EPUs).
- Risks confronting ecosystem components impacted by fishing.
- Conceptual and associated operational objectives (composed of indicators and reference points).
- Management actions to mitigate impacts (specific and cumulative).
- Assessment activities to monitor progress against the objectives.

The transition to EBFM must acknowledge the ongoing requirements of fisheries management while at the same time develop the building blocks for EBFM with full and transparent stakeholder involvement, and consideration of the social values of marine resources.

In its proposal to the NEFMC, the SSC identifies three approaches to EBFM. The first "incremental" approach would gradually build ecosystem considerations into the current nine fishery management plans (FMPs) and represents an "evolutionary" change to the current management system. The second "holistic" approach would consolidate the nine FMPs into two EBFM plans and represents a dramatic and "revolutionary" change to the management system. The third "blended" approach is an interim stage between these two extremes. These approaches are not mutually exclusive but are steps in the transition to EBFM (Fig. 1). Implementation would start with the incremental and blended approaches, by building ecosystem considerations into the current nine FMPs to take into account biological and technical interactions and environmental/climate factors that cut across FMPs within EPUs. The transition would be complete with adoption of the holistic approach to EBFM, which manages fishing impacts on stocks within each EPU. The transition from incremental through blended to holistic would at first add management complexity but later, by focusing on ecosystem productivity as a whole, lead to its simplification. The strategy also recognizes that development of EBFM must be flexible and respond to eventualities as they arise.

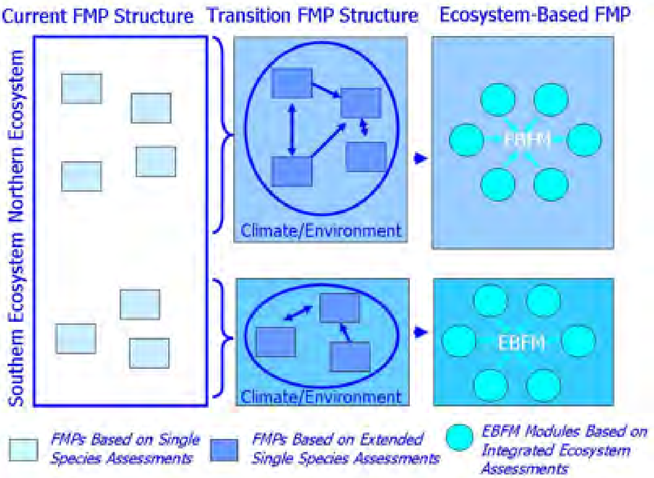


Figure 1. Illustration of ecosystem-based fishery management (EBFM) development from current approach in which separate fishery management plans (FMPs) are implemented for individual stocks (left panel), through incremental/blended approach (middle panel), which retain FMPs but begin to take into account biological/technical interactions and environmental factors across FMPs within defined ecosystem production units (EPUs) to a holistic approach (right panel), which manages stocks within an EPU (M. Fogarty pers. comm.).

Incremental approach

This approach retains existing scientific and management frameworks but either adds to or expands upon them. Incremental does not imply straightforward as the associated data, understanding, and management will often be complex. Examples of the incremental approach are either procedural or scientific: procedurally, omnibus amendments can modify two or more FMPs simultaneously; scientifically, ecosystem influences can be incorporated into single-species stock assessment and management.

Regarding omnibus amendments, the MSRA (like the former Magnuson-Stevens Act) requires that councils summarize effects of fishing on habitats, conclude whether and how fishing adversely affects Essential Fish Habitat, and take steps to mitigate these impacts. The NEFMC is meeting these requirements through an "omnibus" habitat amendment prepared by its Habitat Committee, which has developed the Swept Area Seabed Impact (SASI) model, an analytical tool for evaluating impacts and developing management alternatives. SASI can provide metrics by which management options are assessed against thresholds or targets. Whereas each stock has a unique assessment model to develop harvest strategies, and even different currencies for determining stock status (e.g., absolute versus index-based biological reference points), SASI can provide a common currency for management of habitat impacts of all fisheries. It takes account of the intensity and type of fishing, as well as the type, vulnerability, and recovery rates of habitat and provides an index of habitat alteration for different spatial configurations of fishing activity. If appropriate, steps required to mitigate habitat impacts by a particular fishery can then be amended in the appropriate FMP.

The incremental approach can be used to address other aspects of EBFM without fundamentally altering existing FMPs. Changes in the abundance of prey will affect the productivity and harvest potential of predator stocks, and vice versa. The nature of these predator-prey relationships and resulting management changes could be captured in FMP amendments that, although not applied to all FMPs, would be held in common to those with interacting species. The incremental incorporation of ecosystem factors into single-species stock assessment has already been effectively implemented for fishery management. For instance, large-scale environmental influences have been included in the estimation of expected productivity (e.g., maximum sustainable yield of Pacific sardine, Jacobson and MacCall 1995). Single species stock assessment models have been extended to account for predator-prey dynamics (e.g., multispecies virtual population analysis of Atlantic menhaden; Garrison et al. 2010). These methods for including ecosystem processes in fishery management can be effectively implemented within the existing FMP structure and do not require an omnibus approach.

Another example of the incremental approach involves stock structure. Fish stocks in New England are defined by broad spatial areas (e.g., the northern and southern monkfish stocks) and characterized primarily by gross biomass in modeling and setting management targets. An ever-growing body of knowledge suggests that the coarseness of broad spatial definitions and gross biomass as a stock status indicator might be oversimplifications that hinder stock status assessment and both single-species and ecosystem-based fisheries management (e.g., Cadrin and Secor 2009). Sub-stocks exist within most, if not all, large stock units and play key roles in driving stock dynamics. Connectivity among these sub-stocks provides a buffer against localized depletion through recruitment subsidy and recolonization. When sub-stocks are lost, the larger metapopulation can suffer. Preserving a diversity of local adaptations also creates a larger reserve of genetic options for coping with ecosystem changes. Models accounting for spatial structure are necessarily complex, but their development is progressing (Cadrin and Secor

2009). Finally, fish of different sizes play very different roles as prey, predators, competitors, and even symbionts. Having a more diverse population structure allows more of these roles to be filled, further affecting their ecosystem function, and providing greater responsiveness to ecosystem change.

Holistic approach

Although the incremental approach can effectively incorporate ecosystem considerations in fishery management, there are data limitations and expanding complexities that will preclude incorporation of all ecosystem components. Layering more and more extensions on current FMPs will lead to an overly complex and data-hungry management system. The ultimate move to the holistic approach would address this problem. M. Fogarty (pers. comm.) fully describes the SSC's holistic approach to EBFM, so only an overview of its elements is provided here. Some elements build upon existing initiatives while others are new. In all cases, the transition period is seen as an opportunity to learn by doing and allow flexibility to adapt as experience grows.

Ecosystem-based fisheries management is inherently place-based, identifies the need to consider the interaction among system components in management, and highlights the ways in which human communities both influence and are affected by changes in the ecosystem (M. Fogarty pers. comm.). Thus, a fundamental element of EBFM under the holistic approach is the specification of management units based upon spatially defined ecosystems. As pointed out by M. Fogarty (pers. comm.), the production of individual species/stocks is a function of growth, mortality, and recruitment, but the production of ecological regions depends upon biological processes at the base of the food web, which also underlie production at the species/stock level. They emphasize that the common currencies of space and production serve as a bridge between the current management system and EBFM based on ecologically defined spatial units. M. Fogarty (pers. comm.) defines four northeast regional ecosystem production units (EPUs) based on physiographic, oceanographic, and ecological features:

- Western-Central Gulf of Maine (GOM)
- Eastern Gulf of Maine–Scotian Shelf (SS)
- Georges Bank–Nantucket Shoals (GB)
- Middle-Atlantic Bight (MAB)

Three of these EPUs fall within U.S. waters and one (Eastern Gulf of Maine–Scotian Shelf) is primarily in Canadian waters. Two EPUs (Western-Central Gulf of Maine and Georges Bank–Nantucket Shoals) are in the area of responsibility of the NEFMC. If this schema were to be adopted, the NEFMC might have the lead responsibility for two EPUs while the Mid-Atlantic Fishery Management Council might have the lead responsibility for one EPU. Close interaction and coordination among the NEFMC, the MAFMC, and the Atlantic States Marine Fisheries Commission will be crucial. As well, consultation with Canada will be required on the transboundary EPU.

EPUs can provide a new focus of efforts to manage and monitor the cumulative impacts of fishing across fleets and species. As noted above, the NEFMC is required to summarize the effects of fishing on habitats, conclude whether and how fishing adversely affects essential fish habitat, and take steps to mitigate impacts. Under the holistic approach, the SASI model mentioned above can be used to define areas that require special protection within the EPUs and not just within the FMP area, as is the case in the incremental approach. These efforts can be used to determine and manage the cumulative impacts of fishing on the habitat throughout each EPU.

M. Fogarty (pers. comm.) describes efforts to estimate fishery production of these EPUs and associated target ecosystem exploitation rates designed to obtain maximum aggregate harvest from an EPU, while avoiding the collapse of many of its component species. Multispecies fisheries management will still require catch allocation strategies for individual species, because species have substantially different economic values in the marketplace, and differential exploitation and selection patterns must be anticipated. M. Fogarty (pers. comm.) suggests a strategy in which upper and lower biomass limits are established for total catches set by an estimate of systemwide production and a corresponding target ecosystem exploitation rate. They point out that structured decision-making and trade-off resolution are essential in this approach.

Blended approach

While the incremental and holistic approaches are complementary and seen as phases in a broader EBFM implementation process, it is recognized that there will need to be a transition between the two in which elements of both will be simultaneously applied. This is termed the "blended approach."

Single-species catch limits can be constrained by evaluation of multispecies production. This approach would consider the results of multispecies production modeling to modify short-term yield and biomass expectations from single-species stock assessments (e.g., comparing the sum of single-species productivity expectations to multispecies production models; NEFSC 2008). However, it would require data collection and human resources to support simultaneous analyses. Similarly, long-term productivity expectations (e.g., MSY and B_{MSY} reference points) may be modified by multispecies modeling. Conventional approaches to multispecies production modeling express competitive interactions as a function of overlapping carrying capacities, which in turn should consider B_{MSY} as a multispecies concept.

National Standard 1 guidelines require that FMPs identify ecosystem component species for consideration as management units. The performance of single species stock assessment models and harvest control rules can be evaluated using a holistic ecosystem model as the "operating model" in a management strategy evaluation framework. For example, the ATLANTIS model developed for the northeastern U.S. could be used as the "operating model" (i.e., the virtual reality that includes as much as we know about the ecosystem) to evaluate the performance of single-species stock assessments and harvest policies, similar to the way EBFM is implemented in Australia (e.g., Punt et al. 2001).

A blended approach may also have application in managing interactions among FMPs. For example, the NEFMC's Interspecies Committee has been considering cross-FMP issues and the feasibility of managing species groups (e.g., all demersal fishery resources: groundfish, monkfish, skates). Multispecies approaches to managing bycatch and optimizing multiple fishery objectives have been successful in other regions (Sugihara et al. 2009) and can be implemented in New England (e.g., avoiding yellowtail flounder bycatch in the scallop fishery; O'Keefe et al. 2010). These forms of EBFM that consider technical interactions (e.g., bycatch) are narrower in scope than more holistic forms, but can be expanded to include habitat and other ecosystem-related objectives (e.g., Dankel et al. 2009).

Consequences for council processes and procedures

NEFMC processes and procedures ("institutions") need to be redesigned to address the implications of cumulative ecosystem impacts of fishing. Institutional changes required by EBFM depend on the high-level objectives and the form of EBFM that the NEFMC decides to implement. If the NEFMC decides to remain with an incremental or blended approach and not adopt full EBFM in the holistic approach, current FMPs with omnibus amendments (e.g., habitat, forage, and environment) and possibly the addition of fishery ecosystem plans may meet target species and bycatch species objectives. However, a plan-specific approach may not resolve many of the challenges of mixed-stock fishery management, and overall be ineffective in meeting newly developed ecosystem objectives. If the NEFMC decides to proceed to a full holistic EBFM approach, additional institutional changes will be required.

The NEFMC is not alone in its efforts to implement EBFM. Most, if not all, of the regional fisheries management councils are in various stages of implementation of EBFM (Table 1). Additionally, terrestrial management authorities have worked through these conceptual shifts and may provide insights particularly in terms of process. It will be instructive for the NEFMC to stay informed of these efforts.

Implementation of EBFM in the northeast region presents unique challenges and opportunities. For example, FMPs are currently defined by species and stocks. The ecosystem production unit approach advocated under the holistic approach would require realignment of these stocks with EPUs. Some current stock definitions will cross EPU boundaries, and realignment will ideally reflect spatial population structure. Also, interagency coordination will be necessary for EBFM. As noted above, one of the EPUs (Middle Atlantic Bight) is within the jurisdiction of the Mid-Atlantic Fishery Management Council. Dialogue may be required with state agencies to ensure that its EBFM efforts are complementary to those in state waters. Another EPU (Eastern Gulf of Maine–Scotian Shelf) will require dialogue with Canada to ensure that transboundary EBFM efforts are also complementary.

EBFM may require the NEFMC to broaden public input into its EBFM process. Single species management has led to the establishment of constituents with historical interests in particular fisheries, which will heighten difficulties and potential disagreements that may arise in setting objectives and making trade-offs. To accommodate different roles or to expand public input, the NEFMC may have to change some of its consultative processes, ensuring a participatory and transparent governance process.

A number of New England states (e.g., Massachusetts) have made considerable progress of state-based ecosystem-based management plans, and there has been dialogue with the Gulf of Maine Council on the Marine Environment on the needs of EBM in the northeastern U.S (http://www.gulfofmaine.org/default.asp). The EBFM initiative within the NEFMC will well place fisheries in these broader discussions.

Major shifts in management approaches (including the implementation of a number of catch share programs) have required significant changes in the way fishermen and fishing communities operate and relate to the marine environment and to each other. These changes have relied on public participation and involvement. While this creates opportunities (stakeholders are arguably better organized than they have been in the past), managers need to be sensitive to the fact that stakeholders will once again need to be actively involved in the shifts in management approaches that will transform their lives and livelihoods. Given this, transparency and early public involvement will be key. Additionally, the current NEFMC plan development process may be too cumbersome for EBFM plan development, making it difficult to include the full range of expertise needed. Furthermore, the institutional requirements for developing and implementing EBFM are likely to

Table 1. Synopsis of EBFM implementation efforts in U.S. regional fishery management councils.

Council	EBFM implementation efforts
Mid-Atlantic	• Not currently developing FEP or EBFM plan; addressing ecosystem-related issues on a case-by-case basis through its Ecosystems/Ocean Planning Committee.
South Atlantic	• Has developed FEP that evolved from council's Habitat Plan; recently developed amendment to FEP that will protect specific areas of sensitive habitat, deemed Coral Habitat Areas of Particular Concern (under review). • EBFM Committee established to develop and update FEP; consists of council members and scientists. • Although no technical committee or advisory panel specifically assigned to FEP process, there is Habitat and Environmental Protection Advisory Panel.
Gulf of Mexico	• In initial phase of developing FEP. • Has recently established Ecosystems SSC, which has different membership from "regular" SSC, to advise on development of EBFM. • No other council committee or technical committee assigned to this effort.
Caribbean	• Currently not developing FEP or EBFM plan and has not formed any EBFM-associated institutions.
Pacific	• Beginning process to develop recommendations on EBFM plan, which is envisioned to complement, but not replace, council's four existing FMPs. • Ecosystem Plan Development Team and Ecosystem Advisory Subpanel established; subpanel is 11-member multidisciplinary group representing industry, policy, and conservation from states and tribes; Ecosystem Plan Development Team is 13-member group of state, federal, and tribal scientists and policy analysts whose primary responsibility is to provide analyses and recommendation to council on science in support of EBFM and to develop goals, objectives, and policy alternatives for council consideration as Ecosystems FMP takes shape over next few years.
North Pacific	• Has FEP for Aleutian Islands, a policy and planning document; council also participates with 10 federal agencies and four state agencies in Alaska Marine Ecosystem Forum. • Ecosystem Committee established to discuss ecosystem-related initiatives and positions on: (1) defining ecosystem-based management; (2) structure and council role in potential regional ecosystem councils; (3) implications of NOAA strategic plan; (4) draft guidelines for ecosystem-based approaches to management; (5) draft MSA provisions or requirements relative to ecosystem-based management; and (6) coordinating with NOAA and other initiatives on ecosystem-based management. • Technical Aleutian Islands Ecosystem Team assists council staff in updating FEP; team is multidisciplinary federal and state scientists.
Western Pacific	• Has developed FEPs for Hawaii, American Samoa, Mariana Archipelago, U.S. Pacific Remote Islands, and a Pacific Pelagic Fishery Ecosystems Plan. • FEPs developed by regional ecosystem advisory committees from American Samoa, Hawaii, and Mariana Archipelago; each advisory committee includes council members and representatives from federal, state, and local government agencies, businesses, and nongovernment organizations. • Technical analyses may be provided by council staff or plan teams.

EBFM = Ecosystem-based fishery management.
FEP = fishery ecosystem plan.
FMP = fishery management plan.
MSA = Magnuson Stevens Act.
NOAA = National Oceanic and Atmospheric Administration (U.S.).
SSC = Scientific and Statistical Committee.

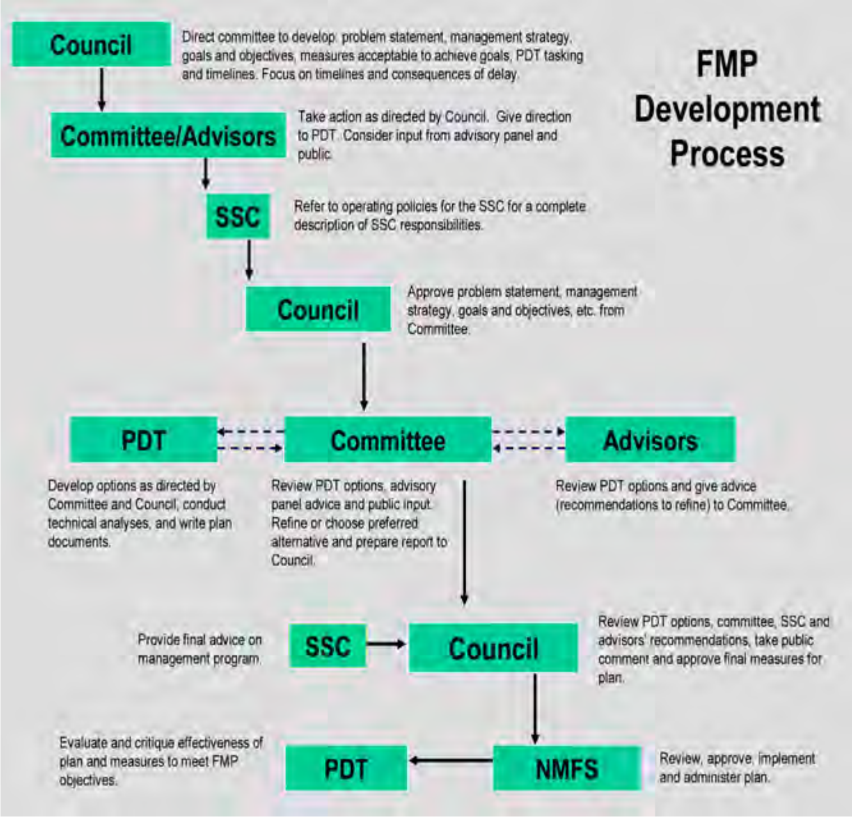
change over time. Recognizing this at the outset of the process should make it easier to implement future needed institutional changes.

Finally, under current national standard guidelines, reference points such as minimum stock size and maximum fishing mortality thresholds must be defined for each stock to the extent possible and each stock must be managed to achieve these reference points within fixed time periods. It will be necessary to establish ecosystem reference points consistent with these guidelines.

Given these challenges, it is useful to consider potential changes to the NEFMC current institutions that may be required to implement EBFM. The current council FMP development process consists of a number of groups (Fig. 2).

The NEFMC uses committees comprised of council members to oversee the development of FMPs or actions taken under each FMP. There are also committees mandated to undertake specific tasks that are common to all FMPs. For instance, the Habitat Committee has been tasked by the NEFMC to undertake EBFM-related tasks. The council, while providing direction to its oversight committees, focuses on approving goals and overall management strategies, and approving specific management options developed by the committees prior to inclusion in any draft or final version of an FMP.

The oversight committees direct the plan development teams to provide analyses of management alternatives, decision support documents, and documents that must be submitted to the National Marine Fisheries Service (NMFS) for review and approval. Plan development teams consist of council staff members, NMFS and state scientists, NMFS plan support personnel, and sometimes academic scientists. The committees receive stakeholder input from advisory panels, which consist of commercial and recreational fishermen, their representatives, and environmental nongovernment organization representatives. Scientific input from outside the plan development team is generally limited to stock assessment advice from the stock assessment workshop and/or the Scientific and Statistical Committee (see below). This includes information on stock status, and may include estimation of biological reference points and catch projections. The SSC may also provide the NEFMC with advice on a range of issues including stock-specific acceptable biological catch, biological and management reference points, peer reviewed biological, economic, or social impact analyses, and so on. Plan development teams work on all aspects of FMP development with the exception of providing stock assessments and acceptable biological catch recommendations. Plan development team members have expertise to meet the procedural requirements and provide analyses under the Magnuson-Stevens Fishery Conservation and Management Act, National Environmental Policy Act, and other statutes. Plan development teams may also contribute scientific work such as providing stock



Direct committee to develop: problem statement, management strategy, goals and objectives, measures acceptable to achieve goals, PDT tasking and timelines. Focus on timelines and consequences of delay.

Take action as directed by Council. Give direction to PDT. Consider input from advisory panel and public.

Refer to operating policies for the SSC for a complete description of SSC responsibilities.

Approve problem statement, management strategy, goals and objectives, etc. from Committee.

Develop options as directed by Committee and Council, conduct technical analyses, and write plan documents.

Review PDT options, advisory panel advice and public input. Refine or choose preferred alternative and prepare report to Council.

Review PDT options and give advice (recommendations to refine) to Committee.

Provide final advice on management program.

Review PDT options, committee, SSC and advisors' recommendations, take public comment and approve final measures for plan.

Evaluate and critique effectiveness of plan and measures to meet FMP objectives.

Review, approve, implement and administer plan.

Figure 2. Institutions of New England Fishery Management Council involved in development of fishery management plans (from NEFMC Operations Handbook: Practices and Policies, revised March 2011).

projections particularly under different management options, analyzing alternative measures in terms of how they might change fishing mortality, analysis of economic and social impacts, and summarizing management impacts on other species, including protected species, and habitat.

The role of the Scientific and Statistical Committee differs from that of all other NEFMC committees. The SSC assists the council in the development, collection, evaluation, and peer review of statistical, biological, economic, social, and other scientific information relevant to the development and amendment of fishery management plans. The SSC also provides ongoing scientific advice for fishery management decisions, including recommendations for acceptable biological catch, preventing overfishing, maximum sustainable yield, and achieving rebuilding targets, and reports on stock status and health, bycatch, habitat status, social and economic impacts of management measures, and the sustainability of fishing practices.

As shown in Fig. 2, the NEFMC initiates an action and directs a committee to oversee its development. If an action includes changing the acceptable biological catch levels, the SSC will provide the council with acceptable biological catch recommendations near the beginning of the development process. Committees are encouraged to listen to input from the advisory panel and provide direction to the plan development teams to develop the appropriate analyses. Committees report back to the council at appropriate intervals and present recommendations for final action. Time permitting, the SSC may provide final advice on management programs before the council makes a final decision on an action. The NEFMC then submits the documents supporting its recommendations to NMFS for review and approval by the Secretary of Commerce.

During the transition to EBFM, the organizational structures required for its effective implementation will need to be designed and the task of modifying current institutions undertaken. Under a holistic approach to EBFM, the focus of planning will become the ecosystem production units, of which there are two under NEFMC jurisdiction (Gulf of Maine and Georges Bank). The development of EBFM plans for each of these will likely require the dedicated efforts of a plan development team for each. During the transition, this may be a task that could be undertaken by an existing council committee (e.g., Habitat Committee) or newly formed EPU committees. It will also be necessary to coordinate the activities of the current plan development teams with the new EPU-based plan development teams. This process will largely dictate the final form of the EBFM planning structures.

No changes in the SSC process are foreseen; its mandate currently addresses the full suite of biological, social, and economic issues confronting the council. On the other hand, there may be more need for the SSC to consider the socioeconomic consequences of EBFM plans than has been the case with single species plans. Expertise of SSC membership may have to be broadened to support EBFM recommendations.

Outside the NEFMC, changes to the stock assessment workshop process may be required. While peer-reviewed stock assessments will continue to be required, there will also be a need for peer-reviewed analyses on the overall state and productivity of the EPUs. Dialogue with NMFS will be required on how best to provide these.

Next steps

If the NEFMC adopts the EBFM vision proposed in this paper, a more detailed implementation plan will be developed that outlines activities during a three to five year transition period. The list of activities to be undertaken is not insignificant and relates to both management functions and structures.

Regarding EBFM functions, the top priority is the definition of the boundaries of the ecosystem production units and associated EBFM management units. The boundaries of the EPUs are a property of the ecosystem and can be expected to change over time. The boundaries of EBFM management units, on the other hand, are governance structures and are likely to be less flexible to change. It will be necessary to identify the issues and ecosystem services associated with each EPU and EBFM management unit that require attention under EBFM. Each EPU consists of many ecosystem components with varying levels of sensitivity to fishing impacts. It will be necessary to evaluate those ecosystem components most at risk to fishing, which in turn implies having EBFM objectives for each of these. This may require a risk assessment similar to those described by Fletcher (2005) and Hobday et al. (2007). Identifying the sensitive ecosystem components will facilitate the development of fishery management strategies under a full EBFM. These will be particularly important to inform the analysis of allocation trade-offs, which will be a critical feature of the new management system.

Finally, it will be necessary to define EPU status and productivity reporting requirements, along with the associated ecosystem-level assessment tools required to monitor progress toward EBFM objectives. These will build upon current efforts of ecosystem assessment that have made significant progress over the past five years.

Regarding EBFM structures, these involve a full range of NEFMC institutions. At the council level, it will be necessary to design consultative processes to facilitate greater participation and transparency for EBFM. This is not a trivial task, given the number of stakeholder groups in the New England region. The NEFMC will also need to have dialogue with the Mid-Atlantic Fishery Management Council, the Atlantic States Marine Fisheries Commission, and the New England states to harmonize EBFM efforts.

At the plan development team level, it will be necessary to outline the EBFM plan requirements for each EPU/EBFM management unit. It will also be necessary to design plan development team structures for each EPU/EBFM management unit and have dialogue with the current plan development teams to develop an institutional transition plan.

At the SSC level, there will need to be dialogue with NMFS and NEFMC staff on stock and EPU assessment needs, as well as those of socio-cultural and economic assessment. In relation to the latter, the

SSC has already planned development of a white paper outlining socioeconomic analyses required by EBFM.

Overall, the implementation of EBFM in the New England region has significant consequences for what the New England Fishery Management Council has to achieve and how it organizes itself to achieve this.

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Accounting for Predators in Ecosystem-Based Management of Herring Fisheries of the Western Scotian Shelf, Canada

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Abstract

The southwest Nova Scotia/Bay of Fundy herring fishery, which has landings of approximately 50,000 t per year, is increasingly being managed collaboratively (with Department of Fisheries and Oceans managers, scientists, and fishermen) using a diverse set of objectives that are consistent with an ecosystem approach. Ecosystem-based management must take account of natural mortality due to predation and the question of how many herring should be left for predation to ensure a healthy ecosystem. We conducted a multispecies VPA (MSVPA) centered on herring and its predators to describe the predation level compared with fishing removals, and answer two main questions: (1) what is the relative importance of predation on herring compared to the herring fishery; and (2) how would the inclusion of predation needs change management? Results show that the biomass of herring removed by predation is similar to or higher than that removed by the fishery. The herring population is now at a historical low biomass and is characterized by the absence of the strong cohorts that were observed in the past. Under the assumption that the recruitment will remain at the average of the last 10 years, fishing at or below F0.1 is the only strategy examined that would lead to an increase in the population size and maintain the biomass of herring available to predation.

Introduction

The southwest Nova Scotia/Bay of Fundy herring fishery (*Clupea harengus*) is one of the major herring fisheries in the western Atlantic, with annual landings in recent years in excess of 50,000 t. The fishery includes coastal traps (weirs), gillnets, and a fleet of mobile purse seiners. This fishery has a tradition of innovative management—it was one of the first fisheries to be managed under limited entry (1970), nationally allocated exploitation limits (total allowable catches [TACs] in 1972), and a system of vessel quotas (individual transferable quotas [ITQs] in 1976) (Stephenson et al. 1993).

As a result of a workshop in 1997 (DFO 1997), the explicit conservation objectives for this fishery were expanded to include diverse elements consistent with what is now recognized as the ecosystem approach. The conservation objectives for this fishery are (1) to maintain the reproductive capacity of herring in each management unit by maintaining spatial and temporal diversity of spawning; (2) to prevent growth overfishing by continuing to strive for fishing mortality at or below F0.1; and (3) to maintain ecosystem integrity/ecological relationships ("ecosystem balance"). The third objective includes consideration of herring as forage for other species, and the issue of the cumulative removals of fisheries at the same trophic level; however, these elements have remained largely unquantified to date.

Intensive exploitation has reduced the abundance of numerous fish around the world, prompting researchers and managers to look at the impact on the ecosystems. There are indications that exploiting all species of an ecosystem at Fmsy could lead to serious changes in ecosystem structure (e.g., Walters et al. 2005). Multispecies virtual population analysis (MSVPA) has been used in several ecosystems to estimate predation mortality and consumption of exploited species and their prey, often forage species (e.g., herring, sandlance, pollock). Resulting models have been expanded to evaluate different fisheries or climate scenarios (e.g., Furness and Tasker 1997, Harvey et al. 2003, Tyrrell et al. 2011 and references within) or the impact of a newly abundant species on the food web (Floeter et al. 2005).

This study aimed to quantify the level of predation compared to fishing and estimate natural mortality at age. Based on these findings, we evaluated various management scenarios on the basis of their benefits for predators, herring population, and fisheries. We employed a simple multispecies virtual population analysis (MSVPA) centered on herring that includes all predators known to commonly consume herring. The study area is the Bay of Fundy and western Scotian Shelf, corresponding to the Northwest Atlantic Fisheries Organization (NAFO) statistical area 4X (Fig. 1). We modeled the period 1970-2008, which encompasses contrasting changes in herring biomass and predator abundance.

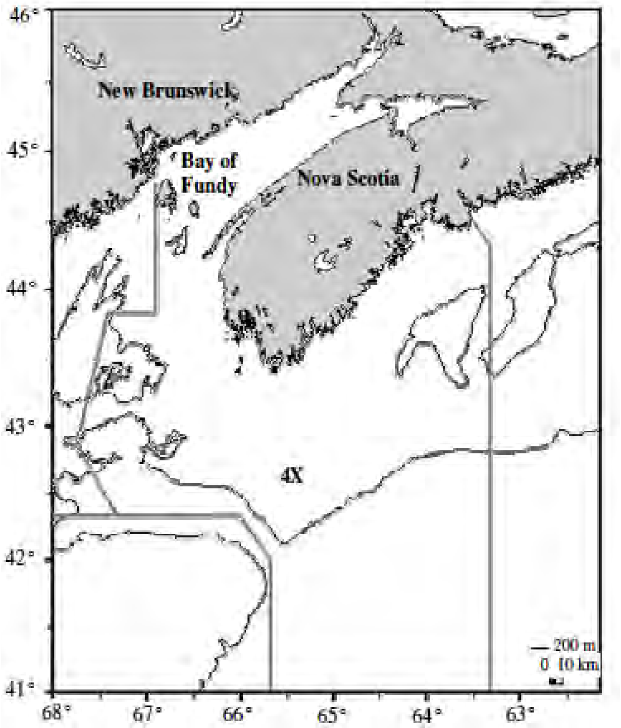

Figure 1. Location of the study area, NAFO division 4X (east coast of Canada) down to the 200 m depth contour.

Methods

Model

We used an MSVPA in which predator abundances are a fixed input in the model, while the population dynamics of herring is a function of the fisheries catch, predation, and the observed age structure. This MSVPA has a simplified structure, in comparison to a full ecosystem model, in that we calculate the dynamics of only one prey item (herring) and there are no population dynamics for predators. A multispecies VPA is an extension of a single species VPA, where predation mortality at age (M2) is calculated instead of being a fixed input into the model that can bias the assessment. The MSVPA follows the formulation described in Magnússon (1995) in which the number of herring of age a eaten by a predator population in year y (predation deaths D) is the result of the biomass of prey available to the predator divided by the suitable total biomass of prey plus the biomass of other prey in the ecosystem. This is expressed as:

$$D_{a,y} = \sum_{pred}\sum_{A} \frac{S_{a,A,pred}\, w_{a,y}\, \overline{N}_{a,y}\, R_{A,pred}\, \overline{N}_{y,A,pred}}{S_{ex,A,pred}\, B_{ex} + \sum_{\forall a} S_{a,A,pred}\, w_{a,y}\, \overline{N}_{a,y}} \qquad \text{eq. 1}$$

where $S_{a,A,pred}$ is the suitability of the prey at age a for the predator *pred* at age A; $\overline{N}_{a,y}$ and $w_{a,y}$ are the annual average prey abundance and weight at age and year y, $R_{A,pred}$ is the ration (in kg per year per individual or by unit of weight) needed by a predator at age A and remains constant; $\overline{N}_{y,A,pred}$ is the average abundance of the predator in year y; B_{ex} is the biomass of other prey in the ecosystem, assumed constant, and $S_{ex,A,pred}$ the suitability of other prey for the predator. The suitability of herring as prey, which is assumed constant over years, is derived using average biomass of the prey for years in which diet composition is available (hereafter called reference years) and the average proportion of herring in their diet composition (P) during the reference years. Suitabilities of herring for a given predator are estimated by iteration using Magnússon's (1995) equation:

$$S_{a,A,pred} = \frac{\overline{P}_{a,A,pred}\,(w_{a,y}\,\overline{N}_{a,y})^{-1}}{\overline{P}_{ex,A,pred}\,(B_{ex})^{-1} + \sum_{\forall a} \overline{P}_{a,A,pred}\,(w_{a,y}\,\overline{N}_{a,y})^{-1}} \qquad \text{eq. 2}$$

$$S_{ex,A,pred} = \frac{\overline{P}_{ex,A,pred}\,B_{ex}^{-1}}{\overline{P}_{ex,A,pred}\,(B_{ex})^{-1} + \sum_{\forall a} \overline{P}_{a,A,pred}\,(w_{a,y}\,\overline{N}_{a,y})^{-1}} \qquad \text{eq. 3}$$

This formulation results in a multispecies type II functional relationship between predation and prey abundance (Magnússon 1995). This means that as herring abundance increases, the number consumed by a predator increases rapidly at low abundance but the rate of increase slows at higher abundance. It also implies that the proportion of herring in the diet of a given predator depends on the abundance of the prey at the moment. As herring abundance decreases, it is assumed the predator will compensate by obtaining a larger portion of its diet from other prey.

The data required for a VPA for herring include the fishery catch, a value for the natural mortality, an estimate for fishing mortality in the last year considered in the analysis (terminal F), and an abundance index. The terminal F (age 11+) was set equal to that of age 10 calculated by the VPA, and R for the last two years were set at 3.5 billion, the average number of herring of age 1 obtained in preliminary runs.

In the MSVPA, the natural mortality is separated into predation (M2) and residual (M1) mortality, which is all the mortality not accounted for in the model (disease, predation by other predators). M1 was set at 0.01 per year for the base model, which assumes that most predators have been taken into account in the model and that disease mortality is small for herring compared with fishing and predation. In an alternate scenario, M1 was set at 0.1, which would assume, as it was in the North Sea model (ICES 2002), that a number of predators (mammals in their cases) were not taken into account. In addition, the MSVPA requires the average weight-at-age for herring, and the time series of abundance for each predator (from VPA or survey), the ration, and the proportion of herring in their diet composition during the reference years.

The MSVPA was performed with ADAPT, the software used by the Department of Fisheries and Oceans Canada (DFO) to perform assessments with VPA (Gavaris 1988, Rivard and Gavaris 2000) to estimate the herring population. The resulting population estimate was input in a program written in J (J software Inc.) to calculate suitabilities (eq. 2 and 3) and herring predation deaths (eq. 1). The algorithm used to solve the MSVPA equations starts by using an initial abundance of herring at age from a VPA that does not include predation deaths. Then iterations are carried out to (1) calculate suitabilities, $S_{a,A,pred}$ and $S_{ex,A,pred}$ using eq. 2 and 3; (2) calculate predation deaths, $D_{y,a}$, and add this to fishery catch $C_{y,a}$; and (3) solve the VPA to obtain new values of N for herring and the combined fishing and predation mortality (zp). Steps 1 to 3 are repeated until zp is stabilized. Following this process, predation mortality was estimated as $M2 = (zp) \times D_{y,a} / (C_{y,a} + D_{y,a})$. Given that the recruitment of the two last years of the population estimate from the VPA was fixed, years 2007-2008 were excluded from the results.

Data

Fish

Herring catch- and weight-at-age (ages 1 to 11+) were obtained from the current time series available for this fishery since 1965 (DFO 2010a). Acoustic surveys coupled with biological sampling provided the index of abundance for years 1999-2009 (DFO 2010a). Herring was separated into two groups in diet compositions: juveniles (ages 1 and 2; <22 cm) that tend to form schools in inshore waters, and adults (3+), which were found in mixed-age schools more offshore. This separation is also convenient because length distributions tend to overlap starting at age 3. Herring predators are structured in 16 groups of which 12 are fish. Cod and pollock were structured by age, as the data were available from assessments or from research vessel surveys (Table 1). Halibut biomass was obtained from the latest stock assessment (K. Trzcinski, DFO, Dartmouth, pers. comm.). For other groundfish species, biomasses

Table 1. Ration, consumption per unit biomass (Q/B), percent herring in diet, reference years, residency, and sources for diet and abundance time series for fish species. Fish lengths are in fork length (FL).

Species	Age/size group	Q/B (per yr)	Ration (kg/yr)	% Herring		Diet reference years	Residency	Sources	
				Juvenile	Adult			Diet	Abundance
Cod, *Gadus morhua*	Age 1-6	3.39	2.83	10.99	10.08	2000-2008	1	a	c
Pollock, *Pollachius virens*	Age 1-6	9.42	4.10	5.36	8.14	2000-2008	1	a	c
Silver hake, *Merluccius bilinearis*	<25 cm	4.86	0.59	0	0	2000-2008	1	a	d
	25-31 cm	3.16	1.11	0	0	2000-2008	1	a	d
	>31 cm	2.47	2.12	23.48	4.98	2000-2008	1	a	d
White hake, *Urophycis tenuis*	<41 cm	7.81	1.91	0.008	0	2000-2008	1	a	d
	>41 cm	4.57	6.99	16.76	3.33	2000-2008	1	a	d
Halibut, *Hippoglossus hippoglossus*	<46 cm	7.15	3.59	3.48	0	2000-2008	1	a	e
	46-81 cm	4.15	11.01	10.92	0	2000-2008	1	a	e
	>81 cm	2.27	39.34	1.87	0	2000-2008	0.5	a	e
Bluefin tuna, *Thunnus thynnus*		7.3		0	52.80	1988-1992	0.33	f	g
Dogfish, *Squalus acanthias*		2.46	3.81	2.84	1.11	2000-2008	1	a	d
Mako, *Isurus oxyrhinchus*		6.94	56.65	0	2	1972-1978	0.33	h	i
Porbeagle, *Lamna nasus*		0.73		0	6.20	1999-2001	0.33	j	k
Blue shark, *Prionace glauca*		1.24	448.95	0	2.50	1972-1978	0.5	l	m
Monkfish, *Lophius americanus*		3.13	3.53	12.46	14.71	2000-2008	1	a	d
Sea raven, *Hemitripterus americanus*		3.82	2.66	14.01	0	2000-2008	1	a	d

a Ration, Q/B, and diet composition from the database (Laurinolli et al. 2004).
b VPA (Clark and Perley 2006).
c Combination of VPA and survey (H. Stone, DFO, St. Andrews, 2010, pers. comm.).
d Annual survey.
e Stock assessment (K. Trzcinski, DFO, Dartmouth, pers. comm.).
f Ration and Q/B (Butler et al. 2010); diet (Chase 2001); residency from tagging data (J. Neilson, DFO, St. Andrews, pers. comm.).
g ICCAT 2008a.
h (Stillwell and Kohler 1978, Kohler and Stillwell 1981); residency (Campana et al. 2007).
i (ICCAT 2008b, fig. 24).
j (Joyce et al. 2002); residency (Campana et al. 2009, ICCAT 2008a).
k (Campana et al. 2009, p. 13).
l (Kohler and Stillwell 1981); residency (Campana et al. 2004).
m (Campana et al. 2004).

were taken from the standardized annual DFO ecosystem trawl surveys and adjusted for catchability and change in boats and gears during the study period (J. de Araújo, DFO, Dartmouth, 2010, pers. comm.). Sharks and bluefin tuna biomasses were derived from assessments obtained for the North or the Northwest Atlantic scaled down to the 4X region assuming that the catch in the region was proportional to the biomass $B_{4x} = B_{Atl} \times C_{4x} / C_{Atl}$. Blue sharks were assumed to have maintained a constant population, at least in the area, based on the lack of clear trends resulting from population models (ICCAT 2008a). According to the summer survey, dogfish population biomass increased since the 1970s, which is consistent with the fact that fishermen have complained since the late 1990s that dogfish abundance was much higher than in the past (D. Clark, DFO, pers. comm.). At the time, dogfish were regularly observed in large numbers around purse seines being emptied (M. Power, DFO, 2010, pers. comm.) and constituted the main bycatch in the herring fishery (Power 2006). Silver and white hakes and Atlantic halibut were structured by broader size groups that take into account diet change and spatial structure.

Diet (percent in weight) and ration (g per day) for demersal fish were obtained from the DFO stomach content database (Laurinolli et al. 2004, Carruthers et al. 2005) and from the literature for sharks and tuna (Table 1). To account for seasonal variations, the proportion of herring in the diet was calculated from a simple mean of estimates from the summer DFO database and the U.S. fall and spring surveys in 4X (J. Link, NOAA Fisheries, pers. comm.). Ration per 3 mm length class derived from the database (R_i) were often highly variable due to sampling variability. Thus, trends in ration were smoothed (Rs_i) using theoretical rations defined as $Rtheo_I = w_i^{2/3}$ (w_i being the mean weight observed in the sample for the size or age class i (Walters and Post 1993), then scaled using the weighted average of observed ration over size classes:

$$Rs_i = Rtheo_i R_i (\overline{R_i})^{-1} \qquad \text{eq. 4}$$

The overlap in spatial distribution of herring and a given predator is a function of their respective behavior and the amount of time the predator spends in the area. The residency in the area, expressed as a proportion of the year (Table 1), was used to adjust the biomass of predator effectively foraging in the area. For instance, adult halibut (>81 cm FL) are known to spend a large portion of their life in deeper waters and were assumed to be present in the study area half of the year.

Marine birds

The seabird group includes the 10 most important pelagic birds reported in the PIROP (Programme intégré de recherches sur les oiseaux

Table 2. Bird biomass, consumption per unit biomass (Q/B), percentage of herring in diets, and number of days present in the study area. Diet was assumed representative of 2000-2006 for all birds.

Common name	Species	N days	Biomass 4X (t)	Q/B (per yr)	% Herring
Northern fulmar	*Fulmarus glacialis*	179	19	84	12.5
Greater shearwater	*Puffinus gravis*	123	134	145	9
Sooty shearwater	*Puffinus griseus*	141	5	112	2.4
Black-legged kittiwake	*Rissa tridactyla*	118	29	111	42
Herring gull	*Larus argentatus*	112	195	97	25
Great black-backed gull	*Larus marinus*	138	189	36	25
Atlantic puffin	*Fratercula arctica*	322	4.7	186	42.6
Arctic tern	*Sterna paradisaea*	220	0.53	202	39
Common tern	*Sterna hirundo*	123	0.16	222	49
Razorbill	*Alca torda*	282	1.33	114	65
Razorbill (fall)		60	1.8	114	
Total or weighted average			578	90	22

pélagiques) database and four nesting birds (Atlantic puffin, arctic tern, common tern, razorbill) (Table 2). Absolute abundance of pelagic birds from the PIROP survey was calculated based on average 10 minute counts per km converted to densities, stratified by season, and grouped over the period 1966-1992 (Huettmann 2000). The abundance of nesting birds was obtained from counts at colonies, and it was assumed that birds counted at the colonies constitute a fourth of the population (A.W. Diamond, University of New Brunswick Fredericton, 2010, pers. comm.). Their biomass was obtained from abundance and body weight, assuming the birds spent the breeding season plus feeding time, according to estimates published for the Northwest Atlantic (NAFO divisions 2J+3, north of the present study area, ICES 2000). In absence of further information, the total biomass of 578 t was assumed to have remained constant during the study period.

Food consumption was obtained from activity budgets, general diet types, and prey caloric content. Pelagic bird data were taken from Huettmann (2000) while rations for the four nesting birds were calculated using caloric needs for the breeding period and the rest of the year, time spent in the area, and prey type for the NW Atlantic (ICES 2000). Diets used in Huettmann (2000) were revised using other

Table 3. **Abundance estimated in 1999 and prorated for percent of the year present, biomass, body weight, consumption per unit biomass (Q/B), and percent of herring in the diet of marine mammals. All scenarios were performed with the conservative percentage of herring in the diet (percent base), while the scenario mmHigh used the higher percentage (percent high).**

					Estimate for 1999				Herring in the diet						
Species	Body weight (kg)	% Year present		Q/B (per yr)	N summer		N pro-rated	Biomass (t)	% Base	% High	Reference years a	Size herring (cm)	% Juv	% Adult	
Other mammals															
Fin whale, *Balaenoptera physalus*	37,000	68	b	4.5	679	c	459	16,966	5	17	indefinite	d	20	80	e
Minke whale, *Balaenoptera acutorostrata*	6,566	51	b	6.3	2,294	c	1,176	7,719	23		indefinite	d	20	80	f
White-side dolphin, *Lagenorhynchus acutus*	92	33	g	14.8	27,297	c	9,099	837	15	44	1991-2006	25.9 (18-32)	20	80	h
Harbor porpoise, *Phocoena phocoena*	31	50	i	18.4	64,671	j	32,335	1,002	44	64	1991-1994	25.5 (18-33)	30	70	k
Common dolphin, *Delphinus delphis*	80	33	l	15.2	2,312	c	771	62	15	50	1999-2002	d	20	80	h
Harbor seal, *Phoca vitulina*	64	100		9.9	9,603	m	9,603	615	27		Indefinite	f	20	80	f
Weighted average				**5.9**					**15**	**53**					
Sum								**27,201**							
Humpback whale, *Megaptera novaeangliae*	30,408	42	n	4.6	300	p	300	9,114	17		Indefinite	d	20	80	e
Gray seal, *Halichoerus grypus*	160	100		7.6	25,391	q	25,391	4,063	37		1988-1990	34.5 (15-45)	20	80	r
All mammals								**40,377**							

a Indefinite reference years were attributed to the 1990s for the calculation of suitabilities in the MSVPA.
b (Link et al. 2006).
c Average of 1999 and 2004 (Palka 2006).
d Assumed proportions based on their distribution.
e (Pauly et al. 1998).
f (Overholtz et al. 1991).
g (Palka et al. 1997).
h (Craddock et al. 2009).
i (Waring et al. 2009).
j Year 1999 (Palka 2000).
k (Recchia and Read 1989, Fontaine et al. 1994).
l (Gowans and Whitehead 1995).
m Year 1995 (Fowler and Stobo 2005).
n (Paquet et al. 1997).
p Year 1999 (Palka 2006).
q (K. Trzcinski, DFO, Dartmouth, pers. comm.).
r Study in 1988-90 (Bowen et al. 1993) but assumed current in the 1990s.

sources. For instance, Brown (1981) has shown that shearwaters fed on capelin in Newfoundland but mainly euphausids at Brier Island (Bay of Fundy), taking advantage of the large shoals of euphausids produced in the area, and feeding to a lesser degree on herring. Diet compositions for nesting birds were obtained from studies at Machias Island between 1995 and 2005 (Atlantic Laboratory for Avian Research, http://www.unb.ca/acwern/msi/diet.htm). For other species, diet compositions were assembled from various studies on the Canadian Atlantic coast over the last 40 years. We chose to use 2000-2006 as reference years for the calculation of suitabilities, assuming that the fragmentary diet information is applicable to recent years.

Marine mammals

Marine mammals were organized in three groups: humpback whale, gray seal, and "other mammals"; the latter includes all species for which we had no trend in population abundance (Table 3). Abundance estimates for marine mammals came from the annual summer survey carried out by NOAA for the northeastern coast, which includes the lower Bay of Fundy and southwest tip of the Scotian Shelf in Canadian waters (Table 3). Presumably, whales gather at the lower Bay of Fundy because of the large production and retention of plankton in this area. In comparison, the upper Bay of Fundy is characterized by tidal flushing of the production and the presence of small fish, winter flounder, kelp, and bryozoans (D. Clark, DFO, St. Andrews, 2010, pers. comm.) and thus, whales are present in lower densities if not absent. Hence, the density of the southern tip of Nova Scotia surveyed by NOAA was extrapolated to the entire western Scotian Shelf in statistical area 4X while densities of the northern Gulf of Maine were not extrapolated to the upper Bay of Fundy. The Atlantic humpback whale population was estimated at 10,752 in 1992-93 with an average increase of 3.1% per year since 1979 (Stevick et al. 2003). Assuming that densities in the study area followed the Northwest Atlantic pattern, and starting from the 1999 estimate of 300 (Palka 2006), humpback whale numbers would be 158 in 1979 and 409 in 2009. Given the variation in the use of feeding grounds observed for this species (Payne et al. 1986, 1990), however, the number of humpbacks present in statistical area 4X in the study area over the period 1970-2006 did not, in all likelihood, follow such a steady rate of increase. The biomass time series for gray seal living in 4X increased from 254 t in 1970 to 4,439 t in 1999 (K. Trzcinski, DFO, Dartmouth, pers. comm.).

Residence time for mammals was obtained either by averaging the proportion of the population abundance per season compared with the peak season (generally summer) as observed in the Gulf of Maine (Kenney et al. 1997, Link et al. 2006) or derived from the literature (Table 3). Phocid seal ration (kg per day per individual) was calculated

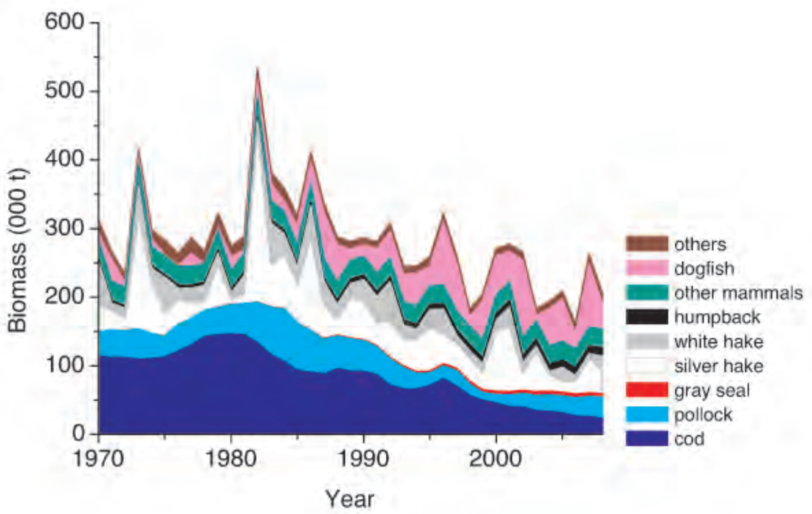

Figure 2. **Trends in herring predator biomass between 1970 and 2008, used as input in the model (sources in Table 1).**

from eq. 5 in Innes et al. (1987): $R = 0.068 \times W^{0.68}$, while that of all other mammals was calculated as $0.1 \times W^{0.8}$ (Trites and Heise 1996) where W is the mean body weight in kg. Diet compositions were derived from the literature (Table 3).

The estimated total biomass of herring predators used in the model decreased from 360,000 t to 225,000 t between the early 1980s and 2006 (Fig. 2). Without the dogfish contribution however, the decline is steeper, decreasing from around 316,000 t to 124,000 t.

Uncertainty, scenarios, and projections

The major sources of uncertainty in this model are residual mortality (M1), consumption rate and proportion of herring eaten by predators, and the predators' abundance trends (especially those of birds and the other mammals group). The MSVPA assumed that catch and predation are known without errors, so it is not well suited to perform a stochastic analysis of uncertainty. Hence, we used scenarios to explore the uncertainty about the percentage of herring in diets; five scenarios were run to investigate the impact on the predicted herring biomass and biomass consumed by predators.

The "base" scenario is the MSVPA model run carried out with small residual mortality M1 = 0.01, and the proportion of diet listed in Tables

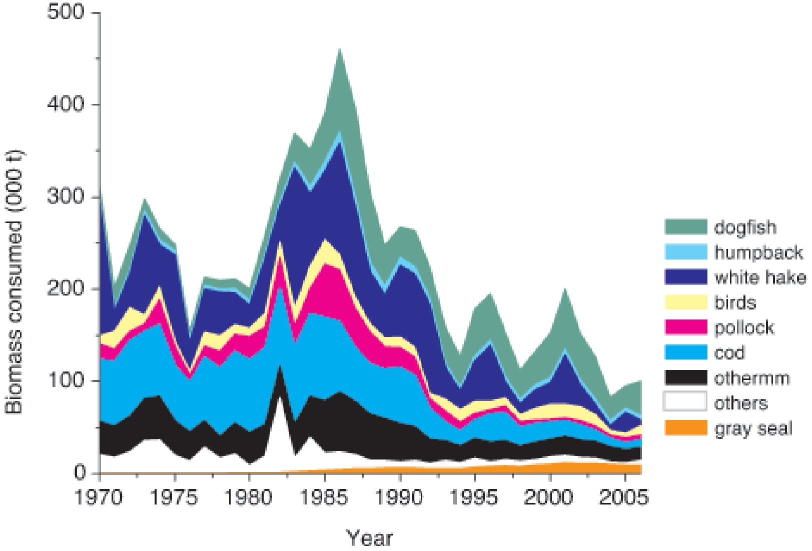

Figure 3. Herring biomass consumed by predators using the base scenario.

1 to 3. We ran an additional MSVPA (scenario "Mhigh") with M1 set at 0.1, which we considered as relatively high in comparison to the default M of 0.2 generally used for this species in stock assessment. The most important predators of herring were cod, pollock, white hake, dogfish, birds, gray seal, and other mammals (Fig. 3), which are responsible for 94% of the herring consumed. The estimate of herring in fish diet varied from 20% to 130% for an average of 50% (Bundy et al. 2011). We used 50% as a first estimate of uncertainty based solely on percent of herring consumed, assuming that the uncertainty was of the same range as for mammals and birds. In scenarios "plus50" and "minus50" the herring consumption by the most important predators entered in the model were respectively increased and decreased by 50%.

The literature consulted suggested both low and high values for the percentage of herring consumed, depending on the area and the period studies, for harbor porpoise, fin whales, white-sided dolphin, and common dolphin, all of which are included in the other mammals group (Table 3). All scenarios used the low percentage (15%) of herring in diets except for the scenario "mmHigh" which used the higher estimate (53%). There is a large uncertainty about the level of feeding carried on the shelf by dogfish, and thus its spatial overlap with herring. The

Table 4. Fcurrent, recruitment at age 1, and predation mortality (M2) used for projections into the future.

			Predation mortality	
Scenario	Fcurrent	R (billions)	Juvenile	Adult
minus50	0.54	1.87	0.49	0.26
base	0.44	3.16	0.64	0.37
plus50	0.36	4.49	0.72	0.45

scenario "dogfish" assumes that dogfish feed on the shelf and thus on herring year-round instead of half the year (residency = 0.5, Table 1) used in all other scenarios.

Projections from 2006 to 2025 were performed using a Thompson Bell yield-per-recruit model using three starting MSVPAs: scenarios base, minus50, or plus50 (the two scenarios producing the most extreme results). The herring population in biomass was obtained using:

$$B_{a+1,t+1} = N_{a,t} W_{beg,a,t} \exp(-Za) \quad \text{eq. 5}$$

$$Z_a = PR_a F + M2_a + M1 \quad \text{eq. 6}$$

where $W_{beg,a,t}$ is the weight-at-age at the beginning of the year t, $M1$ = 0.01, $M2_a$ is defined as the average of the 2000-2006 estimates for the starting MSVPA, PR_a is the partial recruitment at age a defined as the ratio of current *F*, and the average for fully recruited ages (6-8) estimated for years 2000-2006; all parameters are assumed constant over the projection period (Table 4). PR_a resulted from the MSVPA base scenario (0.002, 0.35, 0.54, 0.64, 0.98, 0.97, 1, 1, 1, 1, 1 for ages 1 to 11+) as it does not change much among scenarios. Numbers at age 1 (recruitment) were set at the geometric average obtained from the starting MSVPA for years 1996-2006 (Table 4). Deaths by predation are estimated by:

$$D_{a,t} = N_{a,t} W_{mid,a,t} M2_a [1 - \exp(-Z_a)]/Z_a \quad \text{eq. 7}$$

$$C_{a,t} = N_{a,t} W_{mid,a,t} F_a [1 - \exp(-Z_a)]/Z_a \quad \text{eq. 8}$$

where $W_{mid,a}$ is the average weight-at-age. Projections were carried out using F0.1 (0.23) and Fmax (0.57), which are currently used for fisheries management, and current *F*, resulting from the starting MSVPA (Table 4).

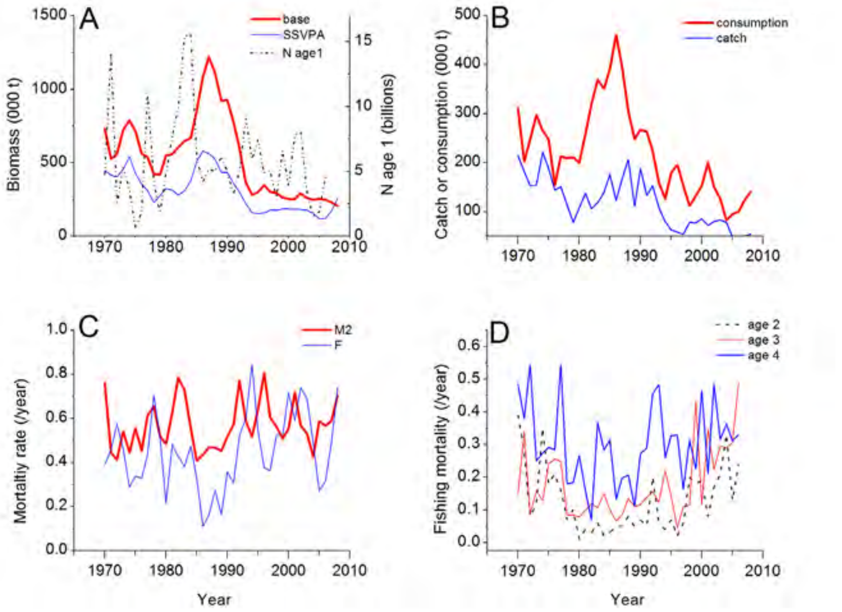

Figure 4. Results from the MSVPA base scenario. (A) Herring biomass estimates from the SSVPA and the MSVPA base, and the number of herring at age 1 estimated by the MSVPA. (B) Herring deaths by fishing and predation. (C) Herring fishing (for ages 5+) and natural mortality rates. (D) Herring fishing mortality by age group under the base scenario.

Results

The biomass estimate for herring resulting from the single species VPA (SSVPA) peaked at around 579,000 t in 1986 and declined to about 166,000 t between 2000 and 2006. Estimated herring biomass increased by an average of 28% when predation was taken into account (in the MSVPA base scenario), reaching a maximum of ~1.2 million t in 1987 (Fig. 4). The percentage of juvenile herring in the population estimated with the MSVPA amounted to 71% on average compared to 52% in the SSVPA. This is partially caused by the fact that predators mainly target juveniles, which compose 72% of deaths by predation in numbers (31% in biomass). Recruitment at age 1 estimated with the MSVPA varied widely over the study period. Since the 1990s, however, the recruitment variation has been reduced and there has been no cohort of the magnitude observed at the beginning of the time series (Fig. 4).

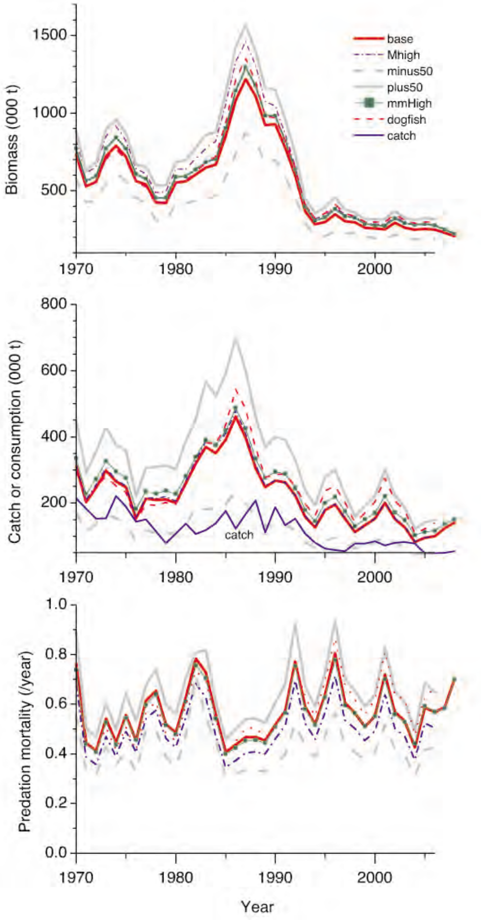

Figure 5. Estimates of herring population, predation mortality, and herring consumed for five scenarios compared to the base scenario. See text for definitions of scenarios.

According to the MSVPA base scenario, predators removed an estimated 130,000 t per year during the period 2000-2006, half the removals of 1970-1975 which averaged 262,000 t per year. In comparison, the current fishery removed on average 71,000 t annually between 2000 and 2006 and 186,000 t per year from 1970 to 1975. The resulting weighted average mortality rates caused by predation varied appreciably each year, with an average of around 0.56 (Fig. 4). During the same period the rate of fishing mortality for age 5+ herring was predicted to have increased from 0.45 in the 1970s to 0.54 in the 2000s (Fig. 4C). The juvenile (age 1-2) mortality rate from predation was estimated at 0.64 and the adult mortality rates at 0.37. The model predicted an increase in fishing mortality since 1990 and more so for herring of ages 2 and 3 (Fig. 4D). For example, 3 year old herring were subjected to a fishing mortality rate of 0.18 in the 1970s, 0.10 in the 1980s, 0.16 in the 1990s, and 0.3 in the 2000s. In 2000-2006, birds accounted for 8.5% (10,974 t) of herring predation, fish for 66% (85,329 t), and marine mammals (all groups) for 26% (33,486 t) (Fig. 3). Estimated consumption of herring by gray seal and dogfish increased with the increase in their population abundance in the area.

Setting the residual natural mortality M1 to 0.1 instead of 0.01 (scenario Mhigh) led to 12% lower estimate of predation mortality M2 and a 16% higher herring population estimate over the study period (Fig. 5). As M1 is assumed to be constant, trends in estimated biomass and mortality remain unchanged and the biomass of herring consumed remained essentially at the same level.

Scenarios minus50 and plus50, which assumed that the consumption of herring was being under or overestimated by 50% for major predators, predicted a corresponding 48% change in the estimation of herring consumed (Fig. 5), and a 25% change in herring population. It is worth noting that the predicted herring consumption with scenario minus50 is at the same level as the fisheries catch. The average predation mortality M2 was estimated at 0.65 in scenario plus50 and at 0.42 in scenario minus50. Conversely, fishing mortality was relatively less important in scenario plus50 (0.36) than in scenario minus50 (0.54). Using a higher proportion of herring in the diet for other mammals (mmHigh) increased the biomass consumed by 12% and the herring population biomass by 8%. Assuming that dogfish eat twice as much herring (scenario dogfish) had large implications for both herring consumption (+35%) and biomass (+18%) reaching levels approaching those of scenario plus50. Due to an increasing trend in dogfish biomass, this scenario was the only one resulting in an increasing trend over time for herring biomass, consumption, and mortality.

Projections

The projection using the current *F* and the MSVPA base scenario predicted lower equilibrium yield of herring (49,000 t), which is equal to 83% of the observed average catch during the period 2004-2006 (average 58,700 t). Scenarios minus50 and plus50 predicted similar values (catch = 50,000 t and 48,000 t or 85% and 82% respectively) (Table 5). Under the same scenario, the spawning biomass was predicted to decrease to 92% (1.01 to 0.89 for scenarios minus50 and plus50) of the starting value, while the biomass of herring consumed would increase by 16% (20-

Table 5. Spawning biomass, predation deaths, catch, and ratio of the projection end/starting value resulting from projections using F0.1, Fcurrent, and Fmax, starting from three scenarios: minus50, base, and plus50.

		Projection					
Scenario	Starting value	F0.1		Fcurrent		Fmax	
Spawning biomass (t)		Value	Ratio	Value	Ratio	Value	Ratio
minus50	121,324	220,382	1.82	123,120	1.01	117,350	0.97
base	157,515	208,018	1.32	145,528	0.92	121,637	0.77
plus50	194,105	214,001	1.10	173,330	0.89	131,541	0.68
Predation deaths (t)							
minus50	48,965	83,175	1.70	58,657	1.20	57,185	1.17
base	92,401	30,200	1.41	107,126	1.16	98,163	1.06
plus50	137,108	175,378	1.28	157,149	1.15	138,103	1.01
Yield (t)							
minus50	58,696	40,391	0.69	50,084	0.85	50,349	0.86
base	58,696	37,255	0.63	48,908	0.83	52,340	0.89
plus50	58,696	38,011	0.65	48,105	0.82	56,952	0.97

15%) from 92,400 t to 107,100 t. The projections using Fmax predicted higher catches (52,300 t) than with F0.1 but still at 89% (86-97%) of the starting value, a decrease in spawning biomass to 77% (68-97%) of the starting values and 6% larger levels of herring consumed (6%, 1-17%). The F0.1 projection led to a 32% (10-82%) increase in spawning biomass to 208,000 t, and a 41% (28-70%) increase in biomass consumed. This scenario implied, however, a 63% (65-60%) decrease in yield to ~37,000-40,000 t.

As expected, predictions were consistent for the three starting scenarios. The stronger decrease in biomass obtained with the starting scenario plus50 and the Fmax projection is caused by the lower Fcurrent (0.36) relative to Fmax (0.57).

Discussion

The MSVPA model presented in this paper is a first attempt to quantify mortality due to predation in this area, and to consider herring population dynamics in light of the needs of their predators. This model is based on the principle that, given the "known" abundance of predators present in the ecosystem during the study period, there must have been a sufficient biomass of herring to meet minimal diet requirements. Therefore, given the catch-at-age structure and the abundance index, the more predators or the larger their consumption rate, the larger the estimates of herring population. There are large uncertainties associated with the diet data and the predators' preference for herring. The present model showed that different estimates in consumption rate, in the percentage of herring in the diet, or in the amount of residual mortality (M1), causes the estimated herring biomass to be scaled upward or downward without any changes in trends. On the contrary, dogfish, with its increasing abundance over the study period and the uncertainty about the percentage of time spent feeding on herring, caused a change in trend as well as in level of predicted herring abundance.

Recent stomach content analysis defined the groundfish consumption of herring for recent years, but the uncertainty was higher for mammals and birds for which local and recent diet analysis are more difficult to obtain. Also, the dynamics of fish such as pollock, which show a net preference for krill, were probably not well modeled here as they are unlikely to increase their herring consumption in the presence of large krill abundance. We briefly addressed some sources of uncertainty listed earlier, but a more stochastic treatment of uncertainties would be warranted in the future. Nevertheless, some sources of uncertainty pertaining to dogfish or mammal population trends, for example, would be difficult to estimate given the current state of knowledge and thus, the range of uncertainty would likely be underestimated. We suggest further fieldwork to address some crucial lack of information such as mammal diets and abundance, and dogfish behavior if MSVPA modeling is to be continued or if predation mortality is to be defined further.

Although the estimates of herring consumed decreased in recent years (a function of reduced herring and predator abundance), the estimates of herring biomass consumed were similar to 2004-2006 landings in the fishery (scenario minus50) or more than twice as much (plus50). Tyrrell et al. (2008) estimated that predators' consumption was 3-5 times as high as the landings in the northeast continental shelf of the U.S. where herring has increased in recent years. On Georges Bank, the six fish predators included in the MSVPA consumed 5-10 times the yield in recent years as herring abundance increased sharply in the 1990s (Tsou and Collie 2001).

The average estimate of mortality rate due to predation (M2) based on the MSVPA was about 0.64 for juveniles (ages 1 and 2) and 0.37 for adults. This indicates that although the value for M used previously in the VPA (0.2) (DFO 2010a) is consistent with the life span of herring, it seriously underestimates the level of predation on this species. In the North Sea, herring natural mortality rate for ages 1 to 4+ has been estimated at 1, 0.3, 0.2, and 0.1 respectively (ICES 2010). Our estimates are lower for age 1 because of the assumptions made for herring distribution and the equal allocation of herring of ages 1 and 2 in the diet of predators. Tyrrell et al. (2008) estimated a very high M2 at age 0 and an average of ~0.8 for all ages of herring in northeastern U.S. waters, compared with our estimated average of 0.56. On Georges Bank (Tsou and Collie 2001) predation mortality was estimated at 0.75 at age 1, lower starting at age 2, and as low as 0.06 at age 5. Their relatively low predation rate on large herring could be due to the limited range of predators considered. Predation mortality predicted by our model was highly variable, a result consistent with all other MSVPA predictions found in the literature. However, we cannot tell at the moment how much of the interannual variation is significant.

We do not know the ecological consequences of the decrease in herring abundance. For instance, cod, known as a ubiquitous predator, has not shown changes in body condition in the Bay of Fundy, suggesting that feeding is adequate, but the condition factor has declined since 2000 on the western Scotian Shelf (Clark and Perley 2006). However, based on the catch-at-age and the results of the assessment, the cod population is characterized by unexplained "mortality" of older cod that could be attributed to unknown predation mortality or to emigration to other feeding grounds. The data currently available do not allow differentiation among hypotheses.

The model assumed that a decline in herring would be compensated by predators turning to other prey, a group that is supposed stable and abundant enough. In reality, prey species are not all equal in terms of availability, quality, and nutritional energy. Fatty fish such as herring are known to be especially important for mammals and birds as they provide an essential energy source for reproductive success (Österblom et al. 2008). In the Scotian Shelf/Bay of Fundy, sand lance and euphausids are possible substitutes for herring for some predators. Humpback and fin whales have been shown to move feeding grounds and target prey species as abundance changes (Brown et al. 1981). Birds' specific abilities to hunt dictate the ratio cost:energy in chasing a particular prey and hence, limit their ability to compensate with other prey (Brown et al. 1981). A future version of this model would have to include other prey species to allow better estimates of herring consumption. To this end, further fieldwork on the abundance trends and behavior of other prey would be instrumental in understanding the food web dynamics of the Scotian Shelf.

A viable fisheries management plan under an ecosystem approach should set precautionary reference points to protect the forage fish and its predators (DFO 2010b). In spite of the large uncertainty in absolute estimates, management could at least aim at maintaining the current level of predation. Our simulations showed that under present conditions the current fishing mortality would likely allow present levels of consumption, but that this would be accompanied by a slight decrease in spawning biomass. A fishing mortality rate at or below the F0.1 policy is required to supply about 40% more herring for consumption (assuming current levels of predators' abundance) and allow for the rebuilding of the population. Although the F0.1 target fishing mortality has been an inherent part of the management policy, this objective has not been met in 20 years. The average *F* in the 1990s and 2000s reached 0.50 and 0.54 respectively. In addition, fishing pressure increased over the last 10 years, especially on younger fish (ages 3 and 4) due to changes in fishing practices.

The herring population and the fishery are dependent on the periodically high recruitment and in the past each large recruitment event sustained the fishery for several years. Interestingly, there have been no strong year classes (no year classes that have survived/remained strong) in the past decade. The chronically low recruitment is probably due to a combination of decline in productivity in the ecosystem (Melvin et al. 2009), a decline in spawners biomass, and the drastic reduction of some sub-stocks (DFO 2010a). In any case, the only solution is to conserve the herring population while waiting/hoping for better recruitment. This suggests that the fishery should aim at fishing below F0.1 for a number of years.

This should not be considered favoring predators over fishermen. Rather, we are treating the issue of sustainability not only for the fisheries, but also for the predator/prey system in accordance with the third objective set for this fishery. The model results suggest that when herring are abundant, both fisheries and predators benefit (as could be inferred from the level of herring consumed and fished in the 1970s). In the case of return to large recruitment in the future, the question of allocation fishery/predation will become crucial and should probably be tackled well in advance in a management strategy plan.

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Using a Regional Level, Risk-Based Framework to Cost Effectively Implement Ecosystem-Based Fisheries Management

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Abstract

Risk-based frameworks to implement the "ecosystem approach" have been developed for a number of different industries in Australia. The framework for individual fisheries has been used for nearly a decade and while valuable it (i) does not address the cumulative effects of fishing, (ii) does not align with regional level planning undertaken by other government agencies, (iii) has not halted the increasingly negative community perceptions about fishing. To address these issues, use of a regional level approach, termed ecosystem-based fisheries management (EBFM), was proposed with the draft EBFM framework trialed for one bioregion in Western Australia (WA). Given the success of the trial, this paper outlines subsequent refinements to the methodology, the progress made in applying the framework in all bioregions of WA, and the broader adoption of these principles by the Department of Fisheries and other agencies.

Being a hierarchical, risk-based process, the EBFM framework avoids merely generating an impossibly complex set of regional level issues, uncertainties, and stakeholder expectations. In the initial case study over 600 ecological assets, social and economic outcomes, governance systems, and external drivers were identified by stakeholder workshops. The complexity was reduced by consolidating them into 60 regional level risks and a multi-criteria analysis was used to integrate related ecological, social, and economic values and risks into 24 "agency level" priorities. This framework has been applied to all six aquatic bioregions in WA with the resultant 88 agency priorities now used as the basis

for all annual budget-setting decisions made by the Department of Fisheries. To fully implement EBFM, WA is currently revising the fisheries legislation and governance arrangements to facilitate creation of regional level strategies to coordinate the management of all individual fisheries/activities and simplify the department's engagement in future multi-sector (EBM), regional planning processes.

Initiating implementation of EBFM did not require detailed data on ecosystems; it required only the holistic consideration of risk to each ecological asset and the associated stakeholder benefits to determine which assets have the greatest requirement for direct management. The cost effective steps for a regional level, ecosystem-based approach using only currently available data combined with expert opinion make implementation of this management planning framework viable in any location.

Introduction

Background

There has been considerable effort within Australia to translate the various ecosystem-based concepts into practical outcomes that are useful for management (Fletcher 2008). Risk-based frameworks are now available for a wide variety of industries including the management of individual fisheries (Fletcher et al. 2005, Fletcher 2010); aquaculture (Fletcher et al. 2004); agriculture (Chesson and Whitworth 2005); and irrigation usage (Camkin et al. 2007). From the application of these concepts across a variety of situations, we identified the four universal management "principles" that ensure a holistic ("ecosystem") approach is taken irrespective of the industry.

Four Universal Principles for Holistic Management

1. What impacts are the activities I control having on the **assets** that I manage?

2. What impacts are these activities having on the **assets** that someone else manages?

3. What economic/social benefits and costs are generated from these activities and the use of my **assets**?

4. What activities managed by others affect my **assets** and me?

(Modified from Fletcher and Chesson 2008.)

In applying these principles, the "my" can be an individual, a company, a fishery, a region, a department, a state, a country, or even an entire continent. So depending upon what the "my" represents, the

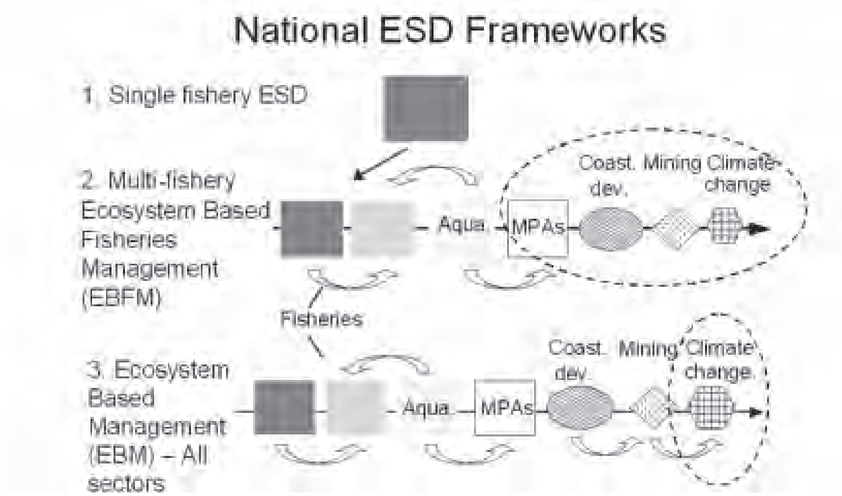

Figure 1. Relationship between the three ecosystem-based framework levels. In this instance single fishery ESD is equivalent to EAF. The elements included in the dashed ovals represent the difference in external drivers between EBFM compared to EBM. Abbreviations are used for aquaculture (Aqua.), marine protected areas (MPAs), and coastal development (Coast. dev.) Modified from Fletcher (2006), Fletcher et al. (2010).

scope and complexity covered by the management system can vary dramatically. For fisheries management, there are three common levels at which an ecosystem approach can be applied (Fletcher 2006): the individual fishery (EAF), multi-fishery level (EBFM), and multi-sector level (EBM) (see Fig. 1).

The steps to apply the ecosystem approach to individual fisheries are based on the international standard for risk management (AS/NZS 4360 1999, AS/NZS 2009), reflecting that fisheries management is really just a specific form of risk management (Fletcher 2005; Fig. 2). These steps are routinely applied in Australia (Fletcher 2008) and have been adapted for use elsewhere (e.g., FAO 2005, Cochrane et al. 2008, Fletcher 2010) with a variety of tools available to undertake each step to suit different fishery and country situations (e.g., De Young et al. 2008; FAO www.fao.org/fishery/eaf-net/en).

A national workshop (see Millington and Fletcher 2008) concluded that while extremely valuable, applying an ecosystem approach at the fishery level did not address the cumulative effects of all fishing-related activities in a region or deal with the conflicting objectives and

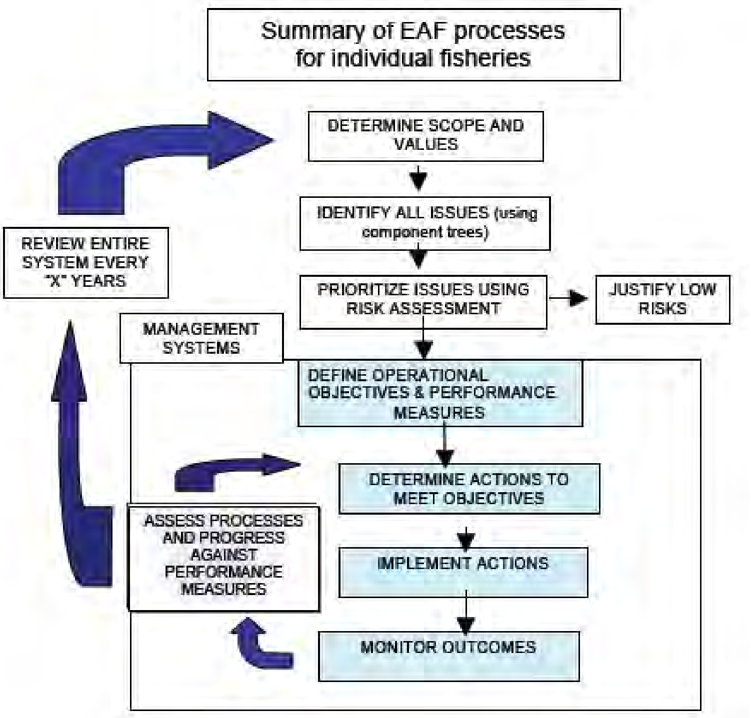

Figure 2. The EAF framework for individual fisheries. Adapted from the AS/NZS 4360 Risk Management framework, and modified from Fletcher (2008)

allocation issues between fisheries or sectors (e.g., commercial and recreational). Managing only at this level can hinder linkages to other government planning processes that operate at a regional level (e.g., establishment of marine parks) and it has not halted the increasing community perception that fishers (especially commercial fishers) no longer have an automatic "social license" to operate. The workshop proposed that taking a regional level (multi-fishery) ecosystem-based fishery management approach (EBFM, see Fig. 1.) would assist in dealing with many of these issues.

Potential and perceived problems

When developing the methods to apply EBFM in Western Australia (WA), a number of potential and perceived problems were identified. First, there was a tendency for many stakeholders to assume this must involve the collection of more "ecosystem" information (Fletcher 2008). But, without increased resources, it was essential that EBFM could begin without increased information. Second, undertaking regional level assessments had to avoid merely generating an impossibly complex set of issues, systems, uncertainties, and expectations. Finally, the EBFM process had to complement, not duplicate, the activities already being progressed within fishery level management systems.

Given these issues, many sections of the department became skeptical that EBFM could be implemented cost effectively and without significant disruptions to services. There was a fear that the EBFM initiative would shift the focus off core activities and even potentially affect career development. Many were convinced that it was just an academically attractive but impractical concept that would not improve management outcomes.

It is against this background that we began to evaluate whether there was any real value to government from implementing EBFM. The rest of the paper outlines the set of activities developed to deal with the issues outlined above which resulted in this risk-based "EBFM framework" ultimately being adopted and implemented by the department in a cost effective, practical, and useful manner.

Materials and methods

Overview of the EBFM framework

To deal with the complexity of regional level issues generated by EBFM, the single fishery framework (EAF[1]) had to be modified into a hierarchical, regional level, risk based framework (Fig. 3). A description of each of the key steps in the EBFM framework, including some recent refinements, is presented below using specific examples from the West Coast Bioregion of WA (see Fletcher et al. 2010 for more details).

Commitment, capacity, and responsibility

The first and most important step in successfully implementing EBFM (or any management system) is having an appropriate level of political commitment and institutional capacity that will enable suitable management arrangements to be developed and enforced. The most common cause of fisheries management failure is the lack of good governance, not the lack of information (Kruse 2011). The lack of governance is often due to the lack of will to undertake what is known should be done.

Operating at the regional level can often require clarification of who has legislative responsibility for what assets and where. The one asset may "belong" to different jurisdictions, and conflict frequently arises

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[1] EAF is the term used by FAO; in Australia it is more generally known as ESD—Ecologically Sustainable Development.

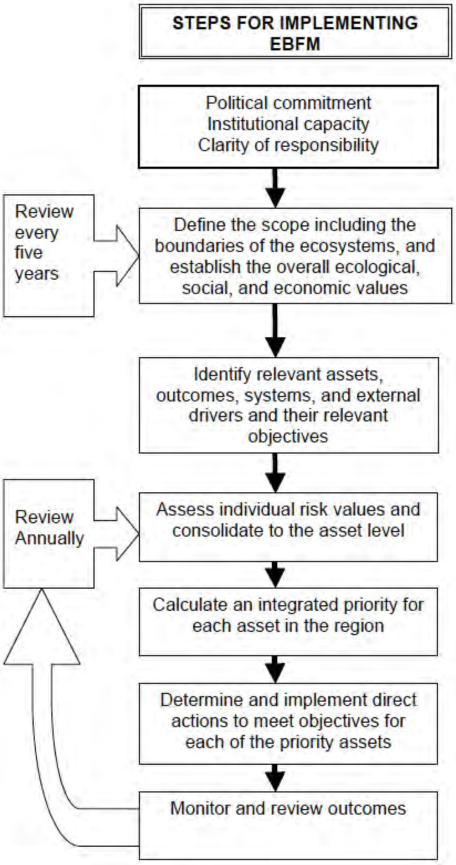

Figure 3. Outline of the planning steps used for the implementation of a regional level ecosystem approach—multi-fishery (EBFM) or multi-sector (EBM).

where more than one agency/group is trying to manage the one asset. This can occur between agencies of the one government, between state and federal governments, and frequently between countries. It is, therefore, essential to understand the limits of your legislative responsibility because this defines what you can directly manage, compared to what you can only influence or react to (see also Fletcher 2006). Depending upon the degree of overlap, it may be necessary for more than one agency (i.e., more than one legislative act) to be directly involved.

Table 1. The main values identified as relevant to EBFM in Western Australia. Note: this list of values does not include food security, which would be a key value in many developing countries

	Value	High level objective
1	Species sustainability	Keeping biomass levels above levels where recruitment could be affected
2	Ecosystem sustainability	Ensuring that any impacts on ecosystem structure and function are kept at acceptable levels
3	Economic outcomes	The economic benefits to the community are optimized
4	Social amenity	The social amenity (i.e., non-economic benefits) derived by the community is optimized
5	Social impacts	Social impacts and negative attitudes associated with management of these resources are minimized

Defining the scope

Based on the legislative arrangements of the agency(s) involved, the scope of what will be included in the EBFM assessment needs to be clearly described. This includes the geographic boundaries of the area that will be encompassed, and developing a very clear description of each of the relevant fisheries and other activities that are being managed in this region. For the WC example, the region encompassed for EBFM was a 1,000 km stretch of coastline in the southwest of WA from Kalbarri (27°S) in the mid-west, south to Augusta (115°30E), out to the 200 m depth contour, including all fishing related activities that occurred in those waters (Fletcher and Santoro 2010).

The scoping process must also generate a shared understanding of the relevant social, economic, and ecological values desired by the various stakeholder groups. Essentially, what does the community want to achieve from undertaking management of the region's resources? The values (or high level objectives) can include ecological sustainability, food security, social amenity, and economic development. The combination and relative priority among these can vary dramatically among countries, among regions, and even between assets within the one region.

Understanding which of these values is the most important has major implications for what should be managed and how best to manage it. In the West Coast case study, the primary objective was ecological sustainability, with social and economic outcomes being secondary objectives; food security was not considered relevant in this situation.

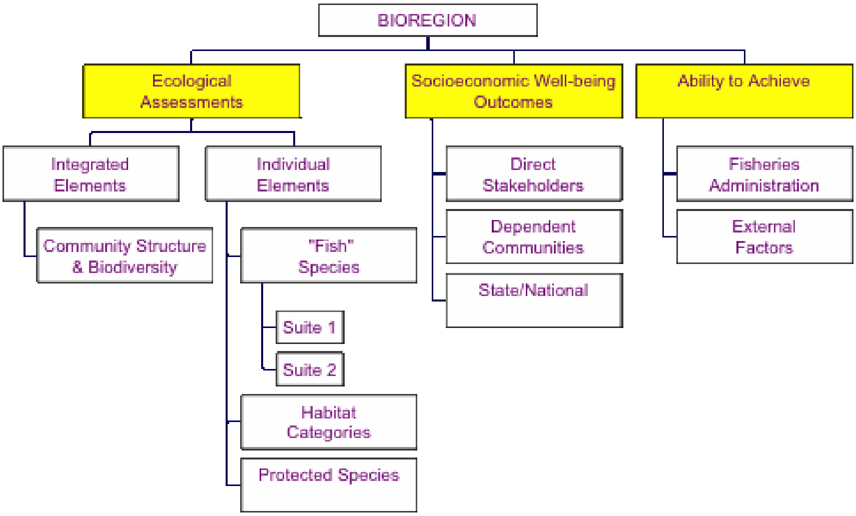

Figure 4. Key EBFM components. Each component expands into its own tree and branch structure, which are subsequently modified to suit the region being assessed.

Identification of EBFM assets, outcomes, and specific objectives.

Using the agreed scope and values, the next step is to identify all the relevant EBFM items (Fig. 4) and determine what specific objectives are to be achieved given any local, regional, or national requirements or global attitudes (see Table 1). The EBFM items were identified through a series of workshops with a diverse range of participants, including fishery managers, other government agencies, sector representatives, and other stakeholders. The items were structured into a hierarchy of related groups using a set of EBFM component trees with one tree for each for the ecological assets, social and economic outcomes, institutional governance system, and external drivers. Each of the EBFM trees was tailored by adding relevant items and deleting those considered irrelevant. For the West Coast case study, in excess of 600 items were identified across the spectrum of EBFM components with over 80 separate items identified just for the "fish species" asset tree (see Fig. 5 top). This number is clearly too large to be useful for agency planning; therefore, methods to reduce this complexity were applied.

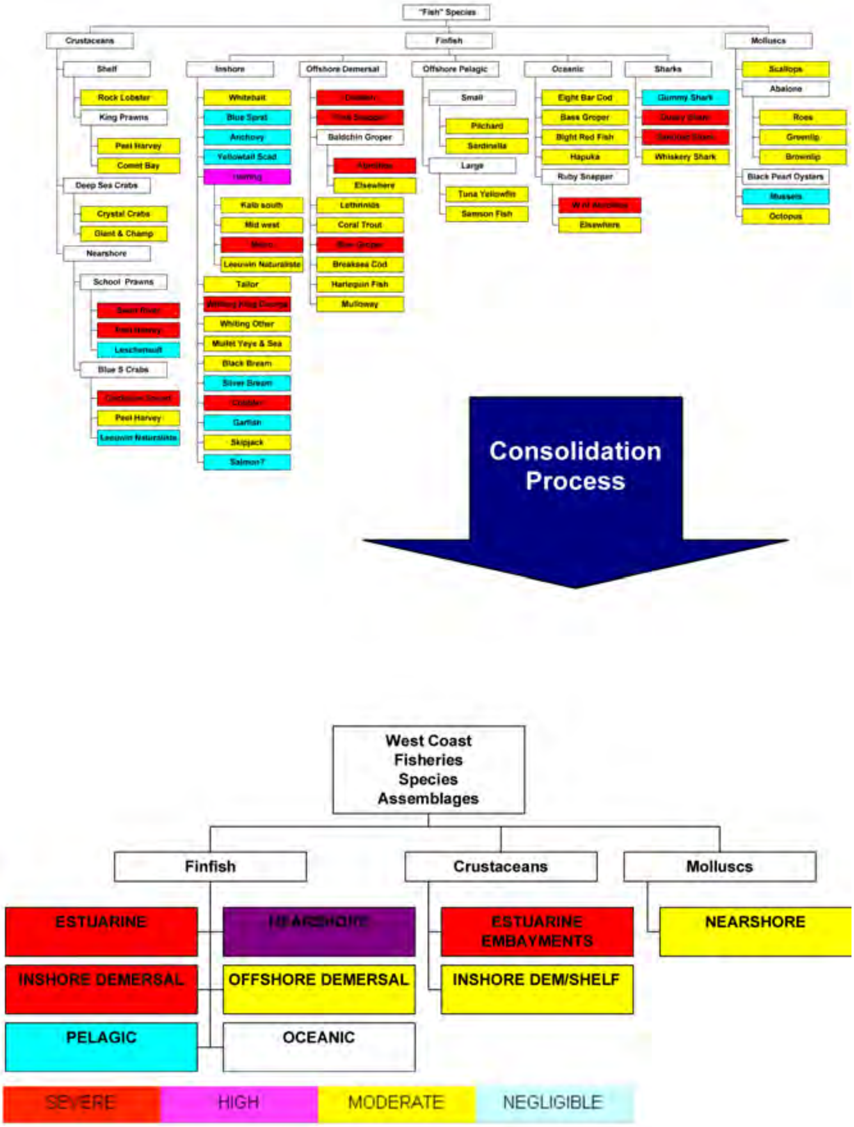

Figure 5. Above: individual "Fish" stock/species risks; and below: consolidated risks of species suites (assets) from the West Coast Bioregion of Western Australia. Colors represent (top) the risk status of the species and (bottom) the entire suite. Modified from Fletcher et al. (2010)

Table 2. Examples from the multi-criteria assessment for the West Coast region showing individual risk and value scores.

Asset	Ecological risk	GVP level	Econonomic risk	Social amenity	Social risk	External impacts (other agencies manage)	Current Dept. of Fisheries score and priority (EBFM)	Whole of govt. score and priority (EBM)	Departmental resources applied
WC crustacean (lobsters)	3	5	5	4	3	0	111 Urgent	111 Urgent	Very high
WC pelagic finfish	2	1	1	1	1	0	4 Very low	4 Very low	Very low
WC estuarine ecosystems	5	3	3	4	4	4	25 Low/mod	125 Urgent	Low
WC governance, external linkages	2	5	4	5	4	0	92 Urgent	92 Urgent	High
WC pests and diseases	3	3	1	3	4	0	45 Medium	45 Medium	Very low

All risk and value levels are scaled from 0 to 5, with 5 the highest value and risk.

GVP = gross value of production; social amenity includes use and non-use values. The values are the relative "weights" used for these criteria.

The EBM formula includes all WA legislation and hence does not include "other agency discounting."

Assessing individual risk values and consolidating to asset level.

The risk levels associated with each of the identified items were assessed using standard risk assessment techniques (e.g., Fletcher 2005, 2010; IEC 2009). While this reduces complexity by showing which items have only low or negligible risks, the number of moderate or higher risk values generated can still be large. Furthermore, many individual risks will already be subject to specific management and planning processes at the fishery level. Therefore, to ensure that the EBFM process recognized and added value to existing fishery level management, these were combined into regional level assets (Fig. 5).

The process of consolidation into broader asset categories utilizes the branch structure of the component trees. Each of the branches represented groups of "like risks" that can/should be managed collectively. For example, the West Coast “fish species” tree had 80 individual species/stock risks, which were consolidated down to just nine species suites (Fig. 5 bottom). Using the risk status for the entire species suite (e.g., inshore demersal finfish) is very efficient for management planning which can be evaluated using the status of one or more indicator species representative of the entire suite. For the case study, this approach enabled the consolidation of the 600 EBFM items down to 60 regional level risks (see Fletcher et al. 2010 for details).

Calculate an integrated priority for each asset in the region

As many of these regional level risks are interrelated, an integrated set of holistic, departmental level priorities can be generated by recognizing that we manage ecological assets to generate economic and social benefits for the community. Each consolidated ecological asset therefore becomes the primary unit to integrate its associated ecological, social, and economic risks and values using a simple multi-criteria function (see Table 2 and Fletcher et al. 2010 for details). Using this approach, the complexity from the >600 West Coast items initially identified was reduced to just 24 agency level priorities.

Agency Priority = (“Stock” Risk – External Impact) × [(Economic Risk × GVP) + (Social Risk × Social Amenity)]

The highest agency priority scores will be calculated when there are risks to the ecological sustainability of the asset, and the asset is valuable economically and/or socially. Within the West Coast case study, the highest departmental score was for the WC crustacean suite, which is dominated by rock lobsters, the largest and most valuable fishery in the state (Table 2). The high priority score reflects the significant issues that were facing the fishery (Fletcher and Santoro 2010), including a series of low recruitment years that required major reductions in the allowable catch to ensure breeding stock levels were not depleted. The economic performance was also being affected by relatively low prices due to overseas market conditions and high exchange rates, which were exacerbating the impacts of increased costs associated with fuel and labor.

One of the lowest Agency Priority scores was for the West Coast pelagic finfish suite (Table 2). This suite of fish currently has only very minor levels of fishing due to poor markets combined with variable and low stock availability in the region (Gaughan et al. 2008), resulting in difficulties maintaining catches at economically viable rates. Consequently, the risks to the stocks are currently low to negligible and there are no additional risks being generated that affect other trophic levels, or social and economic outcomes.

The multi-criteria analysis for EBFM recognizes that if a stock or ecological risk is mostly being generated by human factors external to the fisheries management (legislative) control (e.g., pollution, coastal development), the overall priority for direct departmental activity is likely to be reduced accordingly. These external risks were taken into account in the agency level priorities through use of a "discounting term." For example, the agency level priority for the West Coast estuarine ecosystems was not as high as would be expected given the severe ecological risk level, because the majority of the ecological risks are generated by coastal development and agricultural runoff resulting in sedimentation/loss of habitat, which are managed by other agencies—overall responsibility is under the Swan River Trust. This generates a heavy discount from the agency level priority score for the Department of Fisheries. The department can directly ban only the capture of stocks most at risk (i.e., treat the symptom) and try to influence the other agencies involved to improve water quality (the cause).

In contrast to the formula for EBFM, the "whole of government" formula that would be used for an EBM assessment does not include any "discounting term" because all management agencies and their legislative controls are included. Consequently the EBM priority score for the West Coast estuarine ecosystems is extremely high (Table 2), reflecting increasing community concern about the Swan River Trust's inability to "halt the decline of this most valuable of Perth's natural assets" (*The West Australian*, Oct 12, 2010).

Determine and implement direct actions to meet objectives for each priority asset

The multi-criteria analysis provided a high degree of discrimination in priority among assets for urgency of action and relative use of agency resources. For the 24 regional level assets in the West Coast, there were five urgent priorities, two high priorities, and six moderate, low, and very low priorities. While we have found there was a reasonable degree of concordance between priority levels and levels of departmental funding/activity directed at each asset, suggesting that the previous implicit processes were not completely wrong (see Table 2), for some assets the current activities (and resourcing) were well below and well above that expected (e.g., Table 2). In one case it highlighted the unacceptable risks associated with an asset only recently added to our legislative responsibilities (introduced pests and diseases). To deal with this appropriately, additional resources will be required either directly from government or possibly indirectly from the shipping industry (the main source of the risks), on a cost recovery basis.

In situations where an asset has an unacceptably high sustainability risk (score >3) but low commercial and social value, and therefore a relatively low priority score, this does not mean that the ecological risks are not addressed. If the ecological risk of any asset increases to >3 additional management actions must be generated to reduce the risk level[2], but for those with low social and economic value the actions are more likely to be relatively simple and cost effective.

Monitoring and review

Reflecting the full adoption of the EBFM framework as the basis for management of WA's aquatic resources, the annual monitoring and reporting to the WA Parliament by the department has been amended and renamed the *State of the Fisheries and Aquatic Resources Report* (Fletcher and Santoro 2010). This report now includes bioregional overviews outlining each of the key ecological resources (assets) for the region and summarizes their current cumulative risk status. Furthermore, each of the individual fishery reports has been refocused to become more resource-based rather than activity (sector)-based.

Discussion

Real adoption of the EBFM framework

Merely stepping through any new process, such as the EBFM framework, does not by itself guarantee that the outputs will be used by the agency, or that the process will continue to be used. To ensure effective and ongoing implementation of EBFM, the framework has to become an integral part of the agency's governance system. This is being achieved by changing both the Department of Fisheries' project management system and the budgeting/planning processes.

The department has adopted a full risk-based management system, which requires all activities undertaken by staff to be explicitly assigned to assist the management of risks associated with one or more ecological or organizational assets (see Fig. 6). Having a direct association between activities and assets clearly illustrates to staff and stakeholders that the purpose of a natural resource management agency is to effectively manage the risks (ecological, social, and economic) to the community's assets and outcomes. The differing priority levels for assets will also clearly indicate to stakeholders why some issues will receive greater (or quicker) attention than others.

The annual budget planning cycle for the department has been updated so that the first step in this process is a review of the risk and value scores for each asset. Changes in these values affect the priority scores, which will potentially result in a shift in the level of resources directed to individual assets and regions.

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[2] Except where the risk is being generated by external factors.

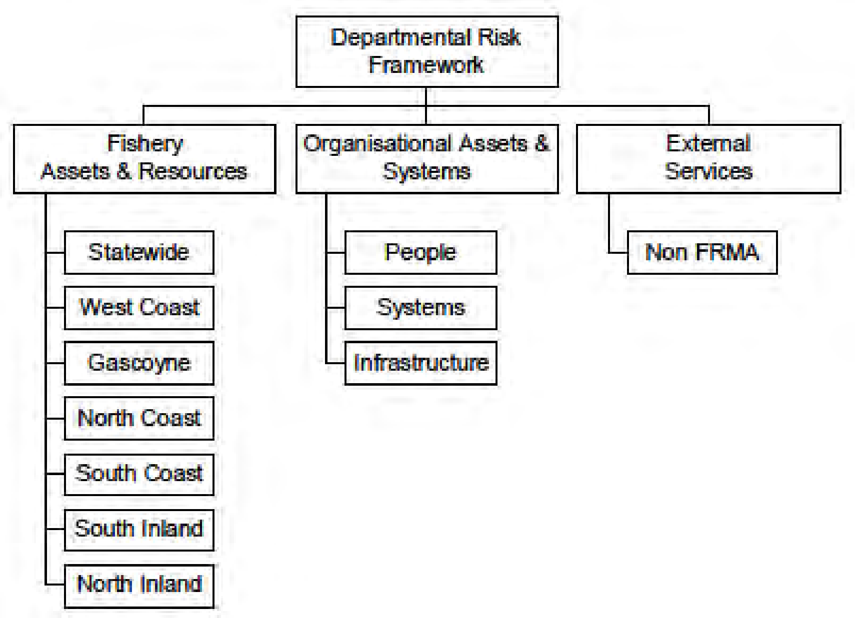

Figure 6. Framework for the Department of Fisheries risk register. Each of these components expands into its own set of branches and assets.

Regional level strategy plans

The next phase in implementing EBFM within WA is to update the Fish Resources Management Act (WA Govt. 1994) to better enable regional level management. The current act only provides a framework to establish management arrangements for individual commercial fisheries and to a lesser extent establish regulations for the management of recreational fishing. There are no “head powers” (legal structures) to manage across multiple sectors, allocate among sectors, or deal with cross government integration (DoF 2010). Consequently the department is in the process of amending the act to require the establishment of a hierarchy of management strategies (Fig. 7). A series of Aquatic Resource Management Strategies (ARMS) will set, at the regional level, overall objectives for management and the parameters for overall resource use and allocation of access. From this overarching plan, the various sectoral harvest use and protection plans can be generated that will outline the specific objectives for each sector in a coordinated manner. This will enable individual fishery issues and actions to be mapped into regional level plans.

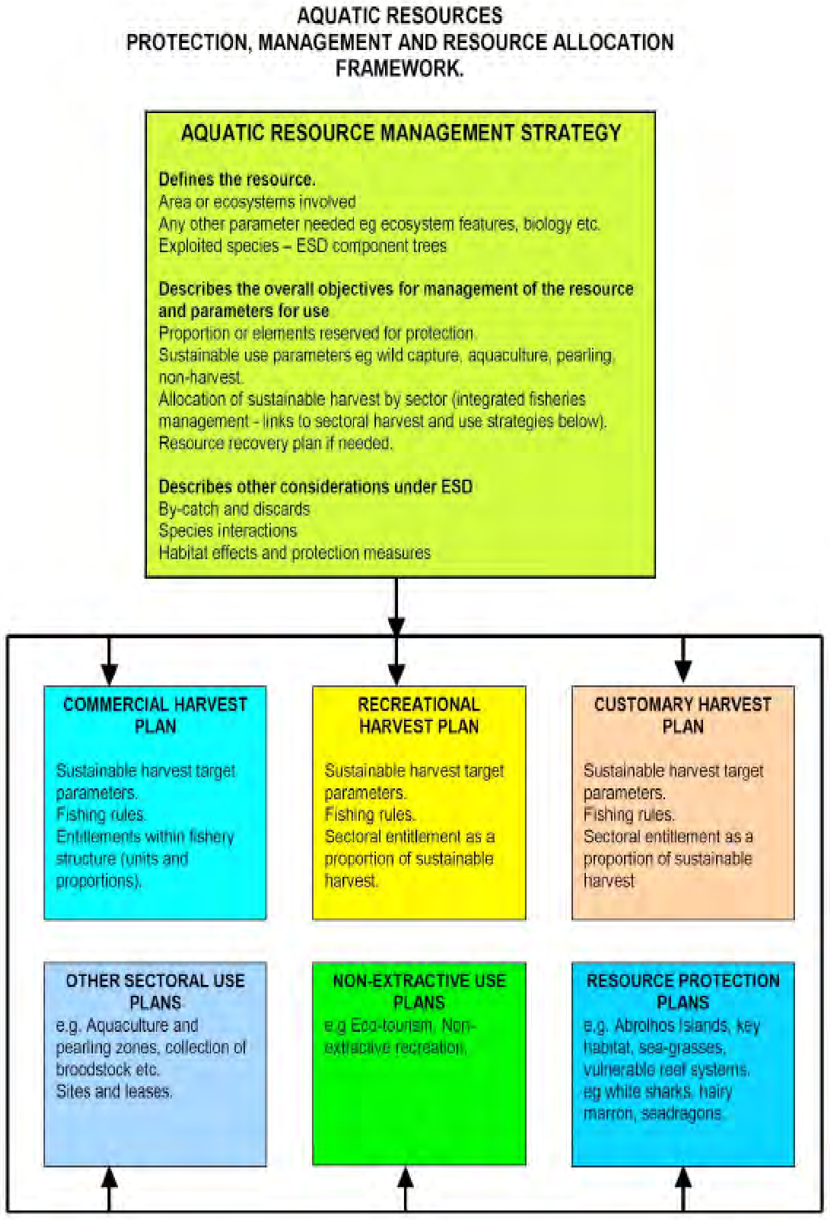

Figure 7. Proposed regional resource planning and management framework for Western Australia (from DoF 2010).

An additional benefit of ARMS is that they can be used as key inputs into any future multi-sector regional planning processes. Without defined regional level strategies, these processes require separate inputs for each fishery and sector, which is highly inefficient, generating a high workload for those involved and often leading to a lack of effective engagement. Everybody involved was very busy attending multiple meetings that resulted in numerous submissions, but the outcomes were mostly superficial or unpopular. The development and use of the ARMS should therefore reduce this level of cross agency inefficiency.

The linking of EBFM with multi-sector (EBM) processes has already begun with the establishment of a national committee that includes representatives of environment and fisheries agencies from both state and federal levels of government in Australia. This group recently agreed that the set of steps to implement multi-sector EBM would be essentially identical to those outlined here for EBFM (MACC 2010). Having a consistent set of steps should increase the likelihood that the outputs from these processes will complement each other and therefore generate improved outcomes for the community, not just activities and reports.

Conclusions

We have found that adopting an "ecosystem approach" at a regional level did not require a detailed understanding of all the relevant ecosystems. Instead, it required the efficient consideration of each ecological asset in the region and its associated stakeholder outcomes, to identify ecological assets that **most** require direct management to deliver the "best" outcomes for the community. The critical step in beginning to adopt EBFM is being able to clearly identify, in a pragmatic and consistent manner, the set of ecological assets to be managed and linking these to social and economic outcomes that they may generate.

The simple set of steps we developed for the EBFM framework has enabled adoption of a fully regional, "ecosystem-based" approach in WA without material increases in funding. It has successfully replaced the previous, disjointed planning systems with a single, coordinated risk-based system that is already generating efficiencies for the use of departmental (government) resources. Having a cost effective process means that it can be applied in all circumstances, not just in regions of the world where a large amount of resources and scientific data are available.

The adoption of risk-based approaches at a regional level as the overarching basis for fisheries management planning, combined with the wider adoption of similar steps to implement EBM, should facilitate more efficient linkages and harmonization with other government policies and processes. Consequently, we have found that there are clear and positive benefits from the implementation of this "ecosystem approach" to assist with fisheries management planning and decision-making. These benefits are much more tangible than merely meeting some long-forgotten global political commitment.

Acknowledgments

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Multispecies Age-Structured Assessment for Groundfish and Sea Lions in Alaska

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Abstract

The current push toward ecosystem-based fisheries management, in conjunction with the limited application of current multispecies models in that context, outlines the need for a more holistic approach that explicitly includes age-structured species interactions. To meet this need, a multispecies age-structured assessment model (MSASA) for the Gulf of Alaska was expanded from three species: arrowtooth flounder (*Atheresthes stomias*), Pacific cod (*Gadus macrocephalus*), and walleye pollock (*Theragra chalcogramma*), to include two major high trophic level predators as external inputs: Pacific halibut (*Hippoglossus stenolepis*) and Steller sea lion (*Eumetopias jubatus*). Inclusion of the large predators resulted in increased predation on older prey ages, including those fully recruited into the commercial fishery. Significant changes to trophic structures and predation linkages from the core model were observed. Estimation of residual natural mortality M_0 was achieved through modification of survey selectivity curves and survey catchability Q values from the core model. Predation mortality, survey selectivity, and M_0 are confounded in their relationship to determining cohort structure. The MSASA model structure is able to track complex population dynamics, but variability in parameter estimates makes clear the need for improved stomach data.

Introduction

The application of mathematical modeling to marine systems is an attempt to explain observable data by mapping unobservable processes (Anderson 2009). Predation mortality is one of the most important of these processes, as it affects every organism in marine systems and can exceed fishing mortality for commercially fished species (Bax 1998, Gaichas et al. 2010). Integration of predation into stock assessments is a fundamental aspect of ecosystem-based fisheries management (EBFM) (Marasco et al. 2007). Multispecies virtual population analysis (MSVPA) and mass-balance models such as Ecopath (Christensen et al. 2000) are currently used by fisheries managers in an advisory capacity but have yet to be fully integrated into stock assessment methods.

Natural mortality M refers to mortality from sources other than the commercial fishery. M has generally been assumed constant in single-species stock assessments and fishery models (Quinn and Deriso 1999). Andersen et al. (2009) and Andersen and Beyer (2006) examined the relationship between natural mortality and growth, and suggested that predation accounts for the entirety of natural mortality for non-apex marine species. Gaichas et al. (2010) concluded that predation comprised the majority of mortality for pollock in the Gulf of Alaska, and that the assumption of a constant natural mortality was erroneous. By separating M into a variable predation mortality P and a residual natural mortality term M_0, model realism is increased and the bias arising from the assumption of a constant natural mortality M is reduced.

Predation mortality, as a major component of M, is confounded with estimates of survey selectivity (Thompson 1994) in terms of defining cohort structure and abundance. Stock assessment estimates of total natural mortality M are sometimes conditioned on assumed selectivity curves (e.g., Turnock and Wilderbuer 2009). Fisheries management is dependent on the estimation of age-specific predation mortality to define conservative biological reference points for species subject to heavy predation (Collie and Gislason 2001, Tyrrell et al. 2011), quantify the cascade of commercial fishery effects through the system, and provide a more accurate assessment of the population structure from which commercial catch is drawn.

The current work expands an existing multispecies age-structured assessment (MSASA) model for the Gulf of Alaska (Van Kirk et al. 2010) from three species—walleye pollock (*Theragra chalcogramma*), arrowtooth flounder (*Atheresthes stomias*), and Pacific cod (*Gadus macrocephalus*)—to five by the addition of Pacific halibut (*Hippoglossus stenolepis)* and Steller sea lion (*Eumetopias jubatus*). Mass balance models show these species to be among the top predators of pollock larger than 20 cm (ages 2+ in the current work) (Aydin et al. 2007). Age 1 pollock are targeted by a number of different predators, but Aydin et al. (2007) showed that arrowtooth flounder and cannibalism remain the largest two sources of predation mortality for all ages of pollock in the Gulf of Alaska. Including these major pollock predators moves the MSASA model closer to a "minimal realistic model" (Punt and Butterworth 1995) in which the major species interactions affecting pollock abundance have been explicitly modeled, allowing practical application to fisheries management.

Predation in the original three-species MSASA model was observed to be disproportionately high on younger prey age-classes, due in part to the enormous abundances of younger ages and in conjunction with the similar sizes of the three modeled species, which limits the number of prey able to be consumed and digested. Larger predators, however, may bring increased pressure on older cohorts. As cod have no modeled predators in the original model but are a major prey of Steller sea lion, the inclusion of sea lion in the model has the potential to alter overall system population dynamics and structure by exerting predation pressure on what was previously a model apex predator.

Methods

Core model structure

Van Kirk et al. (2010) describe the core Gulf of Alaska MSASA model. In overview, standard equations of single-species stock assessment models (Quinn and Deriso 1999) are used to model year-class propagation, commercial catch-at-age, and fishing mortality. Total instantaneous mortality Z is decomposed into fishing mortality F, predation mortality P, and a residual natural mortality term M_0.

Predation mortality is a function of predator and prey abundances, estimated from size- and species-preference parameters in conjunction with annual ingestion requirements. As different data sets utilize different measures of size (length or weight), both length and weight are mapped to age by the application of externally defined length-at-age and weights-at-age bins constructed from the NOAA Alaska Fisheries Science Center (AFSC) Stock Assessment and Fishery Evaluation (SAFE) reports; model equations use the age subscript a for prey species and b for predator species. Predation mortality P is defined as

$$P_{i,a,t} = \frac{1}{B_{i,a,t}} \sum_j \sum_b I_{j,b} N_{j,b,t} \frac{\phi_{i,a,j,b,t}}{\phi_{j,b,t}} \tag{1}$$

in which $I_{j,b}$ is the annual ration for a given predator of species j, age b, $N_{j,b,t}$ is the abundance of predator j,b at the beginning of year t, and $B_{i,a,t}$ is the biomass of prey species i, age a at the beginning of year t. The ratio $\phi_{i,a,j,b,t}/\phi_{j,b,t}$ is the proportion of prey i,a in all food available to predator j,b in year t, assumed equal to the proportion of food within the stomach of predator *j,b* composed of prey *i,a* in year *t,* and defining the overall preference of predator *j,b* to feed upon prey *i,a* in year *t*. The numerator $\phi_{i,a,j,b,t}$ is termed "suitability," as

$$\phi_{i,a,j,b,t} = \rho_{j,i} g_{i,a,j,b} B_{i,a,t} \tag{2}$$

in which $\rho_{i,j}$ defines the preference of predator *j* to feed on species *i*, and $g_{i,a,j,b}$ defines the optimal prey size *i,a* selected by predator *j,b*. The size-preference *g* of predator *j,b* is modeled as a lognormal function following Anderson and Ursin (1977):

$$g_{i,a,j,b} = \exp\left[-\frac{1}{2\sigma_j^2}\left(\ln\frac{w_{j,b}}{w_{i,a}} - \eta_j\right)^2\right] \tag{3}$$

in which σ and η are size-preference parameters specific to each predator *j*, and *w* is the weight-at-age for each age of predator or prey.

The core model makes a number of assumptions, designed to limit model complexity while having minimal qualitative impact on parameter estimates. These include temporally invariant length/weight-at-age, gear selectivity, survey catchability *Q* (set to 1), and predator annual ration. Abundances are annually estimated over modeled years 1981 to 2001. Data for model fitting via maximum likelihood methods, using AD Model Builder (Fournier et al. 2011), include commercial catch and survey abundance taken from SAFE reports, and stomach-content data supplied by the Resource Ecology and Ecosystem Modeling (REEM) database of the AFSC (general information on stomach data collection and processing can be found in Yang and Nelson 2000; relevant data were obtained courtesy of G. Lang, AFSC). The stomach-content data, although primarily sampled during the summer months, are assumed representative of annual feeding habits. Model estimates of commercial catch and survey indices, along with annual abundance trends, are consistent with stock assessments produced by the AFSC (Dorn et al. 2010, Thompson et al. 2010, Turnock and Wilderbuer 2009), and predation curves are in general agreement with similar research (Hollowed et al. 2000), confirming model functionality.

Expanded model

Halibut

The International Pacific Halibut Commission (IPHC) regulatory areas falling within the Gulf of Alaska are areas 3A, 3B, and 2C. Abundance-at-age data for area 3A were supplied by the IPHC and used as indices for abundances in areas 3B and 2C following the relative bottom-area covered by each region; abundances for halibut are fixed model inputs.

Ages of modeled halibut run from 8 to 20+ years. Weights-at-age were supplied by the IPHC. As halibut growth has been exhibiting a drastic decline since the early 1980s, annual mean weights-at-age are used rather than the assumption of a constant mean weight-at-age as applied to pollock, cod, and flounder in the core model.

Mean weight-at-age has a significant effect on predation, as the ratio between predator and prey weight is integral to its estimation (eq. 3). The core model uses a single set of size-preference parameters for each species, with the assumption that size preference is a constant function of gape and physiology; changes in predation in response to strong or weak year classes of prey are better modeled through explicitly coded predator functional response. The continual change in halibut size, however, implies changes in size preference, and three sets of time-specific size-preference parameters were estimated.

Annual ingestion rates were calculated following Aydin et al. (2007). The von Bertalanffy growth equation was used to determine the relationship between the change in weight and total rate of energy assimilation and the rate of energy loss as:

$$\frac{dW}{dx} = HW^{d} - KW^{n} \tag{4}$$

in which *H, d, n,* and *K* are parameters that define the allometric relationship between age and weight as a function of the generalized von Bertalanffy equation, x is age, and W is the weight-at-age; parameter H is the key parameter related to ingestion. Aydin et al. (2007) accepted the suggestion of Essington et al. (2001) that n can be set to 1 with the assumption that respiration and body weight have a linear relationship, and by setting the differential in eq. (4) to zero, obtained an expression for the asymptotic weight as:

$$\left(\frac{H}{K}\right)^{1-d} = W_{\infty} \ . \tag{5}$$

Following the method of Essington et al. (2001) produces:

$$W_{x} = W_{\infty}\left(1 - e^{-K(1-d)(x-x_{0})}\right)^{1/1-d} , \tag{6}$$

in which x_0 is the age at which the weight of the organism is assumed to be zero. Meta-analysis work with predator-prey species in the North Pacific allowed Aydin et al. (2007) to arrive at a value of 0.8 for *d.* Using field studies to set values for weight at age, eq. (6) can be used to solve for W_∞, K, and t_0. Solving eq. (6) for halibut is problematic, however, as

W_∞ and K are correlated, and the rapid shift in halibut growth over time has made parameter fitting difficult. Age of maturation, however, has remained constant, and it was determined that the ratio of weight at 50% maturity (age 11-12) to W_∞ in the early 1990s was 0.4561 (S. Gaichas, NOAA AFSC, Seattle, Washington, pers. comm., 2010). Applying this ratio to the weight-at-age data produced a value for W_∞ that was 1.159% greater than the weight at age 20; this percentage was used to generate annual values for W_∞. Values for K and x_0 were estimated for each year by fitting eq. (6) to observed weights-at-age for ages 8-19; as age 20 is a plus group and thus carries a potentially skewed weight, it was omitted. Then, from eq. (5), the solution for H is:

$$H = Kw_\infty^{0.2}\ , \tag{7}$$

producing an estimate of annual ingestion rate in kilograms consumed for each halibut of age b as:

$$I_b = \frac{H}{A} W_b^d \tag{8}$$

in which A is a scaling parameter to compensate for consumed biomass that is indigestible, set through meta-analysis to 0.6 (Aydin et al. 2007).

Stomach-content data for halibut were supplied by the AFSC Resource Ecology and Ecosystem Modeling program. Of the modeled species, pollock were the most significant prey item in halibut stomachs, followed by arrowtooth flounder, and although some individuals consumed cod, these were infrequent and the data were considered insufficient to provide adequate model forcing. Pollock and flounder are therefore the only modeled prey species for halibut. Stomach data from all sampled individuals within a given year were pooled to show the mean proportion of aggregate prey-at-age weight relative to total aggregate stomach weight for each predator-at-age. A single halibut was considered a sample of one, regardless of the number of prey items contained in its stomach. The total sample size, reflecting predators whose stomach contained pollock or flounder, was 398.

Stomach data were available for 1990, 1993, 1996, and 2001. Data were predominantly gathered in summer months but assumed to represent annual feeding behavior. Estimated halibut stomach-content values were averaged over the first ten years of model run (1981-1990) and fitted to the stomach data from 1990; data for 1993 and 1996 were merged and used to fit model estimates averaged over 1991-1996, and the 2001 data were used to fit model estimates averaged over 1997-2001. This approach was also used in the core model, in which stomach-content data were grouped into three seven-year blocks (period 1: 1981-1987;

period 2: 1988-1994; period 3: 1995-2001). Data scarcity does not allow fitting to each individual year, even where data for a given year exist; averaging over a set of years provides greater forcing to model parameter estimates, and the species-preference coefficient (eq. 3) changes for each time-block, facilitating predation sensitivity to predator-prey abundances. Pooled stomach contents are assumed asymptotically normal relative to increasing sample size; explorations utilizing alternative distributions, including lognormal and multinomial, were unsuccessful. Weightings in the objective function, following Hanselman et al. (2008), are set to the square root of the sample size. The objective function component is a minimized sum of squares.

Steller sea lions

Abundance data for sea lions were taken from National Marine Mammal Laboratory aerial non-pup survey counts in the Gulf of Alaska, available for 1976, 1985, 1989-1992, 1994, 1996, and 2000, and supplied by the AFSC. Observed abundances were multiplied by 1.1331 to compensate for missed animals (Loughlin et al. 1992). A life table (York 1994) and survival rates for males and females (Winship et al. 2002) were used to calculate abundances-at-age with the assumption of a gender-equal birth ratio. Annual pup abundances, assuming a single pup per nursing female, were estimated from maturity and reproductive rates (Winship et al. 2002). Reproductive rates were modified to reflect the decline in observed Gulf of Alaska populations by minimizing the sum of squared differences between the corrected non-pup survey counts and the summed estimated abundances-at-age for years in which survey data exist. Annual ingestion rates for sea lions were assumed different for males, non-nursing females, and nursing females with pups younger than one year, and were taken from the extensive bioenergetic work of Winship et al. (2002). Bioenergetic needs for first-year pups (age 0) are included in the mother's ingestion rate. Weights-at-age are taken from Winship et al. (2001).

Age-classes are modeled from 1 to 13+ years. Age 0 animals are not modeled beyond estimation of their mother's increased ingestion needs as a function of nursing. While some animals continue to nurse until age 3, most have been weaned by age 2, and at age 1 have already begun to supplement nursing with hunting, reducing the drain on parental energy reserves. For convenience, it is assumed that pups aged 1 and above will forage independently and no longer nurse. Maturity and reproductive rates from Winship et al. (2002) are used to estimate the number of nursing mothers per age-class per year. Age 3 is considered the onset of reproductive maturity, and by age 6 all females are considered mature and capable of reproducing.

The literature on sea lion diets is often contradictory. The Marine Mammal Protection Act of 1972 prohibits the taking of live specimens, and much of the available data was gathered prior to the modeled years. Many of the data focus on general prey taxonomy and frequency of prey occurrence, either in examined stomachs or as indicated by the presence of otoliths and other bony parts in sea lion scat, and supply no information regarding prey or predator age, or proportion of a given prey species by weight or volume to the total prey consumed. While it had originally been intended to group sea lions into male, female, and nursing female groups, the structure of the available stomach-content data made this infeasible, and consequently sea lions are merged into a single group with weighted means for weight-at-age and ingestion-at-age. It is assumed that the consumed proportions of each modeled prey species remained constant over time. Upper and lower bounds are set for the total proportion of each modeled prey species in the estimated stomach contents for each age of sea lion; penalties are incurred in the objective function only if estimated stomach-content values fall outside those bounds. Trites and Calkins (2008) provide the most detailed evaluation to date of prey sizes and proportions of species consumed by males and females, from examination of sea lion scat contents recovered from gender-specific haul outs. Sea lion consumption, averaged over gender, was found to be 28.5% gadids, 11% flatfish, and the remainder a mix of salmonids, rockfish, forage fish, and cephalopods. These proportions are used to define minimum-maximum stomach-content bounds: flounder 5%-11%; pollock 11%-28.5%; cod 11%-28.5%. The large variation in the previous studies of Steller sea lion predation precluded any definition of size preferences in the objective function stomach-content matrices. Model estimates of sea lion stomach contents for each combination of predator age and prey age and species are averaged over all modeled years and fit between the minimum and maximum values in the objective function. As pollock appear to be the most commonly reported primary prey item for sea lions (Pitcher 1981, Merrick and Calkins 1996), size-preference (eq. 3) is bound by minimum and maximum pollock weights-at-age. Although sea lions are capable of tearing apart and consuming large prey in smaller pieces, general observation suggests that most sea lions manipulate fish prey in the mouth to facilitate complete ingestion without tearing (S. Atkinson, University of Alaska Fairbanks, School of Fisheries and Ocean Sciences, Juneau, pers. comm. 2010). As with other species, the majority of stomach-content data reviewed in the literature were gathered in summer and assumed seasonally invariant.

Assumed stomach-content distribution and objective function weightings are as other modeled species. For sea lions, the sample size is the sum of sample sizes over all reviewed literature, totaling 2,425.

The expanded model opened both residual natural mortality M_0 and survey catchability Q to estimation, whereas these were input into the core model. Core model assumptions regarding the shape of the selectivity curve were relaxed and alternatives for selectivity estimation were explored, including a double logistic curve, a normalized gamma density function, and a simple vector of point estimates for each age and species.

Results

Initial runs of the expanded model were unable to reach convergence. As with other studies (Fu and Quinn 2000), Q and M_0 were confounded and inversely related: reasonable values for M_0 were associated with unrealistically high values for Q, while setting Q values close to 1 as is commonly done in stock assessments (Dorn et al. 2010, p. 69; Thompson et al. 2010, pp. 166-167; Turnock and Wilderbuer 2009, p. 629) caused residual natural mortality rates to rise beyond acceptable values. As survey selectivity contains implicit assumptions regarding the underlying population structure (Thompson 1994), M_0 values were also affected by the choice of selectivity curves. Setting parameter bounds for Q and M_0 allowed for parameter estimation and model convergence, but these bounds were sufficiently restrictive that they were essentially no better than using input values. Fu and Quinn (2000) recommend setting values for Q while allowing estimation of M_0. Following their work, values for Q were set to those presented in the literature: flounder = 1.3 (Somerton et al. 2007); cod = 0.92 (Nichol et al. 2007, Thompson et al. 2010); pollock = 0.8 (Dorn et al. 2005). Survey selectivity-at-age values s_a from the core model were replaced by a normalized gamma density function for each species (Quinn and Deriso 1999) as:

$$s_a = \frac{a^{\alpha} e^{-\beta a}}{\max_j j^{\alpha} e^{-\beta j}} \,. \tag{9}$$

M_0 was also sensitive to new first-order predation effects from the addition of larger predators and $2^{nd} \ldots n^{th}$ n-order effects from predation cascades. Final model values for M_0 were 0.353 for cod (a decrease from 0.37 in the core model), 0.277 for flounder (an increase from 0.2), and 0.2 for pollock (unchanged).

Objective function values generally improved from core model values, with the largest improvements seen in total annual catch and survey biomass for cod and flounder. An exception was the survey index of cod abundance-at-age, which displayed a poorer fit in the expanded model. Of the stomach-content objective function components, flounder in period 3 and halibut for all periods displayed the poorest fits. Sea lion indices most often exceeded the maximum limit set for feeding on cod ages 4-8, and for sea lions aged 1-5 feeding on ages 5-10 pollock.

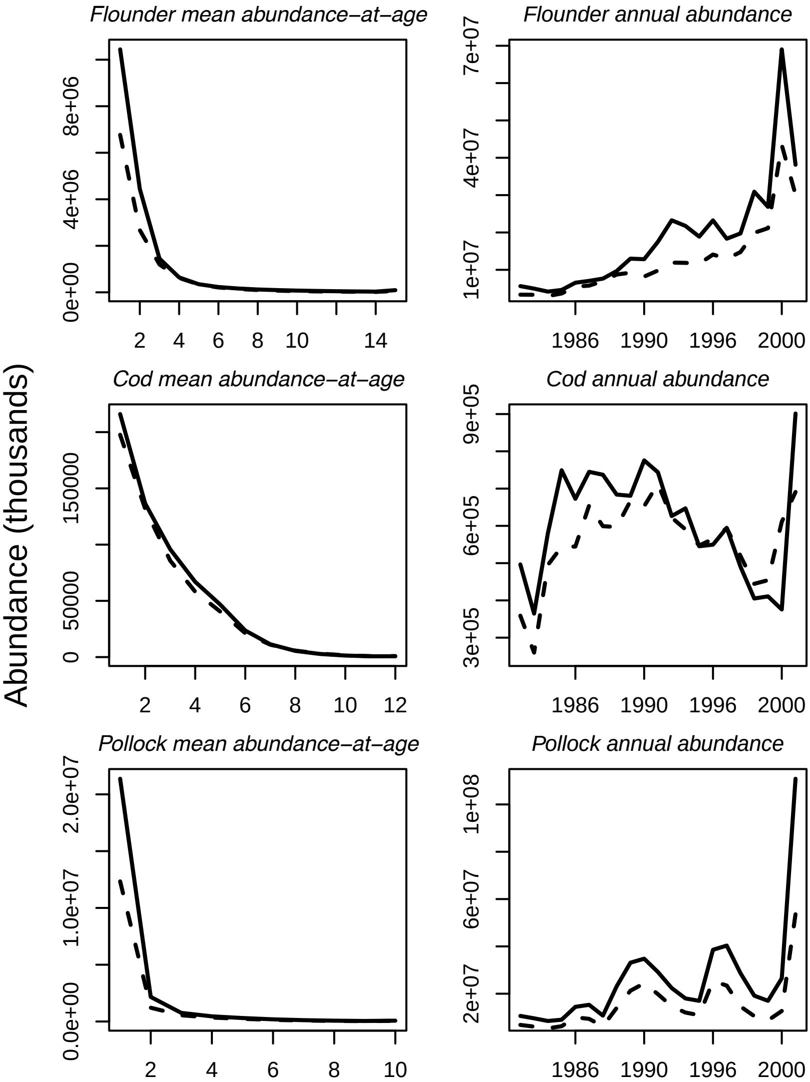

Figure 1. Changes in abundance from core model (dashed lines) to expanded model (solid lines).

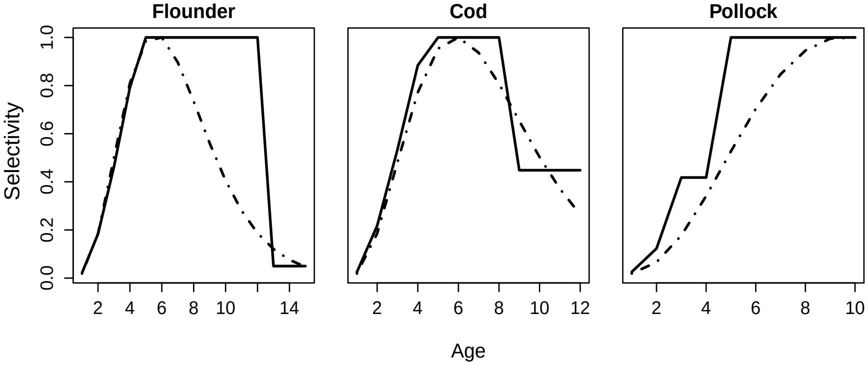

Figure 2. Changes to selectivity curves between core (dashed lines) and expanded models (solid lines).

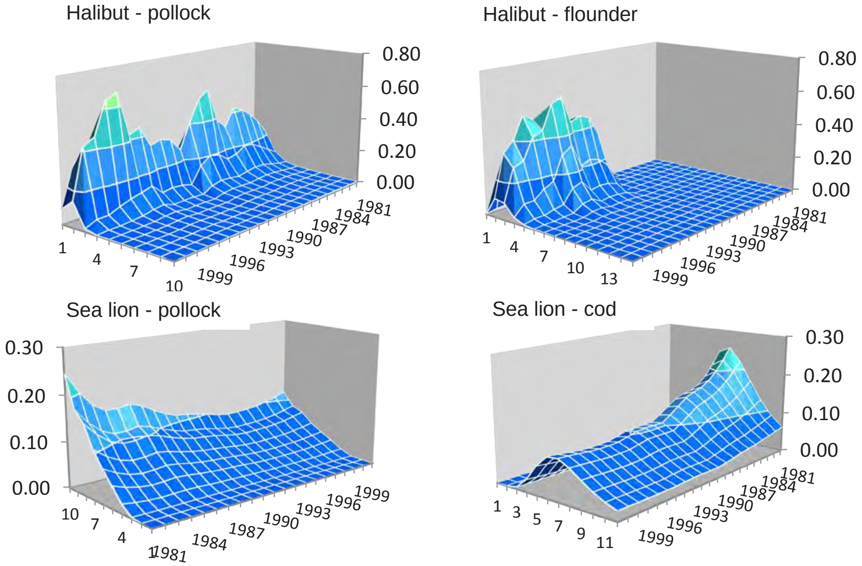

Figure 3. Predation mortality on flounder, cod, and pollock from Pacific halibut and Steller sea lions. Age of prey is given on the x-axis, instantaneous mortality on the z-axis. Note reversed age-axis for sea lion preying upon pollock. Sea lion predation on flounder not shown.

Estimated abundances for pollock and flounder increased in the expanded model (Fig. 1). These increases were generally most pronounced for younger ages and declined closer to core model values over time, although total estimated abundance remained greater for all years. Trends for cod abundance were less clear. Total estimated abundance was greater than the core model early on but fell below it in later years; the increase was primarily for ages 2-6 (Fig. 1). Selectivity curves differed from those assumed in the core model (Fig. 2). Full-recruitment fishing mortality *F* followed trends similar to the abundances in Fig. 1 (not shown).

Halibut preyed primarily on pollock ages 2-5, but shifted the heaviest predation from ages 2-3 in early model years to ages 1 and 2 over time (Fig. 3). Halibut predation on flounder occurred mainly from 1991 to 2001, and was concentrated on ages 2-5 (Fig. 3).

Sea lions consumed predominantly mid-sized pollock ages 5-7, and cod ages 5-10. Predation mortality, distinct from stomach contents as it is affected by relative abundances of predator and prey ages, was highest for oldest pollock ages, while predation mortality for cod was highest for ages 6 and 7 (Fig. 4). Sea lions fed on flounder as well, focusing on the oldest ages, but this predation was extremely minor, with all predation mortality on flounder from sea lions measuring less than 1%. All predation from sea lions dropped over time due to the decline in sea lion abundance. Pollock composed an average of 25.5% of sea lion stomach content by weight, cod an average of 20.2%, other food 53.9%, and flounder less than 1%. Younger sea lions fed more heavily on pollock (71.9% of diet by weight for age 1 sea lions), while older sea lions decreased their consumption of pollock (13.1% for age 13) and switched to cod (23.7%) and non-modeled prey (62.6%).

The addition of halibut and sea lions increased pollock total predation mortality for ages 2-5, and also for ages 8-10 (Fig. 4). Predation mortality on flounder declined, especially for early years and older ages, although the increased mortality from halibut predation on ages 2-5 was visible (Fig. 4). Cod exhibited large changes from core model trends, including a general reduction in prey linkages, decreased predation on flounder, and a shift toward younger prey ages (Fig. 5). These changes increased the relative proportion of M_0 to total mortality for flounder (Fig. 6), while the proportion for cod and pollock (Fig. 7) was roughly comparable between models.

Discussion

Hollowed et al. (2000) constructed a predation model for pollock in the Gulf of Alaska in which the modeled predators were Pacific halibut, Steller sea lions, and arrowtooth flounder. Predator abundances were set external to parameter estimation, and predation was estimated as

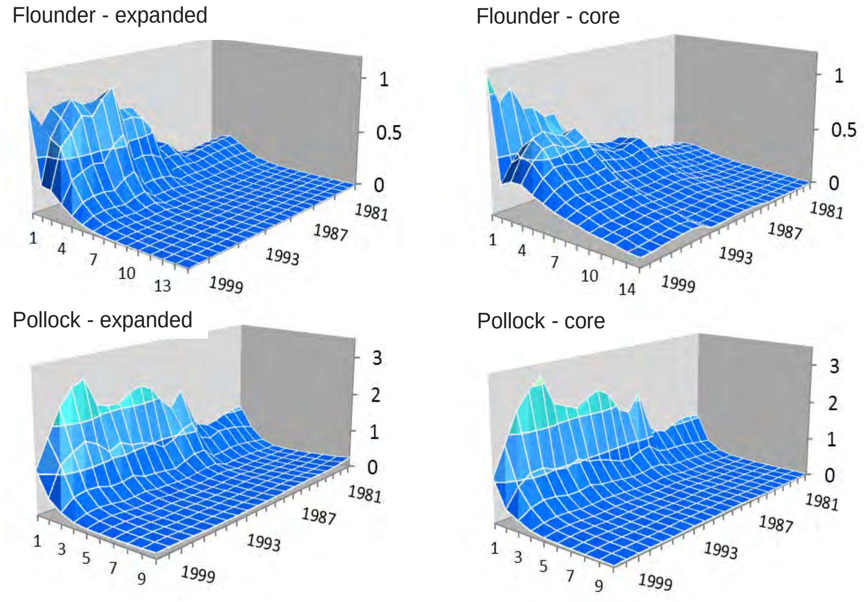

Figure 4. Total predation mortality on flounder and pollock from the core and expanded models. Age of prey is given on the x-axis, year on the y-axis, and mortality on the z-axis.

a function of an age-dependent selectivity coefficient and a catchability term specific to each predator-prey combination. Modeled years ran from 1964 to 2002; predator diet data were taken from the AFSC Resource Ecology and Ecosystem Modeling database from 1990 to 1996. In their model, sea lions consumed an average of 126,500 t of pollock in 1997, halibut an average of 52,500 t, and flounder 329,000 t. In contrast, the current work estimates total sea lion consumption of pollock in 1997 to be 50,000 t, total halibut consumption of pollock at 116,000 t, and total flounder consumption at 188,000 t. Hollowed's work also showed halibut selecting for older pollock, whereas the MSASA model placed the majority of halibut predation on pollock ages 2-5 and reduced halibut predation on pollock aged 6+. Both models displayed flounder preying on younger prey ages and placed flounder predation the highest of the three predator species in common.

The 2010 AFSC stock assessment for pollock in the Gulf of Alaska (Dorn et al. 2010) includes an Ecopath (Christensen et al. 2000) model of pollock trophic dynamics based on AFSC Resource Ecology and Ecosystem Modeling stomach-content data from 1990-1993, based

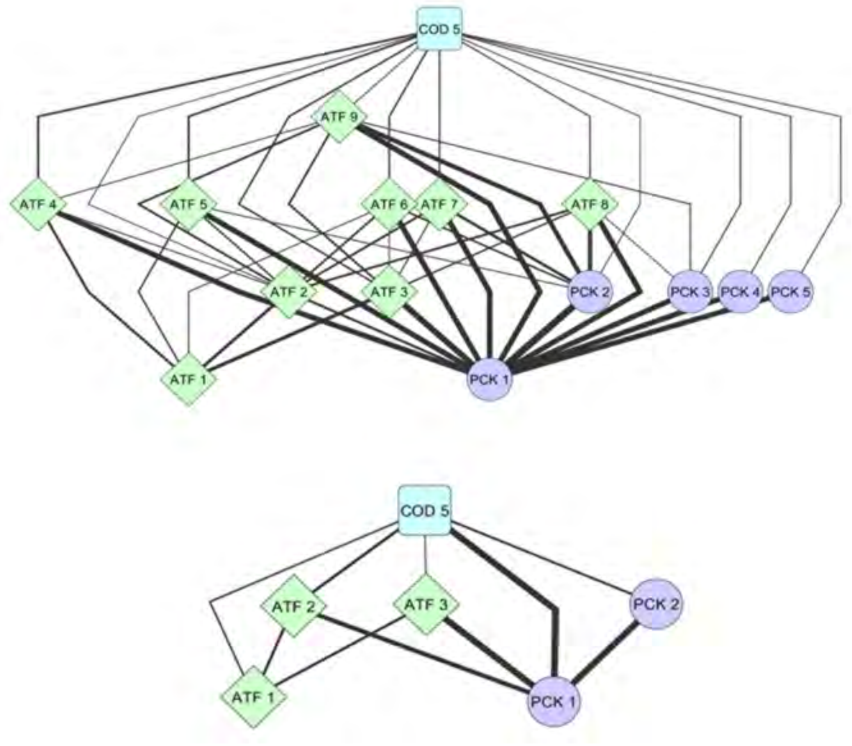

Figure 5. An example of changes in predation structure and trophic linkages for age 5 cod. Predation linkages from the core model (a) are simplified with a move toward smaller prey in the expanded model (b). Heavier lines indicate greater prey stomach-proportion by weight. Numbers refer to age. ATF = arrowtooth flounder, COD = Pacific cod, PCK = walleye pollock.

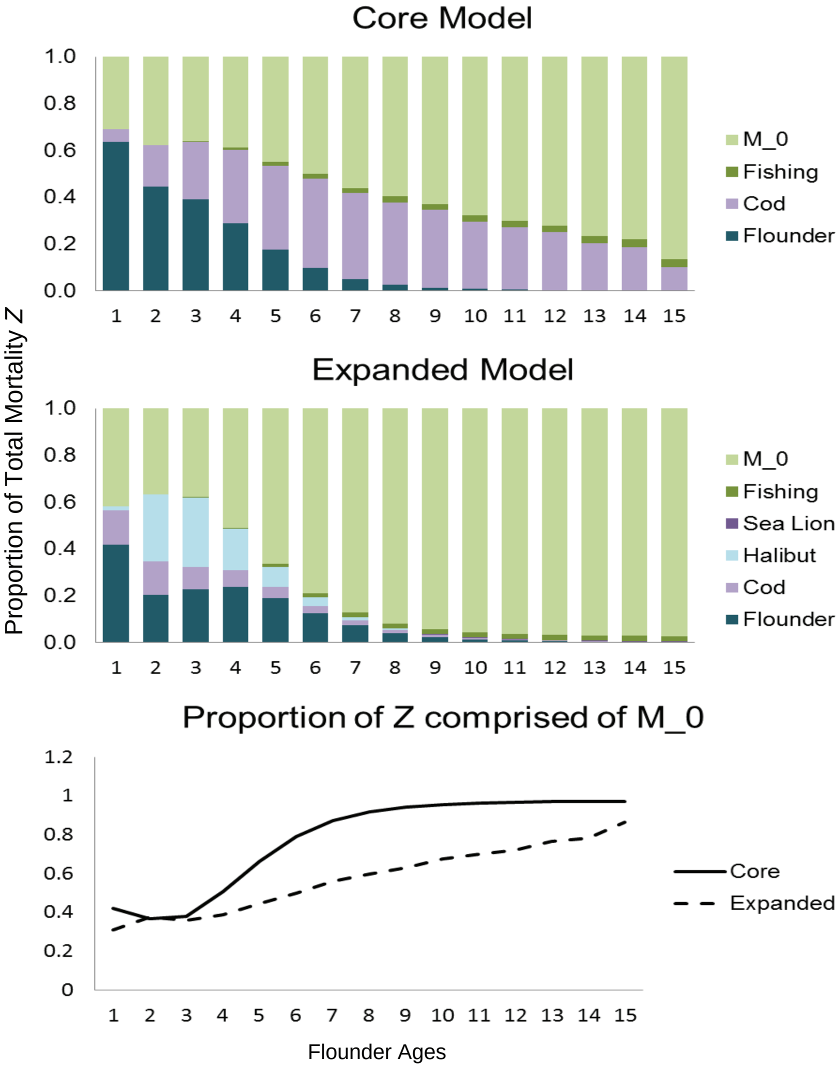

Figure 6. **Components of total instantaneous mortality Z for arrowtooth flounder by age from the core model, the expanded model, and the difference between the relative contributions of M_0 to Z between models. Values are averaged over all model years.**

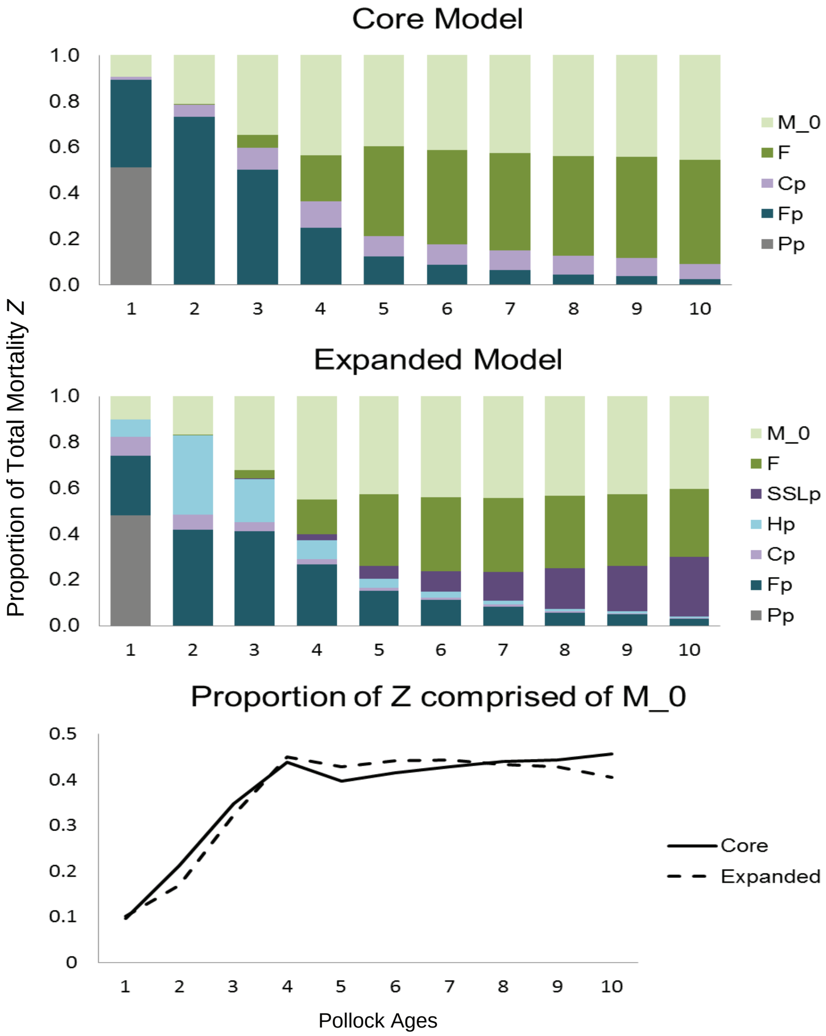

Figure 7. **Components of total instantaneous mortality Z for walleye pollock by age from the core model, the expanded model, and the difference between the relative contributions of M_0 to Z between models. Values are averaged over all model years.**

on Aydin et al. (2007). Pollock were divided into juveniles (<20 cm, corresponding to age 1 pollock in the current work), and adults (>20 cm, corresponding to ages 2-10+). The top two sources of predation mortality for juvenile pollock were arrowtooth flounder (46.8% of total predation mortality) and adult pollock (11%); the Gulf of Alaska MSASA model showed the highest predation mortality to come from adult pollock (56% of total predation mortality), followed by arrowtooth flounder (33.8%). For non-juvenile pollock (>20 cm), the Ecopath model showed the top pollock predators to be arrowtooth flounder (32.8%), Pacific halibut (22.9%), Pacific cod (16.2%), and Steller sea lions (6.2%). The MSASA model also placed arrowtooth flounder at the top of the list (45.4%) but listed Pacific cod as second (30.6%), followed by Pacific halibut (12.6%) and Steller sea lions (11.1%). (It should be noted that the relative proportions of predation mortality are not strictly comparable between the two approaches, as the Ecopath model includes a number of other predators beyond those in the MSASA model.)

The Gulf of Alaska MSASA model differs significantly from the studies discussed above by the magnitude of pollock cannibalism shown in model outputs. While pollock cannibalism is a large trophic pathway in the Bering Sea, Yang (1993) found that this accounted for only 2.5% by stomach weight in the 1990 bottom trawl survey in the Gulf of Alaska. Hollowed et al. (2000) therefore did not include cannibalism as a potential predation vector. Dorn et al. (2010) shows cannibalism on age 1 pollock, but to a smaller degree than the Gulf of Alaska MSASA model.

The preference for mid-sized prey on the part of sea lions is supported by Trites and Calkins (2008), who found mean size of pollock consumed by sea lions to be 46 cm for males, and 39.8 cm for females (ages 3-5 in the current work). This is somewhat different from Merrick and Calkins (1996), who found that for sea lions less than four years of age, 51% of the pollock consumed were under 30 cm (ages one and two), while 79% of the pollock consumed by adults were over 30 cm; the MSASA model showed all ages of sea lions feeding more heavily on fish larger than 30 cm. It is also in contrast to Frost and Lowry (1986), who found mean size of sea lion prey to correspond to age 2 pollock, regardless of predator age, a finding mirrored in the eastern Bering Sea in the mid-1990s by Calkins (1998) and a literature review by Etnier and Fowler (2005). The use of minimum/maximum limits for estimated sea lion stomach contents appears to be the best course of action given the disparities in the literature. The large influence of sea lion predation on predator-prey connections, however, requires improvements in sea lion diet assessment to adequately model such important system dynamics.

Predation mortality changes the structure of a population through the effects of cohort-specific predation. Survey indices exert similar pressure on estimated cohort structure through selectivity values. As catch and survey data are assumed to have the lowest uncertainty of the data used in model fitting, they are consequently assigned the highest weights in the objective function. Model fitting may improve catch and survey fits at the expense of predation mortality components, resulting in erroneous deductions about predation functions, and model performance is highly sensitive to different model assumptions and approaches to data weighting.

Predation models show increased prey biomass relative to single-species models (Kinzey and Punt 2009, Moustahfid et al. 2009). In an age-structured framework, M_0 is raised when that increased recruitment is not completely removed due to predation but is instead passed through the population. In this context, M_0 for species subject to heavy predation is less a realistic indicator of a physiological mortality and more an indicator of the uncertainty contained within the modeled population that has yet to be explicitly defined. M_0 for pollock did not change from the core model because the additional predation was relatively evenly distributed over all age-classes by the new predators (Fig. 7); reductions in predation on younger ages from cod and flounder were replaced by predation from halibut, and remaining increased biomass in older ages was removed by sea lion predation. Conversely, predation from older flounder was drastically reduced (Figs. 5 and 6), focusing the majority of predation pressure on ages 1-5. The disparity between predation on flounder ages was responsible for the increase in the relative proportion of M_0 to total mortality Z for older ages, and the sharp drop in selectivity values for ages 6+ (Fig. 2).

If the asymptotic progression of pollock M_0 toward zero is considered an indication of a minimal realistic predation model, further work is needed, especially as food web work such as Gaichas et al. (2010) found pollock to be fully utilized by Gulf of Alaska predators. Early experiments with unbounded sea lion size-preference parameters produced heightened predation on older pollock and cod, reducing pollock M_0 to 0.05; this assumption should be revisited along with others regarding model structure and included predators. M_0 may be asymptotic not to zero, but to some other measure of mortality indicative of a minimal necessary complexity, and is most likely different for apex species such as halibut compared with forage species such as pollock (Gaichas et al. 2010).

The MSASA structure is capable of displaying the complex population dynamics needed for fisheries management, utilizing easily accessible data. Such an approach is needed in stock assessments to improve estimates of cohort structure, develop predation-robust biological reference points, and assess the impact of commercial biomass removals. Resolution to parameter confounding can be found in more abundant stomach-content data, reducing the number of possible model solution states, as well as external analyses directed toward improved estimates of survey selectivity curves. Updating the model to include the most recent data will aid in this, as well as simulation work to evaluate the influence of data scarcity and model specification on parameter estimates. Implementing these improvements will be a significant step forward in preparing the Gulf of Alaska MSASA model for practical application to fisheries management.

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Developing Strategies for Improved Assessment and Ecosystem-Based Management of Canadian Northern Dolly Varden

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Abstract

Anadromous northern Dolly Varden char are harvested for subsistence by Inuvialuit, Gwich'in, and Inupiat mainly in tributaries west of the Mackenzie River that flow into the Arctic Ocean. Within Canada, substantial declines have been observed in two of the six known anadromous stocks. Although overharvest was thought to have contributed to these declines, the lack of recovery in some cases, despite fishery closures, coupled with observations of habitat change in spawning and overwintering areas, suggest that environmental factors may play an important role in regulating populations in these river systems. In recent years, DFO has begun to adopt an ecosystem approach and consider other factors in the assessment of fisheries. In this particular case we are advocating employing techniques for real time assessment of abundance, models of habitat use and availability, and tracking changes in other fish species that coexist with Dolly Varden to develop a more comprehensive management framework for this species. We present a quantitative approach for integrating ecosystem components into calculations of sustainable harvest for this species.

Introduction

Ecosystem-based fishery management (EBFM) or the inclusion of ecosystem considerations in fisheries management has been advocated by a number of advisory panels within the last decade (Pikitch et al. 2004, Morishita 2008). Although there are many definitions for EBFM, some of the key components include moving beyond single species management to consider interactions with other species, effects of environmental factors within the ecosystem, and human/cultural concerns/influences (Morishita 2008, Olson 2011). The Department of Fisheries and Oceans Canada is now moving toward an ecosystem-based approach to management, defined as "a broad approach to studying relationships and interactions in the ecosystem that focuses its efforts on identifying key relationships in nature, and their links to human needs and actions" (DFO 2007). It has been suggested that use of the ecosystem approach should lead to a number of changes in science support for fisheries management and fisheries management itself including: (1) a need for rule-based, risk-based management using reliable indicators; (2) broadened science advice that applies knowledge of expected stock productivity changes with population demographics, harvest, and environmental conditions; and (3) the assessment of the status and trends of non-target species (DFO 2007). This more holistic approach is particularly well suited to the northern Dolly Varden (hereafter referred to as Dolly Varden), *Salvelinus malma malma*, fishery within the Canadian Arctic for several reasons: (1) the river systems in which these populations spawn, rear, and overwinter are highly variable; (2) environmental factors are potentially important in population regulation; and (3) the systems in which these fish live are relatively simple making it more tractable to apply an ecosystem approach.

Like many other salmonids, Dolly Varden are anadromous, migrating annually between marine feeding habitat and freshwater spawning/overwintering habitat following smoltification, which typically occurs between the ages of 3 and 4 (Morrow 1980). Within Canada, there are six known anadromous northern form Dolly Varden populations (five of which have been tested and found to be genetically and/or morphologically distinct) (Reist 1989). Individuals from all of these populations feed along the Beaufort Sea coast west of the Mackenzie Delta, and spawn and overwinter in rivers along the Yukon north slope and tributaries flowing from the Richardson Mountains into the lower

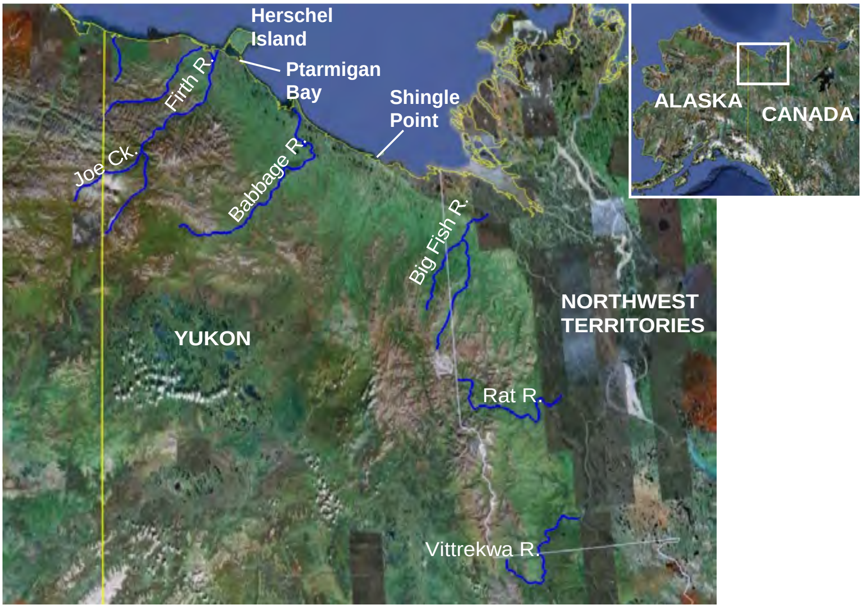

Figure 1. Map illustrating distribution of anadromous northern Dolly Varden stocks and fisheries in Canada and other place names referred to in text. Anadromous stocks occur in the Firth River, Joe Creek (tributary of the Firth), and Babbage, Big Fish, Rat, and Vittrekwa rivers. Yellow shading represents areas where fishing currently occurs.

Mackenzie River system (Fig. 1). These rivers are all uniquely characterized by having perennial groundwater flow creating year-round open water habitat believed to be essential for spawning and overwintering in this species (Krueger et al. 1999, Mochnacz et al. 2010). These areas provide adequate oxygen and suitable temperatures for adult and juvenile survival, and egg incubation during the winter months (Stewart et al. 2010). Because rivers of this size typically freeze to the bottom in winter, this type of overwintering habitat is believed to be limited (Craig and McCart 1974, Mochnacz et al. 2010).

Dolly Varden is an important cultural and nutritional resource for the Inuvialuit and Gwich'in people of northern Canada, and for the Inupiat of Alaska. At present the species is mainly fished for subsistence during the summer months using gillnets in marine coastal locations in both Canadian and Alaska waters, with approximately 25% of the Alaska harvest composed of fish of Canadian origin (Krueger 1999).

They are also harvested by gillnets in the Mackenzie River and its tributaries on their return migration to spawning/overwintering areas and angled recreationally on the Firth River in Ivvavik National Park (Fig. 1). Historically, Dolly Varden were fished using seine nets at the spawning and overwintering areas in tributaries of the more accessible rivers such as the Rat and Big Fish; however, this practice was stopped in the 1980s due to community concerns about declining populations.

Population declines have been documented in the two main harvested rivers, the Big Fish (1980s) (Sandstrom and Harwood 2002) and the Rat (2004) (Sandstrom et al. 2009). In both cases declines were at least partly linked to possible overharvesting. In the Big Fish River, there has been little recovery of the stock despite closure of the river to all fishing since 1987. Anecdotal information suggests that water levels have decreased in this river system, which may be constraining recovery (Sandstrom and Harwood 2002); however, continued coastal fishing may also be a factor. Following closure of the fishery there was an abundance recovery in the Rat River stock; however, the increase was mainly due to the presence of a single strong year class, thus the stock may still be unstable (Sandstrom et al. 2009, Roux et al. 2012). In both rivers it is unclear if habitat change, changes in fishing activity, or a combination of these two factors has influenced recovery. Unfortunately there is limited long-term stock assessment and habitat time series information for most Dolly Varden populations in Canada. An exception is the Rat River where a population assessment program has been in place over the last 15 years and a reasonable time series (1995-2010) exists for catch rates, abundance, and population dynamics (Sandstrom et al. 2009, Harwood et al. 2009, DFO unpubl. data).

Northern form Dolly Varden has recently been listed as “Special Concern” by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2010) and an Integrated Fisheries Management Plan (IFMP) was recently completed for all anadromous Dolly Varden stocks in Canada (DFO 2010). These two initiatives, along with a shift within DFO toward ecosystem-based science and management (DFO 2007), have enabled us to increase assessment efforts and begin exploring ways of enhancing our approach through the inclusion of other ecosystem components in our program. In this paper we discuss the approach used in assessment and management of Dolly Varden in Canada and our knowledge to date based on this approach. We then present some of the key changes we are implementing to develop an ecosystem-based fisheries management (EBFM) strategy for this species. Finally, we present a framework to integrate habitat quantity/quality into calculations of sustainable harvest for this species.

Stock assessment and management of Dolly Varden in Canada

Like many fisheries, assessment of Dolly Varden has traditionally been conducted using a single species approach, utilizing information on abundance, catch rates, and biological indicators to determine stock status and estimate sustainable harvest. Population abundance estimates have typically been made on a periodic basis through mark-recapture methods. Fish are seined and marked on the spawning/overwintering grounds during the fall when all life history stages are usually present and in most cases are recaptured the following year through seining at the overwintering site and/or through the subsistence gillnet fishery. Catch-per-unit-effort (CPUE) and biological information such as size, age, sex, and maturity are mainly collected through the annual Rat River harvest monitoring program involving local fishers and through the fall seining and marking at the overwintering sites. Information on harvest levels is collected separately through harvest surveys in key communities and at coastal fishing locations. Further details on the fall seining and mark-recapture protocols can be found in Sandstrom et al. (2009) while details of the harvest-monitoring program are provided in Harwood et al. (2009).

Since the establishment of aboriginal land claims in this region of the Western Arctic (Inuvialuit Final Agreement [1984], Gwich'in Comprehensive Land Claim Agreement [1992]), Dolly Varden fisheries have been co-managed through adaptive community-based fishing plans. Community needs, community/harvester's observations, and scientific data are taken into consideration in recommending harvest guidelines and other measures for stock conservation. In the absence of any stock- or species-specific safe harvest rates that could be adopted, in recent years a conservative safe removal rate of 5% (based on studies of slower-growing arctic char, *Salvelinus alpinus*; Johnson 1980) has been applied to the most current population estimate for the fishable component of a given stock (fish >300 mm, Sandstrom et al. 2009) and thus harvest guidelines have been set accordingly on an annual (or sometimes less frequent) basis.

Knowledge of Dolly Varden population dynamics based on standard assessment data

The Rat River is the only Dolly Varden population where there has been consistent monitoring and where there is reasonable time series information on abundance, harvest and demography (1995 to present). The available information from this stock suggests that Dolly Varden, like many other fish, exhibit high annual variability in abundance, which may be cyclical (e.g., Myers et al. 1997, Schindler et al. 2006, Zorn and Nuhfer 2007). For example, CPUE data (which we later show is correlated with estimated abundance) from the harvest-monitoring program over the last 15 years, has varied substantially (Harwood et al. 2009, K. Howland and C. Gallagher unpubl. data). A much shorter time series from recent fall seining at the overwintering/spawning site on the Rat River also suggests a high annual variability in the proportion of three- to four-year-olds available for capture in this gear (K. Howland and C. Gallagher unpubl. data). A variety of factors including exploitation and environmental variation may contribute to such fluctuations in population abundance.

Qualitative data suggest that flow regimes in rivers utilized by Dolly Varden can be highly variable both within and between years (Amanda Joynt [DFO Inuvik], N. Mochnacz, and S. Sandstrom, unpubl. data), and thus may influence the carrying capacity of habitat. Habitat that is limiting at key life stages (e.g., spawning, rearing, overwintering) can influence the overall productivity of salmonid populations (e.g., Harvey et al. 2006, Ebersole et al. 2009) and may explain some of the observed year-to-year variability in Dolly Varden CPUE/abundance.

Local traditional knowledge and anecdotal researcher observations also suggest that in some systems there has been an overall decline in water levels from historic levels (e.g., Big Fish River; DFO 2002, Sandstrom and Harwood 2002) that may be related to underlying changes in climate and/or geologic activity. A significant reduction in discharge is likely to result in a decrease to the carrying capacity of available habitat in a river system (Rosenfeld 2003, Harvey et al. 2005, Ebersole et al. 2009). Large-scale ecological regime shifts can alter the overall productivity of populations (i.e., maximum population size), making them more vulnerable to stochastic or anthropogenic impacts and reducing their ability to recover to historical levels (e.g., see review by Folke et al. 2004).

Although changes in habitat availability and quality may play a role in regulating Dolly Varden populations, there are also data to suggest that overharvest has contributed to declines in abundance and shifts in the population dynamics of Dolly Varden in the Rat and Big Fish rivers. In the Rat River there were particularly high harvest rates in the late 1990s (30-32%, six times higher than the current conservative rate of 5%) that resulted from non-compliance with harvest recommendations (Fig. 2a). This period of high harvest was followed by a period of population decline five to six years (one generation) later (Fig. 2b) (Sandstrom et al. 2009, Harwood et al. 2009, Roux et al. 2012). During the same time period there was also a substantial decline in the proportion of spawners both in the gillnet fishery (harvest monitoring) and based on samples collected by seine at the spawning/overwintering site (Sandstrom et al. 2009, Roux et al. 2012). A similar pattern of population collapse

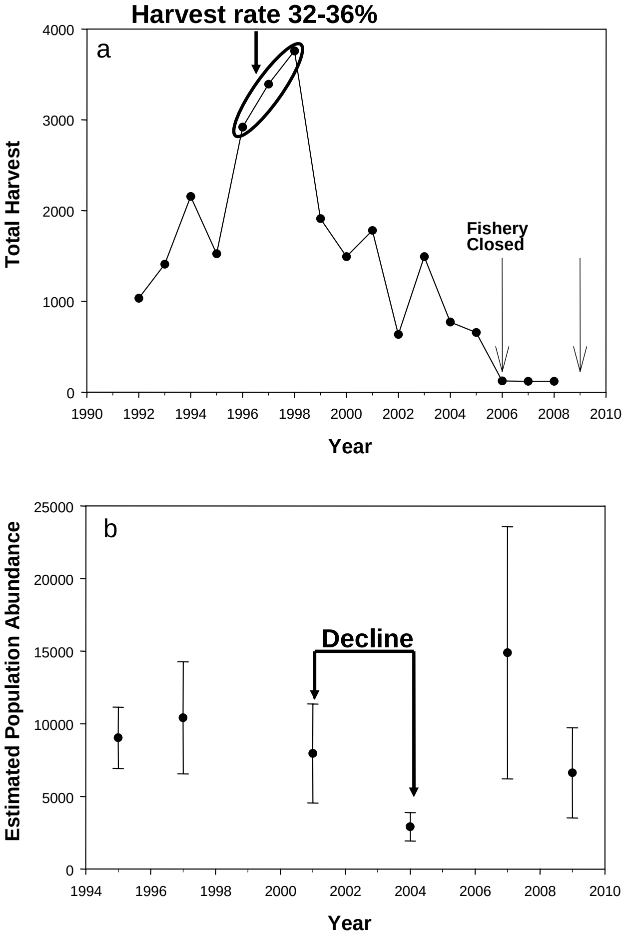

Figure 2. **Total harvest (a) and mark-recapture population abundance estimates (b) by year for Dolly Varden from the Rat River. Harvest rates were calculated by taking total annual harvest divided by current (or previous) year mark-recapture abundance estimate (after Roux et al. 2012). Note: Harvests for 2006-2008 represent a limited number of fish that were collected by harvest monitors for scientific sampling (n = 120 fish per year) during the fishery closure. Total harvest for 2009 and 2010 were not available. Population abundance data for 1995-2007 from Sandstrom et al. 2009, data for 2008-2010 DFO unpubl. See Roux et al. 2012 and Harwood et al. 2009 for details on annual harvest information.**

following a period of exceptionally high harvest was noted for the Big Fish River population (Sandstrom and Harwood 2002, Gallagher et al. 2011). What remains unclear in the case of both populations are the mechanisms of apparent interaction between exploitation and environmental changes in explaining shifts in population abundance over time.

Enhanced assessment and the ecosystem approach

With the knowledge gained through past assessments, in particular the long-term monitoring of the Rat River stock, DFO is positioned to build on the existing data and move toward an ecosystem-based approach to the assessment and management of Dolly Varden. Based on existing knowledge of Dolly Varden and taking an ecosystem-oriented perspective we have identified several areas where the core assessment program could be enhanced: (1) there is a need for consideration of potential annual variability in population abundance when determining sustainable harvest levels; (2) there is a need for more regular population abundance estimates or some proxy for abundance; (3) there is a need for information on the mixed stock composition and improved monitoring of coastal fisheries along with more accurate harvest reporting; (4) interactions and effects of the fishery on other species and vice-versa need to be considered; (5) although the potential effects of habitat changes on recruitment and population abundance have been acknowledged (e.g., Sandstrom and Harwood 2002), there is a need to quantify and incorporate these factors into calculations of sustainable harvest.

We have begun to address some of the above identified needs by implementing a number of changes in the current Dolly Varden assessment program; these can be divided into strategies for more effective assessment and strategies to support ecosystem based fisheries management. Briefly, strategies for more effective assessment include the collection of current year estimates using dual frequency identification sonar (DIDSON) technology, development of measurable key indicators for tracking changes in recruitment, and fishable stock abundance and expansion of coastal monitoring programs to improve our knowledge of mixed stock composition and harvest levels. Strategies to support ecosystem based management include: the collection of data on nontarget species, collection of habitat data and development of outputs and measurable habitat indicators from a two dimensional hydrodynamic fish habitat model, and linking stock assessment and habitat information in the estimation of sustainable harvest. Each of these areas is described in greater detail below.

Strategies for more effective stock assessment

Use of DIDSON technology to provide current year information on stock abundance and the development of stock abundance indicators

DIDSON technology may be used for determining abundance of individual Dolly Varden stocks in order to address the problems of time lag and other drawbacks associated with mark-recapture methods (see Seber 1982 for a review of assumptions associated with mark-recapture). DIDSON is a fixed-location, side looking sonar technique that has the capability of recording high-resolution video of fish as they pass through a cone-shaped, horizontal beam and is particularly useful for monitoring fish behavior and movement in fast flow or turbid conditions (Enzenhofer et al. 1998, Maxwell and Gove 2007, Burwen et al. 2010). This technique has been successfully used to provide real time enumeration of Dolly Varden stocks in Alaska (e.g., Hulahula River, Osborne and Melegari 2008) and is regularly used to enumerate other anadromous species such as pacific salmon (e.g., see Burwen et al. 2010). We pilot tested this technology in 2009 and conducted the first enumeration on the Big Fish River in August-September 2010.

Although there are many positive aspects to the use of DIDSON for providing current year stock size estimates, it is expensive and time consuming, limiting its use on an annual basis in the case of Canadian Dolly Varden stocks. Thus we see this method as being more useful in the context of retrospective analyses using time series information, whereby potential indices of stock abundance that can be more easily measured on an annual basis could be examined in relation to mark-recapture or DIDSON estimates of counts. For example, CPUE from harvest monitoring on the Rat River is correlated with mark-recapture abundance estimates for this stock (Fig. 3) (Harwood et al. 2009, Sandstrom et al. 2009, DFO unpubl. data), although it should be noted that use of CPUE data from coastal fisheries is complicated by mixed stock presence in these locations. However, CPUE data from fall seining surveys within the rivers may serve as another possible index of stock abundance. Future periodic counts and abundance estimates utilizing DIDSON and/or mark-recapture methods can provide a tool for verifying/ground-truthing the relationship with stock size indicators, which can then serve as a proxy for stock size in annual calculations of sustainable harvest.

Expansion of coastal monitoring programs to improve knowledge of mixed stock composition and harvest levels

Several initiatives are currently in progress to address questions regarding mixed stock composition in coastal fishing locations. These include the use of genetics, otolith microchemistry, and tagging. These initia-

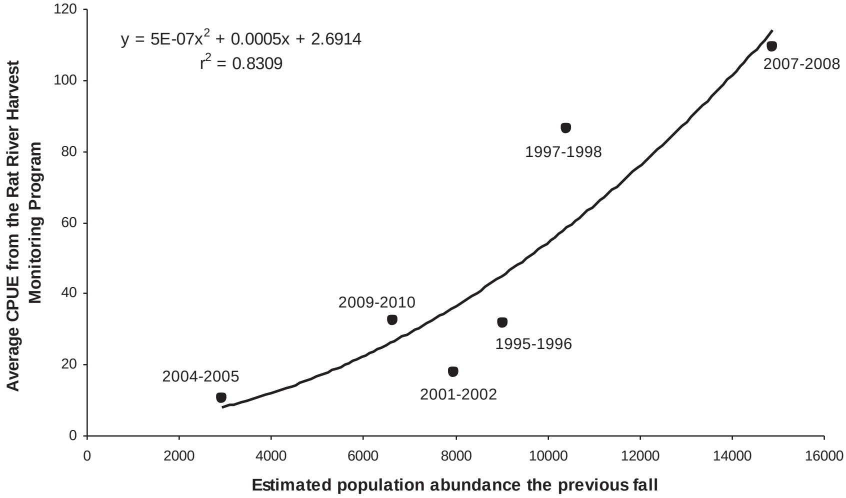

Figure 3. Regression of average CPUE from the Rat River harvest-monitoring program on mark-recapture population abundance estimates from the previous year. Data for 1995-2007 from Harwood et al. 2009 and Sandstrom et al. 2009; data for 2008-2010 are DFO unpubl.

tives are being further enabled through the expansion of the coastal fishery monitoring program in terms of both data collection and geographic scope. Until 2010 this program was focused at the Shingle Point area (Fig. 1) and involved the use of logbooks provided to fishermen to collect harvest numbers only. As of 2011 there has been the addition of direct collection of CPUE information and biological sub-sampling of the catch by trained monitors at the Shingle Point area. Monitors were trained by DFO biologists who were on site throughout the fishery in 2011. In 2011 training of monitors and collection of CPUE/biological sampling was also initiated at two other Canadian coastal fishing locations, Herschel Island and Ptarmigan Bay (Fig. 1). Genetic analyses to examine mixed stock composition are in progress (R. Bajno [DFO Winnipeg] and J. Reist, unpubl. data) and samples collected during the 2011 coastal fishery will contribute an updated sample to examine current stock composition. In 2010, Dolly Varden were tagged in the Rat, Big Fish, and Babbage rivers (C. Gallagher, K. Howland, and S. Sandstrom, unpubl. data). These tagged fish were available for recapture in the 2011 fishery. Tag recaptures will allow us to estimate proportions of Dolly Varden captured from each of these three main contributing rivers. Otoliths collected from Dolly Varden caught in the 2011 coastal fishery will contribute to studies examining differences in otolith microchemistry of Dolly Varden among river systems and the potential for using this tool to examine stock composition in mixed coastal fisheries (T. Loewen, N. Halden [Geological Sciences, University of Manitoba], and J. Reist, unpubl. data). The combined approaches of genetics, otolith microchemistry, and tagging are expected to complement each other in addressing questions regarding the mixed stock composition of Dolly Varden in Canadian coastal areas.

Strategies to support ecosystem based fisheries management

Collection of information on nontarget species

A key aspect of EBFM and DFO's emerging bycatch policy (currently under development) is the consideration of multispecies interactions; the potential impacts of intended removals of one or more species and unintended bycatch on other species in the ecosystem (Perry 1999). We have included several new components to our monitoring and sampling programs to increase our knowledge of nontarget species and our understanding of the role of Dolly Varden within the riverine and arctic marine ecosystems. Collection of daily catch and effort for all bycatch species will be included in the sampling protocols for the ongoing harvest-sampling program at the Rat River. Coastal fishery monitoring programs have recently been expanded (as of 2010-2011) to include the collection of the following information for bycatch species in addition to Dolly Varden: catch and effort, basic demographic data, and tissues that will contribute to ongoing food web/diet studies in the Beaufort coastal ecosystem (L. Loseto et al., DFO Winnipeg, unpubl. data). In the upper reaches of rivers, Dolly Varden coexist mainly with arctic grayling (*Thymallus arcticus*), which also are frequently captured during fall seining surveys at Dolly Varden spawning/overwintering areas. We have recently begun to collect information on numbers/proportions of arctic grayling captured during fall seining and when possible take a subsample for more detailed demographic and diet information. Tracking temporal changes in demographics and/or catch rates of nontarget species may provide additional insight as to the extent to which harvest-independent factors regulate Dolly Varden populations, while information on diet, food web, and species composition should provide a better understanding of potential species interactions, and effects of the fishery and environmental change on nontarget species.

Habitat information, habitat modeling, and identification of key habitat indicators

Although anecdotal information suggests that habitat availability may play a role in regulating Dolly Varden populations, detailed quantitative information on habitat use and availability are lacking as are time series data on hydrological conditions (i.e., changing water levels). Over the last few years we have begun to quantitatively document Dolly Varden habitat use/availability and baseline habitat conditions for key life history stages (i.e., eggs, juveniles, adult) in major spawning/overwintering rivers. This information is being used to identify which habitats are critical for survival (e.g., rearing habitat, perennial groundwater) and estimate the carrying capacity of habitat within selected study reaches using the two-dimensional finite element hydrodynamic and fish habitat model River2D (www.river2D.ca, Steffler and Blackburn 2002). This model can be used to predict carrying capacity based on different water levels (i.e., discharge).

Three components are required to develop outputs from this hydrologic fish habitat model: (1) survey data on the physical habitat, in particular velocity, substrate, and depth, which are used to calibrate the model; (2) information on habitat availability and use, which are used to develop a habitat suitability index (see methods in Mochnacz et al. 2009); and (3) information on discharge (flow) at various water levels (preferably at high, medium, and low levels) to develop a rating curve. Using this two-dimensional model we can estimate optimum discharge ranges for various life history stages. Once these ranges are established for a given life stage (e.g., spawning) we can then investigate if water levels within these ranges are related to population productivity.

To achieve the above goals the River2D model is calibrated with hydrometric data surveyed for open water conditions for each study reach (further details on model calibration available from authors). Fig. 4 shows an example that compares the calibrated water surface profile and the field or the measured water surface for the Fish Hole Creek reach of the Babbage River. Fig. 4 clearly shows a good agreement between the measured and the predicted values. A statistical regression analysis was also used as shown in Fig. 5 to evaluate the difference between the measured and predicted values. Fig. 5 shows a high correlation between the measured and the predicted values, which further confirms the high accuracy of the model's results.

An example of the model output for the Fish Hole Creek reach of the Babbage River utilized by spawners demonstrates a good correspondence between the model's prediction of spawning habitat quality (based on depth, velocity, and flow), and actual habitat use and distribution of high versus low quality habitat; i.e., areas predicted to be of higher quality correspond to areas where redds were observed, while areas predicted to be of poorer quality correspond with the deeper pool in this reach of the river (Fig. 6). This ground-truthing exercise together with results from the above-described calibration curves (i.e., observed vs. expected) confirms that the habitat model is functioning reasonably well.

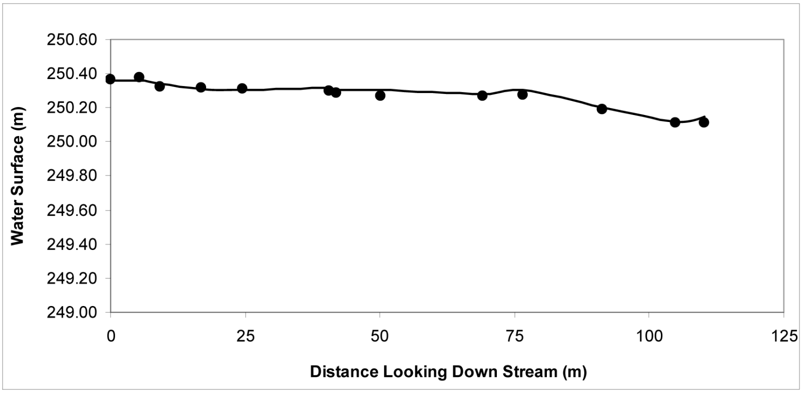

Figure 4. Comparison of measured (solid circle) and predicted (line) water surface elevations for open channel condition at Fish Hole Creek reach of the Babbage River.

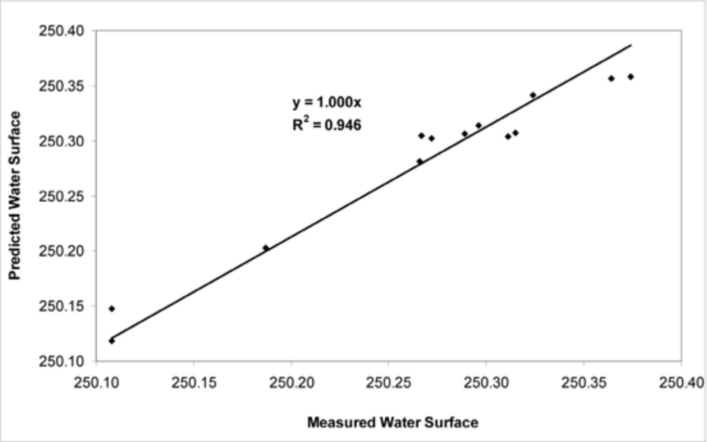

Figure 5. A linear regression or trend comparison of the measured and predicted water surface elevations at the Fish Hole Creek reach of the Babbage River

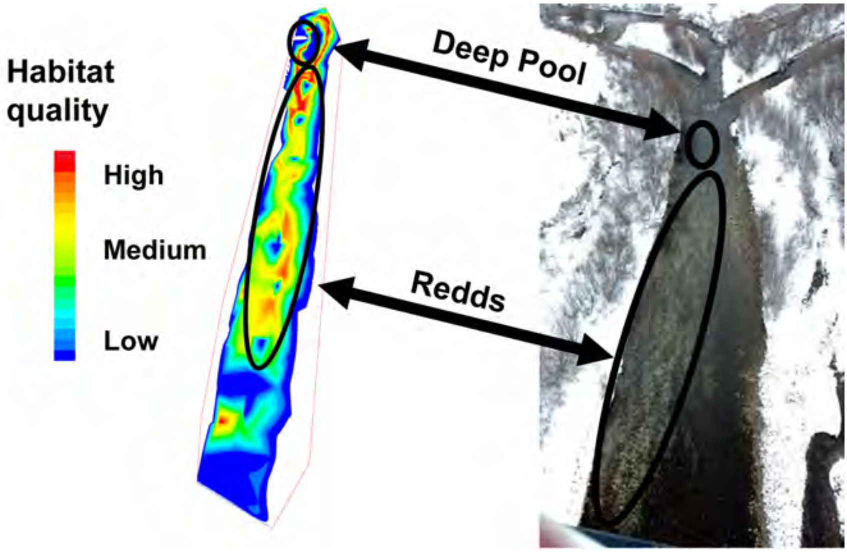

Figure 6. **Two-dimensional hydrodynamic fish habitat model output, ranking quality of spawning habitat for a reach of the Babbage River photographically depicted on the right. Habitat quality was based on depth velocity and flow characteristics. Red to yellow areas are identified as high to medium quality habitat while blue to green areas are identified as low to medium quality habitat. Light stippled areas in the photograph were confirmed to be spawning redds, while the darker encircled area at the top of the photograph was confirmed to be a deeper pool.**

This habitat model can be used to examine the relationship between habitat availability/quality and indices of recruitment at various life history stages by examining water levels over time. A strong correlation between regional habitat/environmental variables and fish recruitment and abundance has been demonstrated in a variety of freshwater and anadromous species (e.g., common roach, *Rutilus rutilus*; brook trout, *Salvelinus fontinalis*; brown trout, *Salmo trutta*; Atlantic salmon, *Salmo salar*; Jensen and Johnsen 1999, Grenouillet et al. 2001, Zorn and Nuhfer 2007). Potential indices of recruitment that could be considered in the case of Dolly Varden include spawner counts/proportions, redd counts, and possibly numbers/proportions of juveniles captured through fall seining.

Incorporating ecosystem components into evaluations of sustainable harvest

We suggest an initial method of incorporating ecosystem variables in the management of this fishery could be the use of the traffic light approach (Caddy 2002). This involves assigning predefined levels representing the state of stock to key indicators (e.g., green, yellow, red; ranging from a safe or acceptable level to a dangerous or unacceptable level). A variety of indicators including fisheries-based, habitat/environment-based, socioeconomic-based, or traditional ecological knowledge–based can be included using this approach (Seijo and Caddy 2000). Typically a set of management rules or responses is set up in advance such that harvest levels can be progressively reduced as an increasing proportion of the indicator variables move from green to red (Caddy 2000). In the case of the Dolly Varden fishery, water levels or other habitat indicators is one of several indices that would be used in combination to determine the state of the stock and adjust harvest levels accordingly.

As the above-described habitat model is refined and time series information on water levels and recruitment are developed, we can consider a more quantitative approach to incorporating habitat information into evaluations of sustainable harvest levels. Based on existing literature (e.g., Jensen and Johnsen 1999, Grenouillet et al. 2001, Zorn and Nuhfer 2007) and the examples presented above, we expect to see a relationship between habitat and recruitment. Our hypothesis is that recruitment will increase as the quantity and quality of habitat increase to some point where carrying capacity is attained through density-dependent factors or based on the morphology of the river. Once this level is reached, recruitment is expected to plateau and possibly even decline (e.g., high water flow/levels) (Fig. 7a). Similarly future sustainable harvest can be expected to increase as recruitment increases up to some maximum level (Fig. 7b). Given that sustainable harvest is related to recruitment and that recruitment is in turn affected by variability in habitat quality/quantity, we should theoretically be able to apply a habitat factor to annual calculations of sustainable harvest (Fig. 7).

In the simplest model for calculating sustainable harvest, we have an estimate of stock size (either based on indicators or more direct estimates) to which we apply a set harvest rate. The calculated sustainable harvest for a given stock could be adjusted up or down annually by applying a habitat factor based on prior knowledge of conditions affecting the particular year classes entering the fishery as follows:

Sustainable Harvest = (Estimated Stock Size)(Harvest Rate)(Habitat Factor)

Inclusion of habitat data will provide greater accuracy in projections of sustainable harvest and remove some of the uncertainty associated with local environmental variability. The incorporation of more complex

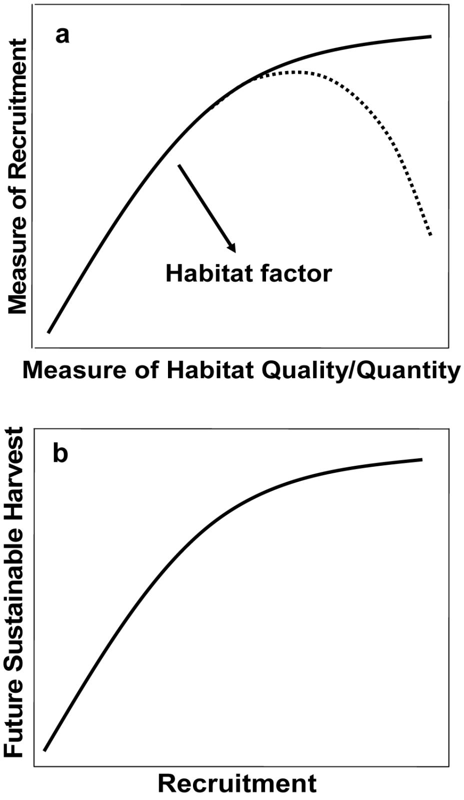

Figure 7. Schematic illustration of the potential relationships between (a) habitat quality/quantity and recruitment, and (b) recruitment and future sustainable harvest of Dolly Varden.

models to determine stock size, sustainable harvest, and potential carrying capacity of rivers (e.g., age structured VPA, surplus production models, area-per-individual habitat models), could be considered in the future if sufficient data become available.

Conclusions

Dolly Varden utilize highly variable environments for key aspects of their life history (spawning, rearing, and overwintering). Populations are highly variable and may be cyclical. Population variability is most likely related to variation in recruitment and survival at early life history stages. Thus there is a need for ongoing, consistent long-term monitoring data (time series) to identify suitable key indicators and understand the combined effects of habitat availability and harvest on recruitment and abundance. Use of combined information from key fisheries-based and habitat-based indicators should lead to better advice on sustainable harvest.

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Management of Ecosystem Effects, Potential and Realized, in Emerging Arctic Fisheries in South Baffin Island

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Abstract

While many agencies have indicated that fisheries must be managed within the context of the ecosystem rather than solely on sustainability of individual stocks, there are few examples of scientific study and management developing together to support an ecosystem-based fisheries management approach. Canada has developed a sustainability checklist for fisheries intended to address both the stock-specific approach and the ecosystem approach. We briefly describe the framework developed by Canada and its specific application to emerging fisheries within the Cumberland Sound region of Baffin Island. The application of EBFM in Cumberland Sound has brought into focus potential conflicts between industry, economic development agencies, the Nunavut Wildlife Management Board of the Nunavut aboriginal land claim, and the Inuit community of Pangnirtung. We outline the issues for development of commercial fisheries for Greenland halibut (*Reinhardtius hippoglossoides*) and arctic char (*Salvelinus alpinus*) against ecosystem concerns for other taxa such as marine mammals and the Greenland shark (*Somniosus microcephalus*). The objectives are (1) to identify the potential and realized ecosystem effects of Cumberland Sound fisheries by reviewing and summarizing available information on key components; and (2) to demonstrate how decision analysis may be used to manage ecosystem effects. Decision analysis is proposed as an EBFM tool to consider fishery economic, target species, and ecosystem sustainability. We conclude that arctic char fisheries have relatively few ecosystem effects while the fisheries for Greenland halibut have both realized effects on Greenland shark and potential effects on marine mammals.

Introduction

The scientific fisheries assessment in Canada and elsewhere has proceeded with an emphasis on the interaction of fish stock and fishery with relatively little reference to the overall ecosystem. This approach has been effective in managing many stocks, especially where other human activities in the system have been minimal. For example, Canada used a basic cohort analysis model (Pope 1972) and yield per recruit modeling (Hilborn and Walters 1992) to restore the northern Atlantic cod (*Gadus morhua*) stock to health in the late 1970s (NAFO area 2 J3KL). However, as human populations have increased so have the demands for various types of management on usage of aquatic systems such as control of shipping, industrial applications, protection of endangered species, and control of invasive species, leading to increased employment of an ecosystem approach to management. Debate persists on how to get to the goal of fishery and ecosystem sustainability (Hilborn 2007). Marine ecologists have predicted that fisheries are unsustainable and advocate using closed areas to protect aquatic ecosystems, while fisheries and many other scientists see a more complex picture with many failed fisheries but also numerous successes (Hilborn 2007). In our view the challenge is to merge the two approaches—using the strength and depth of analytical power of fisheries assessment within a framework that can account for the ecosystem effects of fisheries.

In fish communities no one factor operates in isolation and components of the ecosystem respond differently to each individual factor (FAO 2009). Ecosystem-based fisheries management (EBFM) has the following characteristics: holistic, risk-adverse, and adaptive; maintain an "old growth" structure in fish populations; maintain the natural spatial structure of fish stocks; maintain seafloor habitats; maintain resilient ecosystems; maintain critical food-web connections; adapt to short and long-term ecosystem changes; account for evolutionary changes caused by fishing; include the actions of humans and their social and economic systems in all ecological equations (Francis et al. 2007). Recently, more focus has been paid to the complex effects of fisheries in the ecosystem (H. Browman, Institute of Marine Research, Bergen, Norway, 2010, pers comm.; DFO 2009). Fisheries have direct ecosystem effects through incidental harvest of other species (bycatch) and indirect effects through the trophic structure of the system. As well, for scientific groups providing advice to fishery managers there is another dimension—that of realized and potential ecosystem effects of fishing. The difference is between advising on what is happening versus what might happen.

Realized effects are those that are clearly documented. For example with adequate observer coverage or well coordinated dockside monitoring, the bycatch taken by the fishery may be documented. Other effects are more on the potential basis where it is impossible or possibly permanently destructive to collect data. For example, there may be a potential for marine mammals to become entangled with gillnets. The catastrophic potential of this event in the Arctic (where marine mammal stocks are too fragile to risk the potential for death [DFO 2008a]) precludes actually developing an experimental approach to evaluate the likelihood of occurrence and quantify the extent of ecosystem impacts. Instead, one must model the possibility based on untested assumptions and available information. Regardless, the precautionary approach to fisheries management would dictate that both realized and potential effects be considered in developing management actions for the fishery.

To facilitate the integration of the ecosystem and precautionary approach into fisheries assessment in Canada, the Department of Fisheries and Oceans (DFO) has developed the Sustainable Fisheries Framework (SFF) and Fisheries Renewal scheme (Fig. 1). The SFF summarizes DFO's approach to EBFM, which considers the system of stock and fishery as the center and analyzes the impacts of fishing on ecosystem components as well as the impacts of the state of the ecosystem on fisheries through its approach to integrated ocean management (Fig. 1). The SFF serves to develop and articulate a series of policies to consider the effects of fisheries on sensitive benthic areas, forage fish, and bycatch species (i.e., ecosystem impacts) as well as the precautionary approach policy for management of harvest rate. The sustainability checklist is a central tool in SFF and fisheries renewal that is being used to guide and coordinate the planning process for science, fisheries management and conservation and protection (Bouffard 2010, Fig. 1). The checklist provides a snapshot of the scientific information available for assessment including ecosystem factors such as bycatch, knowledge of forage species, and basic ecosystem components. A gap analysis is done to guide the allocation of future resources to reduce the uncertainty in the system. Several aspects of Canadian policy development surrounding the SFF and Fisheries Renewal scheme are highly relevant to fishery developments in the north.

Cumberland Sound, Baffin Island, Canada, provides an ideal opportunity to develop EBFM because there are newly emerging fisheries in the region and there are not too many other factors such as pollution to obscure the ecosystem impacts of increased fishing. The principal target species are arctic char *(Salvelinus alpinus*) and Greenland halibut (turbot; *Reinhardtius hippoglossoides*). Fisheries are expected to increase with the development of a full harbor by the Department of Fisheries and Oceans. This is the first development of its kind in the Canadian Arctic. One of the main economic objectives in developing the harbor

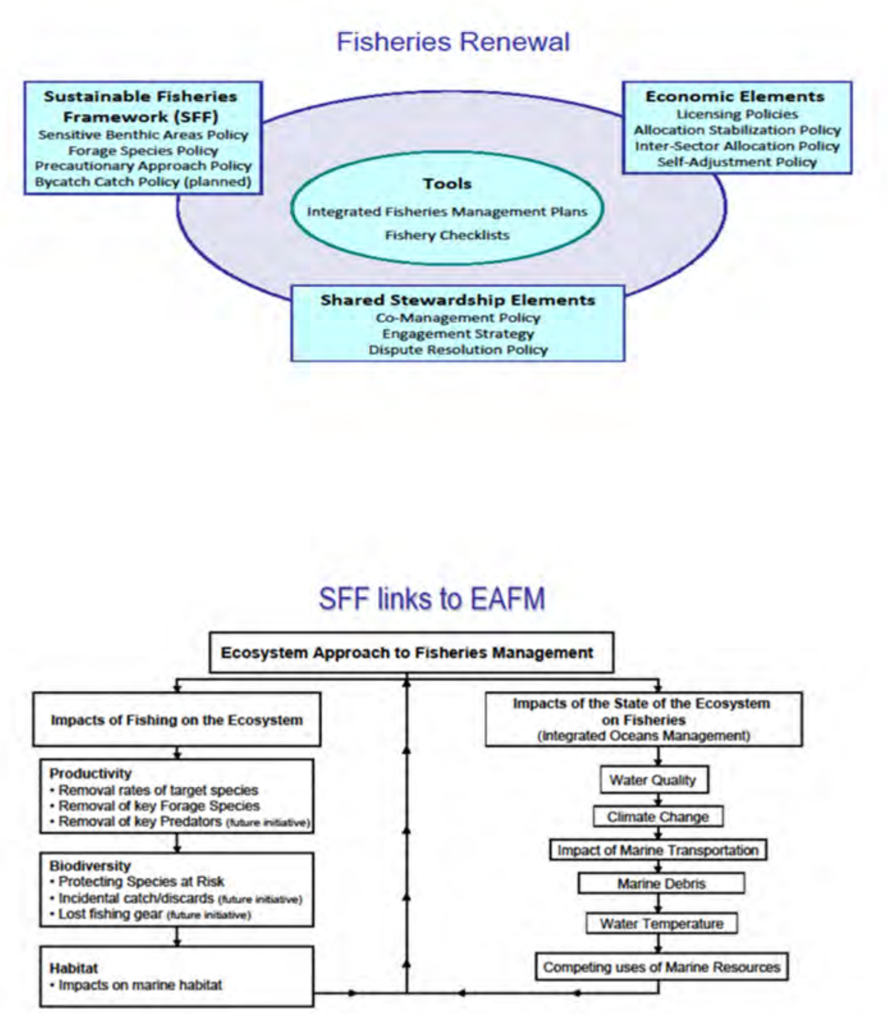

Figure 1. Schematic representation of Fisheries and Oceans Canada's plan for fisheries renewal and the relationship of the sustainable fisheries framework to ecosystem approach to fisheries management. Note that fishery checklists are a central tool in the framework.

is to increase commercial fish harvest. Such development has brought in focus fundamental existing conflicts between various agencies and co-management partners on the use of aquatic resources in Nunavut. The federal DFO is primarily concerned with maintaining the biological sustainability of fish stocks and ecosystem components through the board interpretation of Canada's Fisheries Act. The DFO is under pressure to determine the highest level of harvest for maximum long-term economic benefit to Canadians from aquatic resources. The Nunavut Wildlife Management Board (NWMB) is the agency that provides advice to the DFO Minister on appropriate harvest levels. As an agency involved in the Nunavut Land Claim Settlement it is equally concerned about (1) subsistence harvest of marine mammals and arctic char, and (2) commercial fisheries. The NWMB is under pressure to determine total allowable harvests for fish and marine mammal stocks, including subsistence and commercial harvests. The Nunavut Tingavvik Incorporated (NTI) is an agency concerned with promoting the social, cultural, and economic well-being of the Inuit of Nunavut and therefore is heavily involved in aspects of traditionally harvested marine mammals and arctic char. The Government of Nunavut (GN) Department of the Environment, Fisheries and Sealing Division is concerned with the development of industrial commercial fisheries and maintaining the operation of commercial fish processing plants, such as the one in Pangnirtung, Cumberland Sound. Interestingly, these agencies mirror the conflicts that Hilborn (2007) recognized:

> "*Traditionally, we have looked at four categories of objectives: biological, economic, social, and political. The biological objective, commonly found in legislation and international agreements, is the traditional maximum sustainable yield (MSY) that produces as much harvest as possible in the long term. In recent years there has been an additional emphasis on protection of nontarget species, particularly charismatic ones, such as whales and dolphins, and on protection of ecosystems. Economic objectives consider economic efficiency or "rent" as the desired outcome of fisheries management. Social objectives seek to spread employment and income among many participants in the fishery and the production of food and maintain traditional communities. These objectives are often in conflict.*"

In Cumberland Sound, the mandates of each of the agencies mentioned above mirror these categories—DFO with biological objectives, GN with economic objectives, NTI and NWMB with social and political objectives. Each has a flavoring of the other objectives, but these are the dominating aspects in their operation. The mix between maintaining traditional lifestyles while moving toward commercial enterprises in Nunavut has accelerated the need for tools to make hard decisions balancing the costs and benefits of harvest against ecosystem considerations. Ultimately, the solution to these issues relies on an ecosystem approach to management supported by a scientific model that can integrate all concerns.

For emerging fisheries in arctic regions, uncertainty is important due to data deficiency. For these reasons, ecosystem modeling is relatively undeveloped; however, use of models such as ECOPATH/ECOSIM (Pauly et al. 2000) and decision analysis (Clemen 1997) to dissect the management options at an ecosystem level is being explored. In this paper we demonstrate how decision analysis (DA) can be used to explore the options for management. We chose DA versus ECOPATH at this time in order to focus the result into as simple a set of variables as possible to illustrate the inconnectedness of decisions (e.g., preserve the ecosystem = inhibit the fishery, or promote the fishery = damage the ecosystem). Moreover, the DA approach can be more clearly connected to policy such as the precautionary approach and the ecosystem approach to managing fisheries. Analytically, the precautionary approach (PA), as followed in scientific stock assessment, is difficult to marry to the ecosystem approach. PA analysis involves relating a harvest rate with a stock size. One calculates the limit reference points of stock size or biomass indicating that fishing mortality must change. To attain such precision the system considers only stock and fishery as being the universe. For emerging fisheries such as those in Cumberland Sound, PA policy is active in principle but the determination of formal limit reference points will require several years of data collection. Ecosystem approach considers all components as interrelated to some degree. While it is a more realistic representation of nature it generally defies the calculation at the level of precision expected for fisheries decisions. DA allows PA fishery calculations to be underlying a node that can interact with the less precise node of ecosystem effects. It also can bring in nonscientific information readily, such as traditional knowledge, and create common currency for inclusion of disparate variables.

In this paper, we identify potential and realized ecosystem effects of emerging fisheries in the Cumberland Sound area and demonstrate how decision analysis can be used to explore management options. The study aims to (1) bring together the available information on Cumberland Sound fisheries and ecosystem components; and (2) characterize and develop a model of the system using decision analysis to facilitate analysis of decision-making.

Materials and methods

Description of Cumberland Sound

Cumberland Sound is a large inlet into southern Baffin Island, Canada (Fig. 2). It is a western arm of the Labrador Sea located between Baffin Island's Hall Peninsula and Cumberland Peninsula. It is approximately

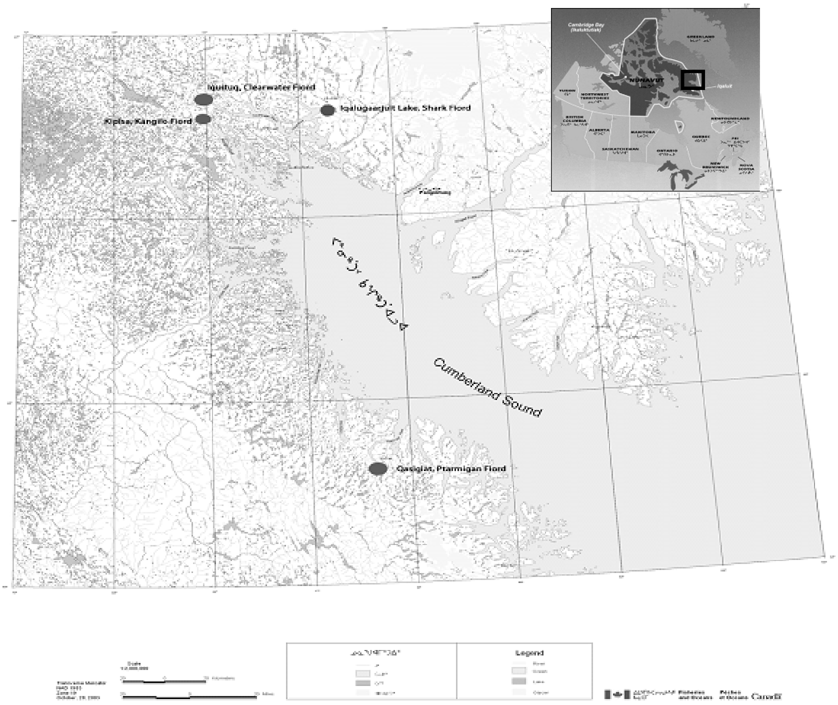

Figure 2. Cumberland Sound, Baffin Island, Nunavut Territory, Canada.

250 km (160 mi) long and 80 km (50 mi) wide. The sound contains many small islands. The tidal variation is considerable, which is likely important for arctic char to ascend their natal streams. The only settlement located on the Cumberland Peninsula is Pangnirtung.

There are numerous shallow streams and rivers connected to the sea around Cumberland Sound, which provide suitable habitat for arctic char to complete their life cycle. It is estimated that there are up to 60 significant char stocks around Cumberland Sound.

The deepwater bathymetry of Cumberland Sound is not documented at the same level of detail as similar marine systems in southern Canada. DFO is currently undertaking research to update navigational and bathymetric charts in the area. The lack of detail has been a factor in planning research surveys in the area. However, there is enough depth (800-1,000 m) to support a stock of Greenland halibut.

Char fishing for subsistence purposes has taken place since the Inuit have colonized the area, probably within the last 1,000 years.

Table 1. Basic elements for decision analysis framework for a fishery. Note that the expectations for the successful fishery are sustainable profit, sustainable target species population, and sustainable ecosystem components.

Node (element)	Type	Depends upon	Influences
Satisfaction with fishery	Desired outcome	Profit, sustainable target spp., sustainable ecosystem	None
Profit	Fixed, discrete probabilistic, probabilistic	Market value, harvest	Satisfaction
Sustainable target spp.	Fixed, discrete probabilistic, probabilistic	Harvest	Satisfaction
Sustainable ecosystem	Fixed, discrete probabilistic, probabilistic	Bycatch, sustainable target species	Satisfaction
Harvest	Fixed, discrete probabilistic, probabilistic	Fishing effort, location	Sustainable target spp.
Bycatch	Fixed, discrete probabilistic, probabilistic	Fishing effort, gear type, location	Sustainable ecosystem
Location (spatial)	Control	NA	Harvest, bycatch
Gear type	Control	NA	Harvest, bycatch
Season	Control	NA	Harvest, bycatch
Sustainable harvest forecast	Fixed, discrete probabilistic, probabilistic	Quota	Science data and analysis
Quota	Decision node	Sustainable harvest forecast	Fishing effort

The commercial fishing of char commenced in the 1980s using modern gillnetting techniques.

Winter fishing for Greenland halibut started in 1986 with the transfer of techniques from Greenland using longlines set at the ice flow edge in the winter (DFO 2008a). Fish are transported to the plant in Pangnirtung in insulated tubs attached to qamatiks (sleds). More recently, exploratory fishing in the summer using commercial longlining vessels has commenced and the GN would like to continue the development to replace losses in production from the winter fishery due to changes in the sea ice formation.

Sources of information on Cumberland Sound fisheries

Information on the current Cumberland Sound fisheries was gathered from published DFO reports, scientific journals, DFO internal documents, the records of Pangnirtung Fisheries (commercial fish processing plant), and from our own observations gathered during recent research efforts and community consultations with fishers of Pangnirtung. Fishery independent research surveys of some arctic char stocks under exploratory fishing licenses from DFO to the community have been conducted since 1998. We were able to observe the fishery directly to determine possible bycatch. We also were able to sample with both commercial fishing and small mesh gear within the estuary and the lakes for ecosystem effects.

There has been no fishery independent survey for Greenland halibut but there are records from the commercial fishery of the catch and bycatch within the fishery. The initial phase of the summer exploratory fishery has had professional fishery observer coverage, which provided information on catch and bycatch. Internal DFO records and reports as well as published stock assessment reports from the DFO Canadian Stock Assessment Secretariat were consulted.

Ecosystem information is scant, so while we also referred to DFO reports, scientific journals, and DFO internal documents, we relied more on direct observation by research biologists and the fishers within the community. DFO has between 8 and 10 meetings with the fishermen per year where observations on harvest, bycatch, and changes in the ecosystem are discussed.

Decision analysis

Decision analysis was used to develop a general model incorporating three potentially conflicting expectations within EBFM: sustainable profit; sustainable target species population; and sustainable ecosystem components (Table 1). Within the decision model framework we present these as nodes in the decision tree (Fig. 3). Each decision d in a set D of available decision options will lead to an outcome $o = f(d)$. All possible outcomes form the set O. Assigning a utility $U_O(o)$ to every outcome, we can define the utility of a particular decision d as:

$$U_D(d) = U_O[f(d)]$$

We can then define an optimal decision d_{opt} as one that maximizes $U_D(d)$:

$$d_{opt} = arg\ max\ U_D(d) \text{ where } d \in D$$

Solving the problem can thus be divided into three steps:

1. predicting the outcome o for every decision d.
2. assigning a utility $U_O(o)$ to every outcome o.
3. finding the decision d that maximizes $U_D(d)$.

In cases where it is not possible to predict with certainty what will be the outcome of a particular decision, a probabilistic approach is necessary. In its most general form, it can be expressed as follows: given a decision d, we know the probability distribution for the possible outcomes described by the conditional probability density $p(o \mid d)$. We can then calculate the expected utility of decision d as:

$$U_D(d) = \int p(o \mid d) U(o) do$$

where the integral is taken over the whole set O. An optimal decision d_{opt} is then one that maximizes $U_D(d)$, just as above:

$$d_{opt} = arg\ max\ U_D(d)\ where\ d \in D.$$

Graphical representation of decision analysis problems commonly use influence diagrams and decision trees. Both of these tools represent the alternatives available to the decision maker, the uncertainty they face, and evaluation measures representing how well they achieve their objectives in the final outcome. Uncertainties are represented through probabilities and probability distributions. The decision maker's attitude to risk is represented by utility functions and their attitude to trade-offs between conflicting objectives can be made using multi-attribute value functions or multi-attribute utility functions (if there is risk involved). In some cases, utility functions can be replaced by the probability of achieving uncertain aspiration levels. Decision analysis advocates choosing that decision whose consequences have the maximum expected utility (or which maximize the probability of achieving the uncertain aspiration level).

In decision analysis, a "decision tree"—and the closely related influence diagram—are used as a visual and analytical decision support tool, where the expected values (or expected utility) of competing alternatives are calculated. A decision tree consists of three types of nodes:

1. Decision nodes: commonly represented by squares.

2. Chance nodes: represented by ovals or circles.

3. End nodes: represented by rounded squares.

Decision models for the Cumberland Sound fishery were developed using fixed parameters, and quasi continuous and continuous variables (Clemen 1997). To describe each type of model, a fixed parameter model employs set values for each node. It is the simplest model. For example, a decision node would have values such as increase, stay the same, and decrease. The increase might be by a set amount such as 20% increase in quota. Other nodes would have one value attached to them. For example, the value for fishing effort would be a single value as a function of the size of the quota, the season of fishing (summer is more efficient than winter) and the gear type (trawls are more efficient than

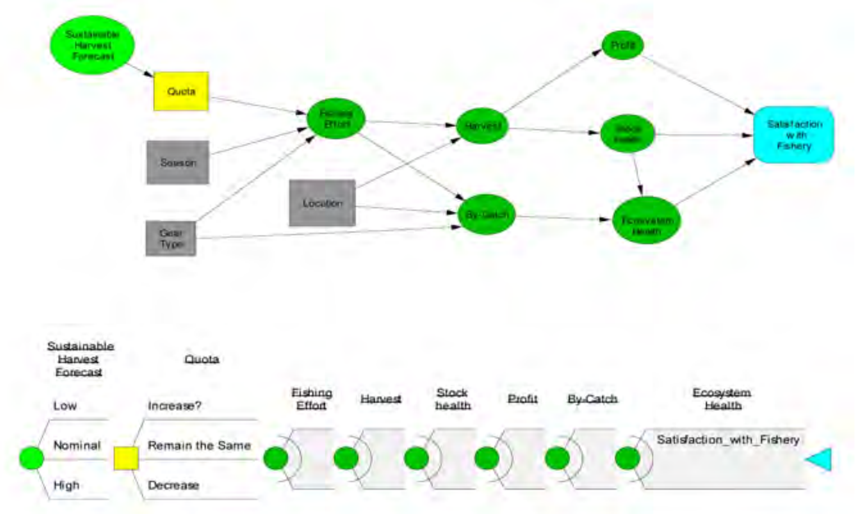

Figure 3. General influence diagram and decision tree for the Cumberland Sound fishery. Note the model represented is the probabilistic form. Probabilistic nodes are represented by ovals. Fishery management decisions are represented as squares. The output variable (satisfaction with fishery) is in the rounded square (diagram) or triangle (tree).

longlines. The other extreme would be all nodes modeled as continuous functions. Thus, the quota decision would be modeled as a continuous function between quota = 0 to quota = 100% of the stock. The effects would be developed as probability functions of the related nodes. Monte Carlo simulation was employed to estimate the range of outcomes on the satisfaction with the fishery.

Results and discussion

Fisheries and ecosystem components

The Cumberland Sound commercial fisheries consist of three parts: (1) arctic char year-round fishery prosecuted using gillnets at the mouths of rivers (summer) and in lakes under the ice (winter) (Harris and Tallman 2010); (2) a winter fishery on the ice floe edge for Greenland halibut using longlines; and (3) a relatively new fishery in summer for Greenland halibut using longlines off commercial fishing vessels. There is also a desire to use large mesh gillnets in the summer.

The sustainability of the commercial harvest of four arctic char stocks, Kipisa, Kingnait Fiord, Isuituq, and Qasigiat) has been reviewed in recent years (Tallman 2005, Toyne and Tallman 2009, DFO 2009, Harris and Tallman 2010, Martin and Tallman 2010). Three of these were considered sustainable at the current level of harvest. The only exception may be Kingnait Fiord where the stock information suggested that subsistence harvest needs might be better served with a reduced commercial catch (DFO 2009). The catches of arctic char are almost completely without bycatch (DFO 2002). The nets are set perpendicular to the shore with one end attached to the shore, in the estuaries of rivers and in some lakes during open water. Char is also fished during the winter by fishing through the ice that covers lakes.

The total allowable catch (TAC) for Greenland halibut in the winter fishery has been set at 500 t since 1994. In 2005, a separate 500 t Greenland halibut allocation was established for the traditional winter fishing grounds in the inner portion of Cumberland Sound (Government of Nunavut 2006). This allocation was based on the results of tagging data, which has indicated that Greenland halibut in this area may be isolated from the rest of Cumberland Sound and the offshore areas of Northwest Atlantic Fisheries Organization (NAFO) fishing area 0B. In 2004 the NAFO Scientific Council recommended that "Based on available information, Scientific Council recommends that a separate stock management area be established for the traditional winter fishing grounds for Greenland halibut in the inner portion of Cumberland Sound." This area falls within the Nunavut Settlement Area (NSA) and as such the full allocation, as per the terms of the Land Claims Agreement, is provided to Nunavut.

The Greenland halibut winter fishery has a moderate bycatch of Greenland shark (*Somniosus microcephalus*) (<10% by weight) but the summer longline fishery has experienced shark bycatches that exceed the harvest of Greenland halibut (Cosens et al. 1995, DFO unpubl. data). This is considered a serious concern and must be addressed.

There are also several species of marine mammals in the area that are harvested by the Inuit for subsistence purposes. The presence of marine mammals has been the rationale for not allowing the use of gillnets in the Greenland halibut summer fishery (DFO 2008b). Ringed seal (*Pusa hispida*) and beluga whale (*Delphinapterus leucas*) are the most important species (S. Ferguson and Pierre Richard, DFO, pers. comm.). The beluga whale is considered to be at low numbers and in danger of being listed under the Canadian Species at Risk Act legislation. The Cumberland Sound population is currently listed as threatened (COSEWIC 2004). Important forage species for marine mammals and arctic char are arctic cod, capelin, and amphipods (G. Tomy and S. Wiley, DFO, unpubl. data). As well, arctic char depend upon freshwater invertebrates in the early part of their lives (T. Loewen, DFO, unpubl.

Table 2. Ecosystem components of Cumberland Sound

Ecosystem component	Trophic position	Sources
Polar bear	Top predator	COSEWIC 2004
Killer whale	Top predator	COSEWIC 2004
Seal	Predator	S. Ferguson, DFO, unpubl.
Beluga	Predator	COSEWIC 2004
Narwhal	Predator	D. Emery, GN, Parks Canada, pers. comm.
Walrus	Predator	D. Emery, GN, Parks Canada, pers. comm.
Greenland shark	Top predator	DFO 2008b
Bowhead	Forager	S. Ferguson, DFO, unpubl.
Greenland halibut (turbot)	Top teleost predator	DFO 2008b
Arctic char	Predator	Harris and Tallman 2010
Capelin	Forage fish	K. Urlich, Pangnirtung Fisheries Ltd, pers. comm.
Arctic cod	Forage fish	K. Urlich, Pangnirtung Fisheries Ltd, pers. comm.
Shrimp	Forage	Dennard et al. 2010
Bivalves	Forage	Dennard et al. 2010
Benthic invertebrates	Forage	Dennard et al. 2010
Amphipods/crustaceans	Forage	COSEWIC 2004
Krill/copepods	Forage	Dennard et al. 2010
Marine zooplankton	Forage	Pomerleau et al. 2011
Marine benthic primary producers	Primary production	Pomerleau et al. 2011
Marine pelagic primary producers	Primary production	Pomerleau et al. 2011
Stickleback	Forage fish	DFO unpubl. data
Freshwater zooplankton	Forage	Assumed
Freshwater primary productivity	Primary production	Assumed

GN = Government of Nunavut. DFO = Canada Department of Fisheries and Oceans.

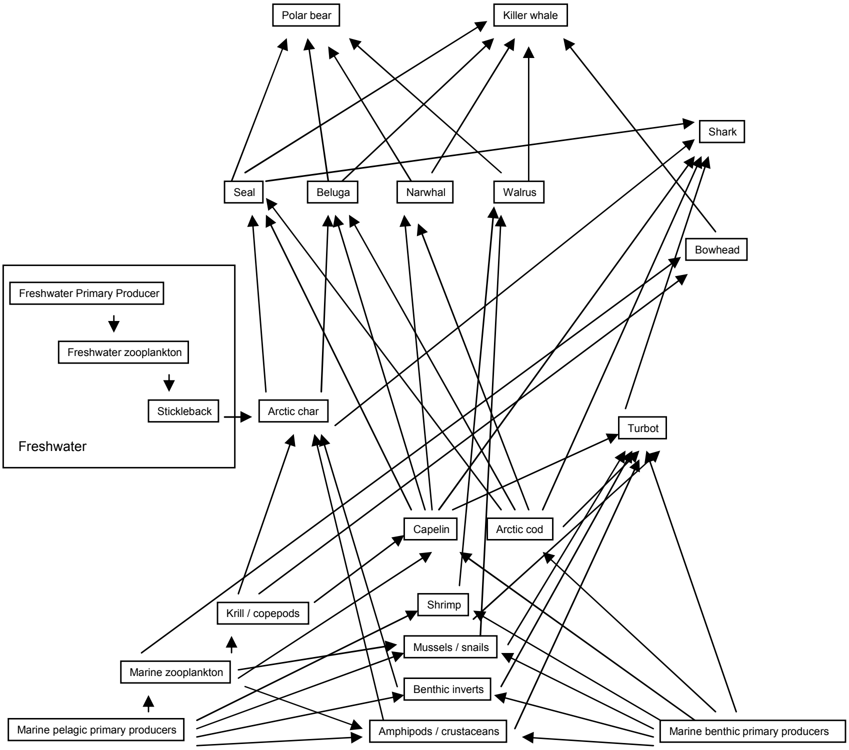

Figure 4. Trophic relationships in Cumberland Sound. Humans are not included in this representation. If included they would be at the top of the web—removing killer whale, polar bear, seal, beluga, narwhal, walrus, shark, bowhead, arctic char, and Greenland halibut (turbot), but having influence on all aspects of the system.

data). Ecosystem components and sources of information are shown in Table 2. Fig. 4 shows a schematic trophic ecosystem model for Cumberland Sound.

For Cumberland Sound fisheries most of the ecosystem effects on fisheries are not presently a concern with the exception of climate change effects on sea ice cover and food web structure. The timing and extent of development of the land-fast ice varies from year to year depending on the latent heat of surface waters and weather conditions. The ice conditions have an effect on fishing location and depth for Greenland halibut and consequently on catches, from one year to the next. The fishermen can access fishing areas near Imigen Island in the southwest corner of Cumberland Sound first. Then they move with the ice as it expands to cover deeper waters. If it is a poor year for ice formation they may not be able to move from the Imigen Island area. In recent years, reductions in sea ice cover have prevented the winter fishery for Greenland halibut in Cumberland Sound from being prosecuted as effectively as in the 1990s. Climate change may also cause changes in trophic structure including an increase in biomass of forage fish such as capelin (*Mallotus villosus*), in turn affecting higher predatory species such as arctic char. In recent years, increased reliance on capelin by arctic char in Cumberland Sound has been suggested to cause changes in the quality of arctic char flesh (i.e., color and taste) by local fishers (DFO 2011).

Realized and potential effects of fishing

The realized effects of increased fishing in Cumberland Sound are shown in Table 3. Presently most of the impacts are potential rather than realized. To date, destruction of fish habitat by fishing activities has not been important for Cumberland Sound fisheries that are prosecuted by gillnets (arctic char), longlines (Greenland halibut), and rifle (marine mammals). Similarly, harvest of forage fish has not been an issue as yet as fisheries are principally focused on non-forage species.

The expected increase in fishing for arctic char may result in some ecosystem effects even though the fishery has a nearly nil bycatch. Inuit fishermen do not simply harvest arctic char when they go fishing. They will conduct subsistence harvest at the same time on ringed seals and other marine mammals. As well, they will scout to plan harvest of more closely controlled species such as beluga whales. Therefore there will be increased impacts but the level of these is difficult to predict. Regardless, it is likely the additional impacts on marine mammals will be small and not controllable by additional regulation.

The Greenland halibut fishery impacts on Greenland shark are substantial and will need to be addressed by management measures. The ecosystem effect may be ameliorated by encouraging alternative gear that can exclude sharks such as traps or pots. However, it is unknown whether this gear will be cost-effective for the fishery. The species may be separated in time and space at certain times and seasonal/spatial regulations may be able to be employed. However, there is at present a limited knowledge of the spatiotemporal patterns of halibut and shark as well as most other species. Surveys and other research to reduce the uncertainty in these parameters are needed.

Many of the potential impacts are related to possible catastrophic bycatch of marine mammals. The most severe potential impact would be if gillnet fishing for Greenland halibut is developed (DFO 2008b). Modeling of the likelihood of a marine mammal impact as a result of tangling with nets should be undertaken as the commercial resource

Table 3. Realized and potential impacts of increased fishing in Cumberland Sound.

Impact	Impact type	Source
Gillnets with marine mammals	Potential	DFO 2008b
Longlines with Greenland shark	Realized	DFO 2008b
Hunting of seals, direct	Potential	S. Ferguson, DFO
Increased hunting of belugas	Potential	P. Richard, DFO
Increased F on Greenland halibut (turbot) (more sustained over many years)	Realized	DFO 2008a
Summer harvesting of Greenland halibut	Realized	DFO 2008b
Trawl scouring effects	Potential	M. Treble, DFO
Increased pressure on char stocks	Realized	Martin and Tallman 2010
Decreased pressure on more local char stocks	Realized	DFO unpubl.
Reduced fishing on char in winter	Realized	Martin and Tallman 2010
Harvesting of nearshore invertebrates	Potential	DFO unpubl.
Increased abundance of inverts preyed upon by Greenland halibut	Potential	Assumed
Change in relative abundance of resident and anadromous char within systems	Potential	Assumed
Hunting of killer whales	Realized	S. Ferguson, DFO

users have a strong interest in developing fisheries of this type (Government of Nunavut 2006). Given the low productivity of marine mammals and cultural importance to the Inuit, we advise considerable caution before moving in this direction. With detailed understanding of the spatiotemporal patterns by Greenland halibut and marine mammals it may be possible to develop management measures to reduce the likelihood of this potential ecosystem effect, but at present this information does not exist.

EBFM fishery decision-making with DA

The most useful DA model for Cumberland Sound fisheries was a combination of fixed and continuous effects. The quota, gear type, season, and location were modeled as fixed effects. Quota fixed effects were to increase, decrease or remain with status quo. For Greenland halibut the low value would be 250 t, nominal was 500 t, high value was 750 t. Gear type was modeled as longline, gillnet, and trawl. Season was modeled by winter and summer. Location was inside or outside NAFO area 0, which borders Cumberland Sound. Continuous nodes were modeled based on functional relationships with the corresponding decision nodes.

Satisfaction with the fishery depends upon three different criteria: fishery profits, stock health, and ecosystem health (Fig. 3). Profit is dependent upon the harvest, market value, and the cost of gathering the harvest—represented by fishing effort. Stock health will mainly depend upon the harvest levels. Ecosystem health is dependent upon stock health and the impact from bycatch. Harvest is influenced by fishing effort and location. Fishing effort is modeled as a continuous variable while location has two values—inside and outside the shallow zone in Cumberland Sound. Along with location, quota, season, and gear type can be regulated in fisheries management. Fishing effort is influenced by quota, season of fishing, and gear type. Whether the quota should be changed is influenced by the advice from scientific analyses—the stock health forecast. The decision tree is shown in Fig. 3.

The fixed variable model, while useful to outline the problem, provided little insight and was unrealistic to the uncertainties in the information that could be gathered on biological factors. The mainly probabilistic model is likely closest to biological reality in this situation but defining each probability distribution was difficult and Monte Carlo simulations could not produce results that were easily interpretable. The most useful model was one where the biological factors (ovals) were modeled as discrete probabilities (e.g., low, nominal, high values).

For arctic char ecosystem effects of fishing were minimal and the fishery management decisions could be made on the basis of single stock dynamics. For Greenland halibut ecosystem effects from direct bycatch and indirect effects due to the reduction of Greenland halibut abundance could be severe, particularly in the summer fishery.

Conclusions

Ultimately our goals are to (1) set harvest levels for char, Greenland halibut, and marine mammals within an EBFM context; (2) model system trophic dynamics (as we know it) with a conceptual, mass balance approach; and (3) incorporate our management scenarios into a decision analysis framework to facilitate management strategy evaluation. We are not at that point yet because there are not sufficient data to fully model the population dynamics of target species and the accompanying ecosystem effects of fishing. In particular, we need spatial and temporal surveys of distribution and abundance of the target species and all relevant ecosystem components. As well, we need to estimate the transfer rates between ecosystem components to fill out the ecosystem model from a conceptual to a quantitative phase.

Our preliminary analysis indicated that (1) expansion of arctic char fisheries in Cumberland Sound can proceed without serious ecosystem effects. Management decisions can be made on a stock-by-stock basis. (2) Greenland halibut fisheries, especially summer fisheries, have the potential to impact sharks and whales in the ecosystem. As these *K*-selected taxa have low resilience management decisions on quota, season and gear type have significant effects on the ecosystem.

At the end of the day we must decide what we consider a success in a fishery. As Hilborn (2007) points out the traditional combination of biological, economic, political, and social conflicts being managed independently of each other and set up in an adversarial way have moved fisheries to collapse. A more modern approach is with the Marine Stewardship Council (MSC) approach, which looks critically at the condition of the fish stock, the impact on the environment, and the management system in place. As of 2005, 14 fisheries around the world have been certified by MSC, including many of the fisheries most commonly cited as being well managed. However, the MSC criteria do not include any economic or employment objectives. In our DA approach we attempted to incorporate an economic objective, which we would also suggest maximizes employment without compromising other concerns. Decision analysis may allow the exploration of alternatives within a complex EBFM approach in Cumberland Sound, but it will be difficult to have reliable data for all parameters. Regardless, it provides an effective framework for management guidance in the context of data deficient, emerging fisheries.

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Do Species Interactions and Stochasticity Matter to Optimal Management of Multispecies Fisheries?

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Abstract

Multispecies fisheries management looks at a bigger picture in addressing the long-term consequences of present decisions. This implies an ecosystem management that includes a number of species and their physical, biological, and economic interactions. These interactions make the growth of resources stochastic and increase complexity in understanding stock dynamics and optimal catch for such a stochastic and multiple-stock system. To address the issue of identifying optimal catch of stochastically growing multi stocks, we have formulated and applied a time-continuous stochastic model. The model contributes to multispecies bioeconomic management of marine ecosystems. An application of the model in a predator-prey relationship in the Barents Sea revealed that the optimal catch for stochastically growing stocks in a multispecies interaction model is different from the deterministic model.

Introduction

Marine fisheries are vital resources for the ecology and economy. They also play a very crucial role in ensuring food security for the growing population (FAO 2008). Rendering these fisheries as a sustainable source of food for the world, however, requires an effort of addressing overexploitation in the fishing industry and improving fisheries management (Speer 1995). Among the majority of instruments available for marine fishery management, harvesting strategy is considered the best instrument (Agnarsson et al. 2008). The strategy is considered optimal if the rent from the fishery is maximized over the planning horizon. There exists a large body of literature on bioeconomic modeling that serves the purpose of rent maximization in public fishery management. These bioeconomic models date back to the early work of Gordon (1954) and Scott (1955). Smith (1969), Clark (1973), Hannesson (1975), May et al. (1979), Grafton et al. (2000), Sandal and Steinshamn (2001a), Arnason et al. (2004), and many others suggested optimal management rules to maximize the economic rent from fisheries in a deterministic setting, primarily on a single species modeling approach.

However, there are biological, physical, and economic interactions among species in the ecosystem. There are management problems for species that involve interactions between species at different trophic levels (May et al. 1979). The most common approach, single species management in multispecies fisheries, ignores the ecological relationships among species as well as the technological and economic relationships between species (Kasperski 2010). This may lead to misleading results and incorrect policy decisions causing overexploitation or underexploitation of the stocks (Fleming and Alexander 2003, Hoff et al. 2010). Therefore, multispecies management is a key approach for sustainable management of such marine fisheries.

The importance of multispecies fishery management was realized during the early 1970s when the world's major commercial fisheries collapsed (for example, see May et al. 1979). Multispecies fishery management looks at the bigger picture, addressing the long-term consequences of present decisions. It implies an analysis and management of a marine ecological system that includes a number of species and their biological and physical interactions, rather than managing different species individually. Needless to say, the economic interaction plays a crucial role in creating the overall harvesting pressure on commercially important species.

The earlier studies focused mainly on a predator-prey relationships on different trophic levels in their multispecies management model (for example Bogstad et al. 1997, May et al. 1979, Yodzis 1994). Those studies, however, focused on biological yields without considering the economic aspects of harvesting. Later authors like Fleming and Alexander (2003) and Kar and Chaudhuri (2004) suggested a deterministic bioeconomic model with an optimal equilibrium solution. However, they also remarked that it is extremely difficult to find the optimal paths, even in the cases with linear objective functions. Hollowed et al. (2000) compared multispecies models with single species models and found that multispecies models provide a distinct advantage over the single species models, allowing users to model natural mortality and growth rates more realistically. They also indicated that multispecies models improve the understanding of fish population dynamics. Unfortunately, multispecies bioeconomic models are very limited due to unavailability of the analytical solutions (Posch and Trimborn 2010) and due to computational difficulties (Singh et al. 2006), particularly in solving nonlinear dynamic models in higher dimensions. Some bioeconomic models by Clark (1990), Woodward and Bishop (1999), Iversen (2006), Agnarsson et al. (2008), and Sandal and Steinshamn (2010) suggest optimal management of multispecies fishery in deterministic settings. Nevertheless, in reality most of the decisions are required to take place in an uncertain environment (Charles and Munro 1985). The physical interactions, such as growing environment, different external shocks, and diseases in the ecosystem create stochastic growth of marine resources. The stochastic process is central in explaining the uncertainty in growth and development of natural resources.

Although researchers such as Reed (1979), Charles (1983), Charles and Munro (1985), Clark and Kirkwood (1986), Hannesson (1987), Sandal and Steinshamn (1997a), Sethi et al. (2005), Singh et al. (2006), Kugarajh et al. (2006), and McGough et al. (2009) include stochasticity in their single species models, stochastic multispecies models lack in the bioeconomic literature (Agnarsson et al. 2008). To address the issue of optimal management of stochastically growing multi-stocks, we have formulated and applied a time-continuous stochastic model that contributes in multispecies bioeconomic management of marine ecosystems. Our study advances previous models by Agnarsson et al. (2008) and Sandal and Steinshamn (2010) and compares stochastic models with deterministic models in single species and multispecies fisheries. We explore how species interaction and stochasticity affect the optimal management of multispecies fisheries.

Our study employs a feedback approach (Sandal and Steinshamn 1997b, 2001b), where the optimal control (harvest) is a direct function of the state variable (stock). In contrast to the commonly used time paths approach (optimal harvest as a function of time), the feedback approach is superior when faced with uncertainty (Agnarsson et al. 2008). The feedback models take prevailing stocks as inputs, and therefore these models automatically respond to the unexpected changes in the stock and thus adapt to new situations (Sandal and Steinshamn 1997b). We apply a dynamic programing (DP) technique to obtain the optimal feedback solution. In the DP technique, value function iteration is carried out to solve for the optimal solution (Judd 1998). DP is especially a useful method when considering a multispecies management model under stochasticity (Sanchirico and Springborn 2011).

The bioeconomic model

Bioeconomic models are dynamic and combine both economics and biology—an economic part that characterizes the optimal management policy and a biological part that defines the natural constraints for such optimal policy. Bioeconomic models can be a recruitment model (Ricker

1954), a surplus production model (Schaefer 1957), or a year-class model (Beverton and Holt 1957). These models can be formulated in discrete or continuous time (Sandal and Steinshamn 2010). Here we use a continuous time surplus production model.

Single species model

Following Clark (1990), Sandal and Steinshamn (1997b), McDonald et al. (2002), and Agnarsson et al. (2008), a general deterministic growth function for a single species stock can be given as:

$$dx = [f(x) - h]dt \tag{1}$$

A stochastic growth function can be obtained by adding a stochastic term in equation (1)

$$dx = [f(x) - h]dt + \sigma_0(x)dB \tag{2}$$

where $f(x) = rx^2(1 - \frac{x}{k})$

is a modified logistic growth function with $f(0) = f(k) = 0$, x is the stock biomass, h is the harvest rate, r denotes the intrinsic growth rate, and k denotes the carrying capacity of the species. The term $\sigma_X(x)dB_X$ represents the stochastic part of the stock incremental growth relationship; σ_0 is the diffusion term and represents volatility in the growth model. The term dt is time increment and dB_X is the Brownian motion, which are i.i.d. with mean zero and variance dt. We assume the natural condition of non-negativity of x and h.

Multispecies model

The general biological interdependent deterministic growth functions for two interacting predator-prey species can be obtained from Agnarsson et al. (2008) and Sandal and Steinshamn (2010). Let x be prey species and y be the predator species. The deterministic growth increments of the species can be given as:

$$\begin{aligned} dx &= [f(x,y) - h_x]dt \\ dy &= [g(x,y) - h_y]dt \end{aligned} \tag{3}$$

The function $f(x,y)$ and $g(x,y)$ are the biological growth functions of two fish species respectively; h_i represents the fishing mortality or the harvest rate of species ($i = x,y$). Furthermore, a two-species interaction model with a stochastic dynamic can be formulated by adding stochastic terms in equation (3) as:

$$\begin{pmatrix} dx \\ dy \end{pmatrix} = \begin{pmatrix} f(x,y) - h_x \\ g(x,y) - h_y \end{pmatrix} dt + \begin{pmatrix} \sigma_{11}(x,y) & \sigma_{12}(x,y) \\ \sigma_{21}(x,y) & \sigma_{22}(x,y) \end{pmatrix} \begin{pmatrix} dB_1 \\ dB_2 \end{pmatrix} \tag{4}$$

or formally as $dZ = F(Z,h)dt = \sigma(Z)dB.$

Although it is likely that stochastic events are correlated among species, we assume it to be small enough to be neglected, i.e., $\sigma_{12}(x,y) = \sigma_{21}(x,y) = 0$. We further simplify by setting each species' volatility as a linear function of its own stock level, $\sigma_{11}(x,y) = \sigma_1 x$ and $\sigma_{22}(x,y) = \sigma_2 y$. This assumption is made for two reasons: for simplicity in the numerical approximation and little is known about more complicated functional volatility dependence. Since the species are prey and predator species, the stochastic processes that affect prey directly may not affect the predators in a direct sense or vice versa. The other species are still indirectly affected by such a process through interaction in the drift (deterministic) part. Epidemics are typically of this kind. Therefore, we believe that inclusion of stochasticity improves the model compared to the deterministic case. Equation (4) can now be written as:

$$\begin{aligned} dx &= (f(x,y) - h_x)dt + \sigma_1 x dB_1 \\ dy &= (g(x,y) - h_y)dt + \sigma_2 y dB_2 \end{aligned} \tag{5}$$

The biological growth functions are specified as:

$f(x,y) = a_1x^2 - a_2x^3 - a_3xy$ and $g(x,y) - b_1y^2 - b_2y^4 + b_3xy$

and a_1, a_2, a_3, b_1, b_2, and b_3 are parameters. The term xy is the interaction between the species where the predator feeds on the prey. In equation (5), the term $\sigma_{(.)}(.)dB_{(.)}$ represents the stochastic part of the stock growth relationship; $\sigma_{(.)}$ is the diffusion term and represents volatility in the growth models. The terms $dB_{(.)}$ are uncorrelated Brownian motions, which are i.i.d. with mean zero and variance dt. We assume stocks and harvests rates to be non-negative.

Our basic biological model is now given by:

$$\begin{aligned} dx &= (a_1x^2 - a_2x^3 - a_3xy - h_x)dt + \sigma_1 x dB_1 \\ dy &= (b_1y^2 - b_2y^4 + b_3xy - h_y)dt + \sigma_2 y dB_2 \end{aligned} \tag{6}$$

Equation (6) shows that both species have stochastic growth and interactions among them. But what if only one of the species has stochastic growth? To understand the effect of stochastic growth of one species on optimal exploitation in a multispecies ecosystem, equation (6) can be simplified by setting one of the stochastic parameters to zero. For example, the prey species stochastic growth in multispecies ecosystem is modeled by assuming $\sigma_{22} = 0$ and written as:

$$(6.1)\qquad \begin{aligned} dx &= (a_1x^2 - a_2x^3 - a_3xy - h_x)dt + \sigma_1 x dB_1 \\ dy &= (b_1y^2 - b_2y^4 + b_3xy - h_y)dt \end{aligned}$$

While the predator species stochastic growth in a multispecies ecosystem is modeled by setting $\sigma_{11} = 0$, equation (6) can be modified as:

$$(6.2)\qquad \begin{aligned} dx &= (a_1x^2 - a_2x^3 - a_3xy - h_x)dt \\ dy &= (b_1y^2 - b_2y^4 + b_3xy - h_y)dt + \sigma_2 y dB_2 \end{aligned}$$

Although no stochasticity is assumed in one of the species in equation (6.1) and (6.2), both species become stochastic due to the existence of the predator-prey interaction term (xy) in the growth function.

The second part of the bioeconomic model consists of economic components, which are obtained from Agnarsson et al. (2008) and Sandal and Steinshamn (2010) as expressed below:

$$(7.1)\qquad \pi_x(x,h_x) = p_x(h_x)h_x - c_x(x,h_x)$$

$$(7.2)\qquad \pi_y(y,h_y) = p_y(h_y)h_y - c_y(y,h_y)$$

$\pi_x(x,h_x)$ and $\pi_y(y,h_y)$ are the net revenues from species x and y respectively. The net revenue from the harvesting of two species is the sum of the revenues from each species.

$$(7.3)\qquad \pi(x,y,h_x,h_y) = \pi_x(x,h_x) + \pi_y(y,h_y) = p_x(h_x)h_x - c_x(x,h_x) + p_y(h_y)h_y - c_y(y,h_y)$$

The function $\pi(x,y,h_x,h_y)$ is the net revenue. Functions $p(.)$ and $c(.)$ are inverse demand functions and cost functions respectively. The demand functions and cost functions are specified as:

$$(8)\qquad \begin{aligned} p_x(h_x) &= p_1 \\ c_x(x,h_x) &= q_1 h_x^{\alpha_1} \\ p_y(h_y) &= p_2 - p_3 h_y \\ c_y(y,h_y) &= q_2 \frac{h_y^{\alpha_2}}{y} \end{aligned}$$

After substituting the expression from equation (8) into equation (7.3), the profit function can be specified as:

$$\pi(x,y,h_x,h_y) = p_1 h_x - q_1 h_x^{\alpha_1} + p_2 h_y - \frac{q_2}{y} h_y^{\alpha_2} - p_3 h_y^2 \tag{9}$$

where p_1, q_1, a_1, p_2, q_2, a_2, and p_3 are economic parameters obtained from Agnarsson et al. (2008) and Sandal and Steinshamn (2010).

We assume that prey is a schooling species, and therefore the unit cost of harvest is independent of stock size. Our revenue function depends only on the predator stock and harvest level of prey and predators and the revenue function equation (9) can simply be written as a function of three arguments: $\pi(y,h_x,h_y)$.

Given the growth functions and profit function, the management objective is to maximize expected net present value of the return from the harvest schedule over an infinite time horizon. This can be achieved by maximizing following function:

$$J(y,h_x,h_y) = E\left[\int_0^{\infty} e^{-\delta t} \pi(y,h_x,h_y)\,dt\right] \tag{10}$$

The non-negative parameter is the discount rate and *E* is the expectation operator. Along with dynamic constraints and appropriate boundary conditions, the dynamic optimization problem can be written as:

$$\begin{aligned} V(x_0,y_0) &= \max_{h_x,h_y \geq 0} J(y,h_x,h_y) \\ x(t=0) &= x_0 \\ y(t=0) &= y_0 \end{aligned} \tag{11}$$

The optimal solution in the predator-prey model can be obtained by solving the following Hamilton-Jacobi-Bellman (HJB) equation (Kushner and Dupuis 2001) along with appropriate boundary conditions. This is an equation for any feasible initial condition and hence we replace (x_0,y_0) with (x,y).

$$\begin{aligned} \delta V(x,y) = \max_{h_x,h_y \geq 0} \Big\{ &\pi(y,h_x,h_y) + V_a^T(x,y) F(x,y,h_x,h_y) \\ &+ \frac{1}{2} tr\left[\sigma(x,y)\sigma^T(x,y) V_{aa}(x,y)\right]\Big\} \end{aligned} \tag{12}$$

where

$$V_a(x,y)=\begin{pmatrix} V_x(x,y) \\ V_y(x,y) \end{pmatrix}\quad,\quad F(x,y,h_x,h_y)=\begin{pmatrix} f(x,y)-h_x \\ g(x,y)-h_y \end{pmatrix}=\begin{pmatrix} a_1x^2-a_2x^3-a_3xy-h_x \\ b_1y^2-b_2y^4+b_3xy-h_y \end{pmatrix}\quad,$$

$$\sigma(x,y)=\begin{pmatrix} \sigma_1 x & 0 \\ 0 & \sigma_2 y \end{pmatrix}\quad,\ \text{and}\quad V_{aa}(x,y)=\begin{pmatrix} V_{xx}(x,y) \\ V_{yy}(x,y) \end{pmatrix}$$

are matrices.

The subscripts of V denote partial derivatives with respect to the index $i = x,y$. The HJB equation (12) can be rearranged as:

$$\delta V = \max_{h_x,h_y\geq 0}\ \{\pi(y,h_x,h_y)+(f(x,y)-h_x)V_x+(g(x,y)-h_y)V_y + \frac{1}{2}(\sigma_1 x)^2V_{xx}+\frac{1}{2}(\sigma_2 y)^2V_{yy}\} \tag{13}$$

After substituting $f(x,y) = a_1x^2 - a_2x^3 + a_3xy$ and $g(x,y) = b_1y^2 - b_2y^4 + b_3xy$ and

$$\pi(y,h_x,h_y)=p_1h_x-q_1h_x^{\alpha_1}+p_2h_y-\frac{q_2}{y}h_y^{\alpha_2}-p_3h_y^2$$

equation (13) yields:

$$\delta V = \max_{h_x,h_y\geq 0}\left\{p_1h_x-q_1h_x^{\alpha_1}+p_2h_y-\frac{q_2}{y}h_y^{\alpha_2}-p_3h_y^2-h_xV_x-h_yV_y\right\} + (a_1x^2-a_2x^3-a_3xy)V_x+(b_1y^2-b_2y^4+b_3xy)V_y+\frac{1}{2}(\sigma_1 x)^2V_{xx}+\frac{1}{2}(\sigma_2 y)^2V_{yy} \tag{14}$$

Optimal solution can be derived by solving the HJB equation (14). The inner optimum with respect to controls (h_x and h_y) are given in Appendix A. While it is difficult or impossible to solve analytically the HJB equation together with boundary conditions, we solve it using numerical approximation methods.

Numerical approximation approach

Our model is a two-dimensional model and is strongly nonlinear in control. Analytical solutions are usually not available to such problems and it is difficult to solve the Hamilton-Jacobi-Bellman (HJB) equation together with nonlinearity and given boundary conditions. Numerical approximation methods are the only viable alternatives. The Markov chain approximation approach, based on probability theory, is one of the most effective methods (Song 2008). Numerical algorithms for optimal stochastic control problems of this kind can be found in Kushner and Dupuis (2001). While dealing with the convergence of the numerical methods, it is shown that the value functions to which our approximations converge are the optimal value functions.

The numerical technique entails discretizing the state space for the HJB control problem (14), constructing transition probabilities for the controlled Markov chain by applying finite difference techniques and then iterating on the HJB equation with initial guess V_0 for the value function. The combined approximation in policy space and in value space is more powerful and faster as the value function is updated with the new policy at each step. The iteration is carried out until the value function converges to the optimal value functions (for details of the approximation refer to Kushner and Dupuis 2001).

Application to cod and capelin species in the Barents Sea

The Barents Sea is one of the most productive ocean areas in the world (O'Brien et al. 2004), and it harbors two key species: capelin (*Mallotus villosus*), the plankton feeder and northeast arctic cod (*Gadus morhua*), the main predator of capelin. Cod is considered to be the main basis of the Norwegian commercial white fish industry (Kugarajh et al. 2006), while capelin is the largest pelagic fish species in the Barents Sea. Capelin is also potentially the largest stock in the world (Gjøsæter 1998, Gjøsæter and Bogstad 1998) and is of crucial importance as a prey for the growth of juvenile cod (Hamre 2003, Dalpadado and Bogstad 2004). The relationship between these two species is highly dynamic and is essential in the Barents Sea ecosystem (Bogstad et al. 1997).

Management of fisheries in the Barents Sea already includes species interactions to some degree. The importance of multispecies management in the Barents Sea was realized after the capelin collapse in the mid-1980s and subsequent dramatic effects on cod (e.g., cannibalism) and other species, such as the mass migration of harp seals (*Phoca groenlandica*) to the coast of Norway where many of them drowned in fishermen's nets (Haug and Nilssen 1995). Following this, a large multispecies research program was initiated. The predation by cod on mature capelin is included into the assessment of capelin and cod cannibalism; and cod predation on haddock is included in the assessment of cod species (ICES 2004). Furthermore, cod recruitment and survivability are directly affected by climatic environment such as temperature, spawning season, and the availability of food, such as prey stocks. Higher temperature at spawning time and more capelin have a positive effect on cod recruitment. However, high temperature results in a decrease in capelin biomass through high herring recruitment, which affects cod recruitment negatively through cannibalism (Hjermann et al. 2007). Given these uncertainties, we choose a stochastic growth model consisting of capelin and cod as a foundation for decision-making in multispecies management.

Specification of biological and economic parameters

Since the purpose of the paper is to extend and compare findings with previous studies, both the biological and economic parameter values are obtained from Agnarsson et al. (2008) and Sandal and Steinshamn (2010), and they are specified below:

Growth for capelin in single species model:

$$r_1 x^2 (1 - \frac{x}{k_1}) = 0.00021781 x^2 (1 - \frac{x}{8293}) \quad (10^6 \text{ kg per year})$$

Growth for cod in single species model:

$$r_2 y^2 (1 - \frac{y}{k_2}) = 0.000665 y^2 (1 - \frac{y}{2473}) \quad (10^6 \text{ kg per year})$$

Growth for capelin species in multispecies model:

$f(x,y) = a_1 x^2 - a_2 x^3 - a_3 xy = 0.00018x^2 - 1.19e^{-8} x^3 - 0.00021xy$ (10^6 kg per year)

Growth for cod species in multispecies model:

$g(x,y) = b_1 y^2 - b_2 y^4 + b_3 xy = 0.00022y^2 - 3.49e^{-11} y^4 + 1.82e^{-5} xy$ (10^6 kg per year)

Demand for capelin:

$p_x(h_x)h_x = p_1 h_x = 1h_x$ (10^6 NOK with price $p_1 = 1$ NOK per kg)

Demand for cod:

$p_y(h_y)h_y = (p_2 - p_3 h_y)h_y = (12.65 - 0.00893h_y)h_y$ (10^6 NOK with price $p_2 =$ 12.65 NOK per kg)

Cost for catch of capelin:

$c_x(h_x) + q_1 h_x^{\alpha_1} = 0.07 h_x^{1.4}$ (10^6 NOK)

Cost for catch of cod:

$$c_y(y,h_y)=q_2\frac{h_y^{\alpha_2}}{y}=5848.1\frac{h_y^{1.1}}{y}\quad (10^6\ \text{NOK})$$

It is also worth mentioning that the price of cod (predator) is much higher compared to capelin (prey) species. We also assume a constant cost of harvesting for capelin, which is independent of the stock size because of the schooling nature of the species (Aanestad et al. 2007, Sandal and Steinshamn 2010). Moreover, all the optimal feedback solutions are calculated with 5% discount rate ($\delta = 0.05$) and with different levels of stochasticity, where the stochastic parameters are defined exogenously.

Results and discussion

Optimal harvest in deterministic growth models

Single species model

The single species deterministic growth model specified in equation (1) is employed to obtain the results for each species individually. In a single species capelin model, it is suboptimal to harvest if the stock biomass level is below 1.2 million t (Fig. 1a) because the future profit gain from conserving the stock will be higher. The harvest can be increased gradually if the stock biomass is between 1.2 and 2.5 million t. It is optimal to harvest capelin as a "bliss" or static optimum if the stock reaches over 2.5 million t.

The optimal harvest in cod species is more conservative than in capelin species. The moratorium is around 1.2 million t of biomass near the maximum sustainable yield (1.5 million t). There are several reasons for a higher moratorium level in cod. First, there could be low net revenue from the harvest due to stock-dependent higher harvesting costs if harvested at a low stock level (if fixed costs are assumed in the model, which could also lead to a higher moratorium). Second, there is a possibility of higher future gain from conserving the stock. Although we don't look at the risk aversion case, the downward sloping demand curve has the same effect as risk aversion. Furthermore, a downward sloping demand curve will give smoother but continuous harvests. The curve (Fig. 1b) shows that it is optimal to increase harvest gradually with an increase in the stock size up to 2 million t.

Multispecies model

The deterministic growth model specified in equation (3) is employed to find an optimal solution for capelin and cod species simultaneously.

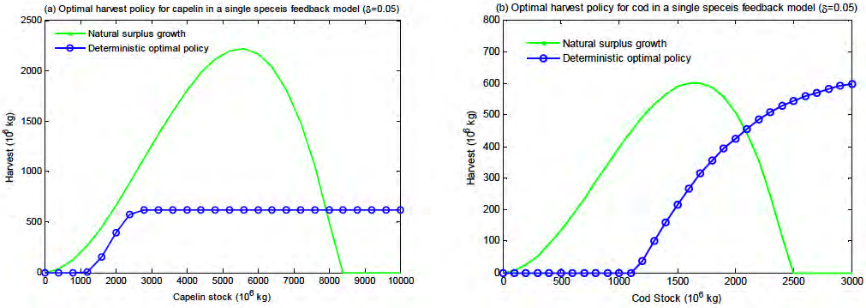

Figure 1. Single species deterministic optimal harvest policy: (a) Capelin. (b) Cod.

The optimal feedback solution for capelin and cod harvest in the two-dimensional cod-capelin state space is presented in Fig. 2.

The optimal harvest policy is considerably affected in capelin due to the interaction with cod except for zero cod stock. At zero cod level, the harvest policy is similar to the single species capelin model, as expected. The harvest pattern of capelin for a large cod stock level is less intuitive (see Fig. 2a). The pattern consists of considerable harvest at low capelin stock levels, then a moratorium over a certain range followed by a gradual approach to the static optimum. The intuitive explanation of harvest at low stock levels is that it would go extinct because of cod predation. Sandal and Steinshamn (2010) have clearly shown that the presence of cod in the model induces critical depensation along the optimal paths. In other words, if the capelin biomass level is below 2 million t, it goes extinct even without harvest due to cod predation. Therefore, it is optimal to increase the harvest of capelin because if they are not harvested for human consumption they will be eaten by cod anyway. This occurs only if the cod stock is sufficiently abundant and therefore its commercial existence (profitability) does not depend too much on its present feeding on capelin. Hence, whether capelin is harvested or not, it does not reduce the value of the cod fishery as it approaches the single species fishery. When capelin stock increases over 2 million t, it should be conserved for cod. At this level, cod can be harvested in large amounts so that profit will be increased due to the higher cod price.

Sandal and Steinshamn (2010) also show that the "valley region" in Fig. 2a means capelin stock could go extinct if it is below roughly 4 million t and could rebuild if it is over 4 million t. If the capelin stock

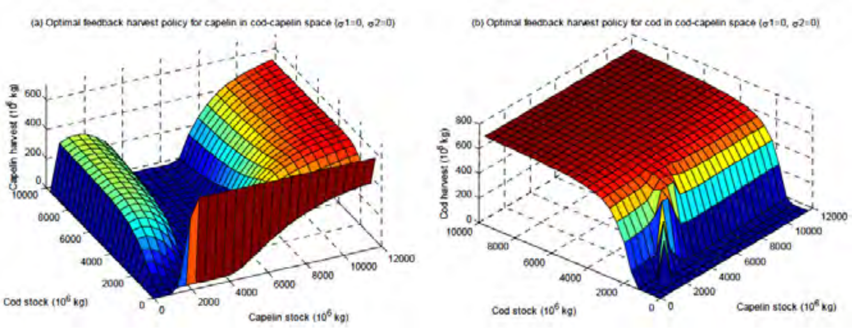

Figure 2. Multispecies deterministic optimal harvest policy: (a) Capelin. (b) Cod.

increases to over 6 million t, it can be harvested at bliss or myopically as in the single species solution.

The valley region reflects at least two properties in the bioeconomic system. It represents an intrinsic precautionary policy trying to avoid capelin from going extinct and at those ranges of stocks it is more profitable to harvest the capelin through the cod. That is, the value created by letting cod eat a unit of capelin is higher than the value created by landing that unit.

On the other hand, species interaction has a minor effect on the harvest policy for cod species. Due to interaction, the cod moratorium shifts toward a higher biomass level compared to a single species model at low capelin stock level. For instance, the moratorium level is 1.4 million t in the multispecies model compared to 1.2 million t of biomass in a single species model. This is because of low food availability for cod in the ecosystem. While the capelin stock becomes large (over 2 million t), the cod moratorium level remains unaffected in the two species interaction model. The single species model suggests exploitation at a lower level because it does not take account of food availability of the cod stock. Furthermore, with an increase in the cod biomass level, a higher exploitation is possible with more capelin in the ecosystem because of the increased food availability for cod.

In conclusion, the multispecies management approach makes it possible to increase the total profit by managing the stocks simultaneously. There is a possibility of harvesting capelin at the same level as the single species optimal level if the capelin stock is maintained to a higher biomass level (>6 million t), while we can observe that a sufficiently large catch of cod stock is possible in the cod-capelin model compared to the single species. The single species model imposes constraint on the ecosystem (i.e., the stocks grow as though they are in a multispecies model but we ignore the interaction terms when predicting the following year's stock and therefore harvest) and the profits will be less than or equal to multispecies management, which does not impose that constraint.

Optimal harvest in stochastic growth models

Single species model

The single species stochastic growth model specified in equation (2) is employed to obtain the optimal policy and is compared with the deterministic solution at various levels of stochasticity. Fig. 3 shows the optimal solutions for different levels of stochasticity for capelin and cod.

A high level of stochasticity in capelin leads to a more conservative harvest compared to the deterministic case. Compared to the sharp increase from moratorium level to bliss in the deterministic case, the optimal solutions approach the bliss level more gradually in the stochastic case. At a high level of stochasticty (for example $\sigma_0 = 0.7$), it is optimal to harvest myopically at low stock levels because there is a significant possibility that the stock could go extinct due to stochasticty. But one should be more conservative if the stock becomes large (>1.2 million t) because there is only a small chance of extinction of the large stock even under high stochastic growth. With a fairly large stock (>7 million t), the optimal harvest policy for capelin remains more or less unaffected for all levels of stochasticity (Fig. 3a).

For stochastic levels below 0.5, there is a very small effect on optimal harvest of cod. For example, we see that the curves in the deterministic and stochastic cases stay close together and for practical purposes they can be regarded as identical. Due to the stock dependent harvest cost, the moratorium is large in cod and a large stock is relatively less influenced by stochasticity. For an increased level of stochasticity ($\sigma_0 = 0.7$), the harvesting strategy becomes myopic for smaller stock levels and conservative in the case of large stocks. But at fairly high levels of stochasticity ($\sigma_0 = 1.5$), it is optimal to harvest myopically at all stock levels (see Fig. 3b).

Multispecies model

As in the single species model, we have employed different levels of stochasticity to study the effect on the optimal harvest policy. We have also observed the effects in three different cases: with prey stochasticity, with predator stochasticity, and stochasticity in both species.

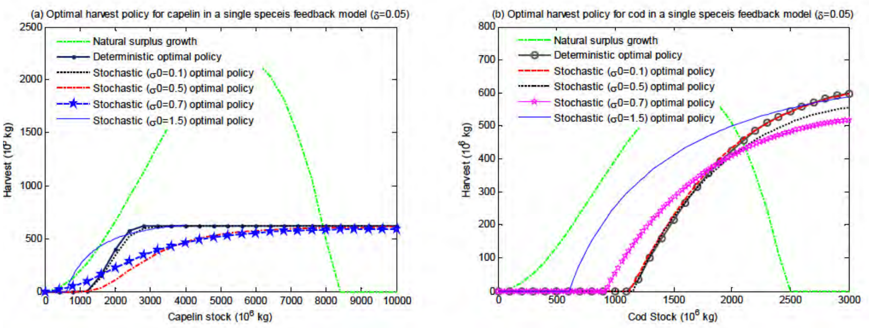

Figure 3. Effect of stochastic growth for optimal harvest policy in single species models: (a) Capelin. (b) Cod.

Stochastic growth in capelin (prey) species

We have employed equation (6.1) to obtain optimal feedback policies in the multispecies model where capelin has stochastic growth. The optimal solution for exploitation of capelin and cod in a two-dimensional state space with different levels of stochasticity in capelin is presented in Fig. 4.

With an increased stochasticity in capelin, the optimal harvest of capelin becomes more conservative compared to the deterministic optimal policy. For low levels of stochasticity (for example, $\sigma_1 = 0.1$), the valley region that appears in the deterministic case becomes larger, suggesting no harvest of capelin in this particular region. By not harvesting capelin, we can harvest the high priced cod that yields higher net revenue. But in a region with a very large capelin stock, it will be optimal to harvest capelin as bliss along with harvest of cod. A further increase in stochasticity (for example, $\sigma_1 = 0.3$) makes the valley region much wider, suggesting that one should be conservative in the capelin harvest even for very large biomass levels.

There is no strong influence from stochasticity on the optimal harvest of cod as long as the level of stochasticity is less than 0.3. The main reason is that uncertainty is managed by reducing the harvest of capelin. At higher level of stochasticity (for example $\sigma_1 > 0.5$), the valley region in capelin gets narrower. This suggests that capelin should be harvested earlier compared to the low stochastic growth model, because conserving the highly stochastic capelin stock may not contribute to the growth of cod and subsequently its harvest. However, one should still be conservative and save the capelin stock from extinction at large biomass levels due to stochasticty and natural predation by cod.

As stochasticity increases to 1 ($\sigma_1 = 1$), the valley region disappears but the optimal exploitation level is clearly different from the deter-

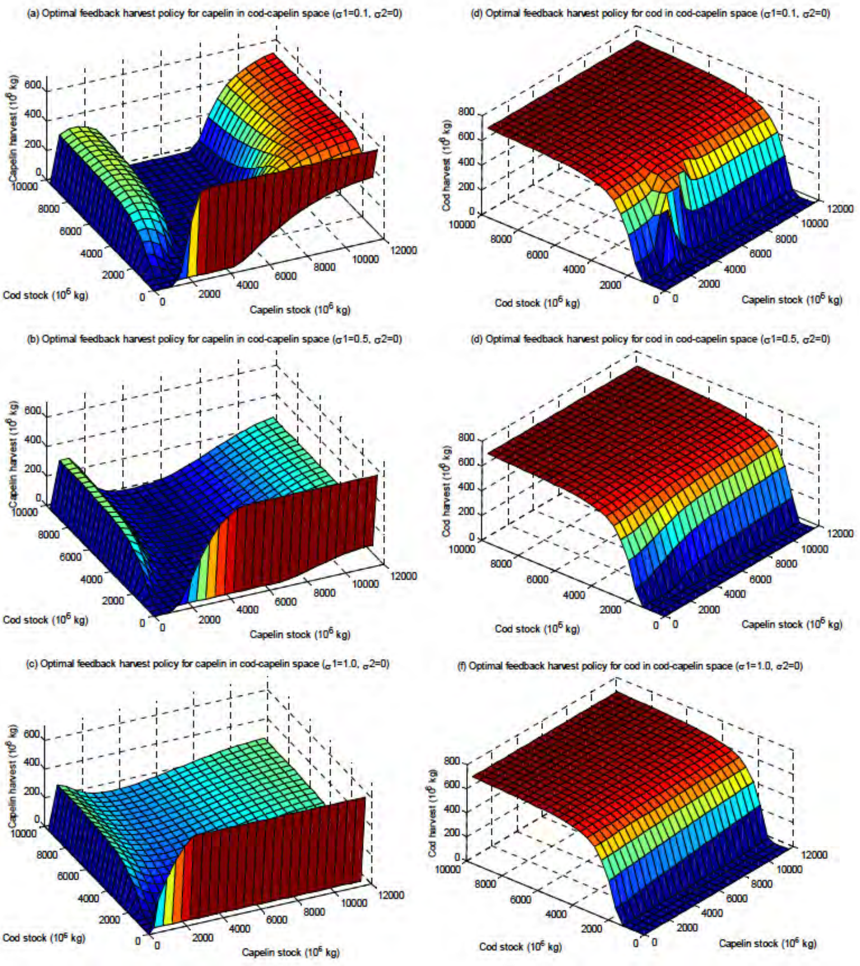

Figure 4. **Effect of capelin stochasticity in optimal harvest: (a) Capelin with $\sigma_1 = 0.1$. (b) Capelin harvest with $\sigma_1 = 0.5$. (c) Capelin harvest with $\sigma_1 = 1$. (d) Cod harvest with $\sigma_1 = 0.1$. (e) Cod harvest with $\sigma_1 = 0.5$. (f) Cod harvest with $\sigma_1 = 1$.**

ministic solution. The optimal harvest of capelin is less conservative when the stock is low and becomes conservative at large stock levels. The reason is that a small stock of capelin may not be utilized by cod to a profitable level but a large stock of capelin could be utilized by cod that increases net profits. Therefore, it is optimal to be conservative for the low priced capelin and let cod prey on it when the capelin stock is high. But for very high stochasticity ($\sigma_1 > 1.0$), there is no gain of conserving a highly stochastic capelin stock for the purpose of cod feeding. Therefore, capelin should be managed as in a highly stochastic single species model, and the cod stock should also be managed as in the single species model (not shown in the graph).

Stochastic growth in cod (predator) species

We have employed equation (6.2) to obtain the optimal harvesting rules in the multispecies model where cod has stochastic growth. Optimal solutions for exploitation of capelin and cod stocks in two-dimensional state space for different levels of stochasticty (σ_2) are presented in Fig. 5.

It can be noted that a small level of stochasticty in cod (for example $\sigma_2 = 0.1$) affects the harvest of capelin stocks slightly by widening the valley region in the deterministic solution. An increased level of stochasticity (for example $\sigma_2 = 0.5$) further widens the moratorium (the valley) with almost no harvest of capelin stocks until they are above 7.5 million t. This is very intuitive because the stochastically growing cod needs more food so that cod can be harvested unaffected. For stochasticity below 0.5, cod harvest is unaffected but capelin harvest changes. At a substantially high stochasticty level (for example $\sigma_2 = 1.0$) in cod, the capelin should be harvested as in a single species deterministic model because it does not pay off to feed a non-stochastic species to a highly stochastic species. Therefore, the highly stochastic cod should be harvested as in the single species solution and at a myopic level.

Stochastic growth in both species

The effect of stochastic growth of prey and predator species to optimal harvest in a multispecies model is obtained by using equation (6). Optimal exploitation policies with different levels of stochasticty (σ) for capelin and cod in a two dimensional state space is presented in Fig. 6. To avoid many plots, we have only plotted the result for cases with $\sigma_1 = \sigma_2 = \sigma$ to capture the general features in the result.

At low levels of stochasticity (for example σ_1, $\sigma_2 < 0.3$), the capelin species should be harvested in a conservative way in order to allow cod to feed on it so that cod can be harvested unaffected. On the other hand, with an increasing level of stochasticity (for example $\sigma_1 = \sigma_2 = 0.5$), the capelin harvest should be more conservative. At this stochasticity level, the stochastic cod requires more food than the less stochastic cod and therefore it is optimal to be very conservative in harvest of capelin at

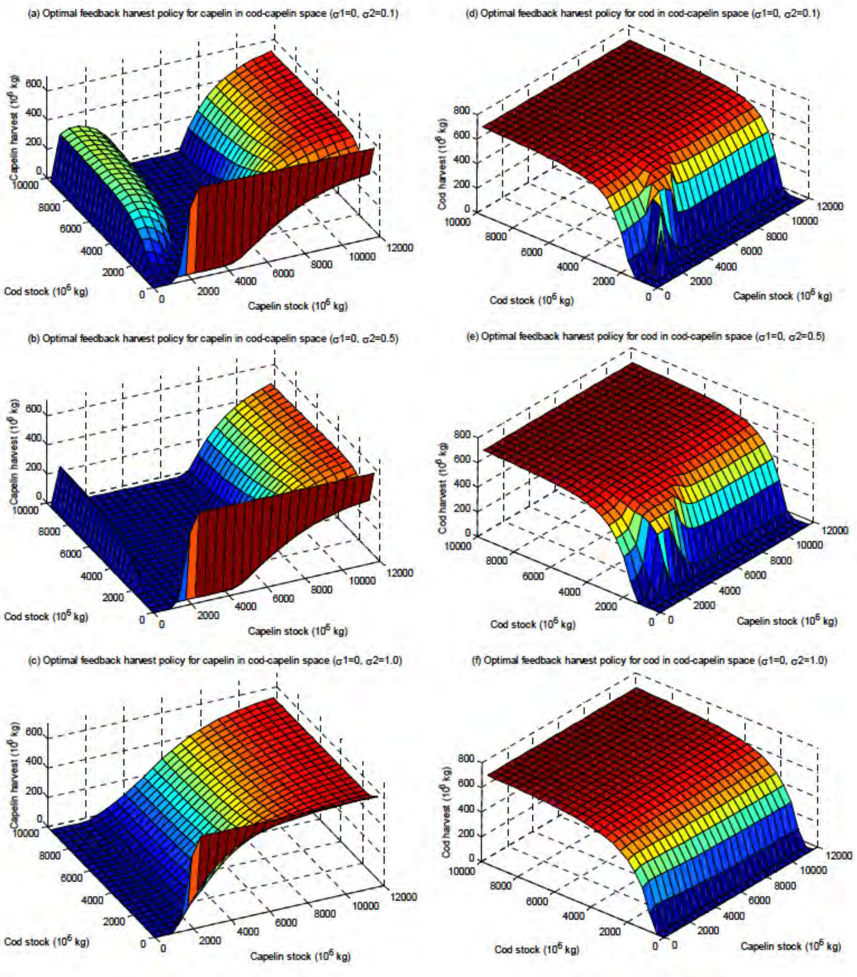

Figure 5. Effect of cod stochasticity in optimal harvest: (a) Capelin harvest with $\sigma_2 = 0.1$. (b) Capelin harvest with $\sigma_2 = 0.5$. (c) Capelin harvest with $\sigma_2 = 1$. (d) Cod harvest with $\sigma_2 = 0.1$. (e) Cod harvest with $\sigma_2 = 0.5$. (f) Cod harvest with $\sigma_2 = 1$.

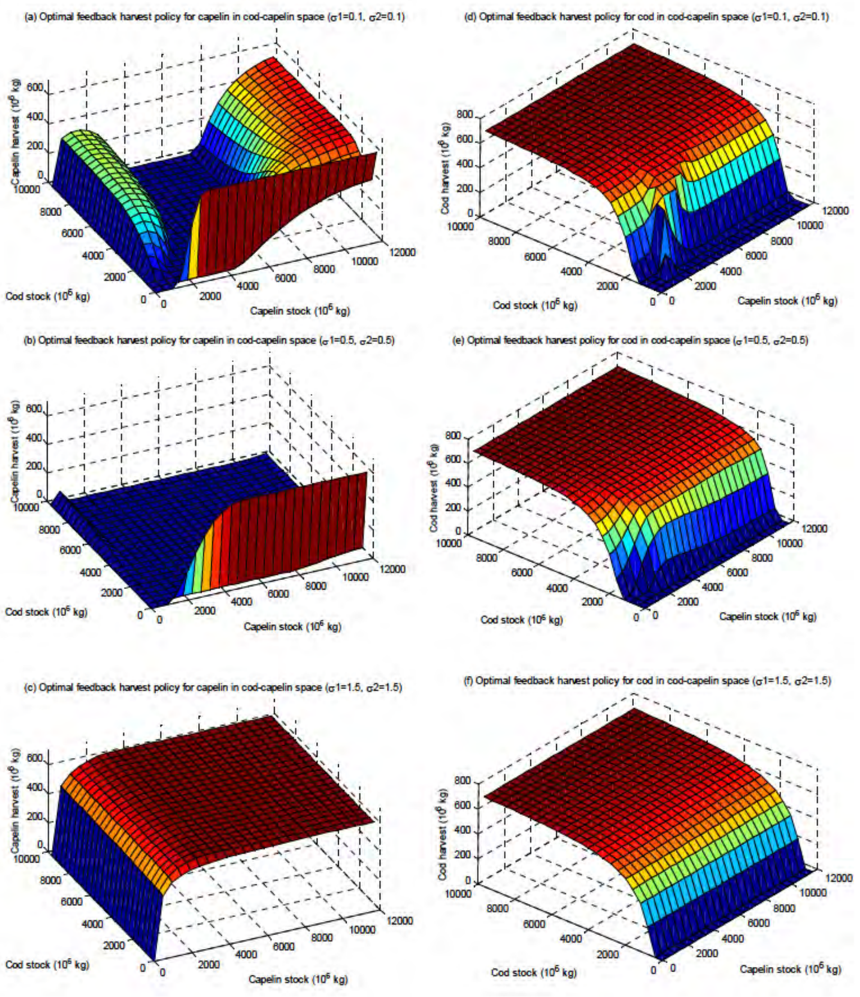

Figure 6. Effect of stochasticity in optimal harvest: (a) Capelin harvest with $\sigma = 0.1$. (b) Capelin harvest with $\sigma = 0.5$. (c) Capelin harvest with $\sigma = 1$. (d) Cod harvest with $\sigma = 0.1$. (e) Cod harvest with $\sigma = 0.5$. (f) Cod harvest with $\sigma = 1.5$.

any level except at zero cod level, and cod harvest should also be more conservative to ensure better growth under this level of stochasticity. However, at a high level of stochasticity (for example $\sigma_1 = \sigma_2 = 0.7$), the capelin stock should be harvested conservatively. For a substantially higher level of stochasticity (for example σ_1, $\sigma_2 > 1$), it should be harvested myopically at small stock levels and conservatively at large stock levels. This is similar to the single species stochastic solution. The cod stock should be harvested as in the single species stochastic solution and one should be myopic when the stock is relatively small. In both species the possibility of extinction of a small stock is high due to stochasticity and the capelin extinction risk is further exacerbated by cod predation.

Looking at the three different cases—stochasticity in prey species, stochasticity in predator species, and stochasticity in both species—at different levels of stochasticity, we observe that there are various effects on the optimal harvests compared to the deterministic setting. Effects naturally depend on the stock size. If the stock is small, stochasticity has a more pronounced effect.

The stochastic growth either in prey species or in predator species or in both species affects only the prey species at low stochasticity levels (σ_1, $\sigma_2 < 0.3$). There are three possible reasons. First, by conserving the low priced prey species as a food for the higher priced predator, profits can be increased. Second, due to stock independent costs of harvest, there is a low moratorium level for the prey (capelin) in the deterministic solution and the stock becomes more sensitive with increasing stochasticity. Third, cod predation makes the prey more sensitive to optimal exploitation.

In the deterministic and small stochasticity case, we observed that it is possible to harvest capelin at exactly the same level as in the single species solution for large stock levels, and a sufficiently large harvest of cod stock in a cod-capelin multispecies model compared stochastic single species model. In other words, there is an advantage of multispecies management when the stock is large.

Although we can observe a higher harvest of capelin in the single species solution compared to the multispecies solution, it is important to note that it is profitable to increase the harvest of cod by one kg while sacrificing up to 12 kg of capelin because of the price difference. Furthermore, our model suggests that capelin should be conserved at low to moderate levels of stochasticity in all three cases because the increase in harvest of cod is more profitable compared to harvesting both species conservatively in a single species model under the assumption of stochasticity. The general impression is that ecosystem management is more profitable than managing individual stocks separately when there is limited uncertainty, because the single species model imposes constraint on the ecosystem and the profits will be less than or equal to multispecies management that does not impose that constraint.

At moderate to high stochasticity levels (for example 0.3-0.7), the effect on optimal harvest strategy cannot be generalized as the effect depends on the stock size and the stochastic species. In general, effects are stronger at low stock levels.

Furthermore, at a substantially high stochasticity level (for example 1.0), the stochastic species should be harvested myopically and the deterministic species should be harvested as in the single species model. If both species are highly stochastic, they should be harvested myopically. This means that with very high stochasticity, the effect of multispecies interaction cannot be observed. Although an assumption of a very high level of stochasticity may not be observed in real-world fisheries, with the precautionary principle in mind, we still found it instructive to study such high levels of stochasticity in the model. However, we put emphasis on the effect of stochasticity at a low to moderate levels of stochasticity, which is important in real-world fisheries management.

Optimal harvest versus historical harvest

We compared the optimal solutions from four different models with the actual harvest data over the last 33 years. The optimal harvest and actual harvest are presented in Figs. 7 and 8. It can be noted that the actual harvest is somewhat higher compared to the optimal harvest models during most of the periods in both stocks.

The capelin curves reveal a dramatic overfishing of the capelin fishery until 1987 before the closure of the catch (Tereshchenko 2002). In a later period, it is close to the identified single species deterministic optimal catch level. Looking at the different multispecies optimal harvests, some models suggest that there was underexploitation during 1994-1999 and after 2004. However, this is not pronounced compared to the case of overexploitation.

A similar overexploitation over the whole period is revealed in cod species compared to optimal catch identified in different models. However, the trends of historical actual catch and optimal catch are analogous over most of the periods. It is interesting to note that the very high stochastic model most closely predicted the historical harvests. Since the optimal harvests in the very high stochastic model were myopic single species management policies, this implies that we expect these to have been the policies in place at the time.

The general overexploitation of the fish stocks in Barents Sea could be the result of a policy that aims at an MSY or maximum sustainable yield (Agnarsson et al. 2008) from biological point of view and the economic aspect might have been ignored. Moreover, MSY is a single species concept and it is normally infeasible to have both stocks at MSY simultaneously.

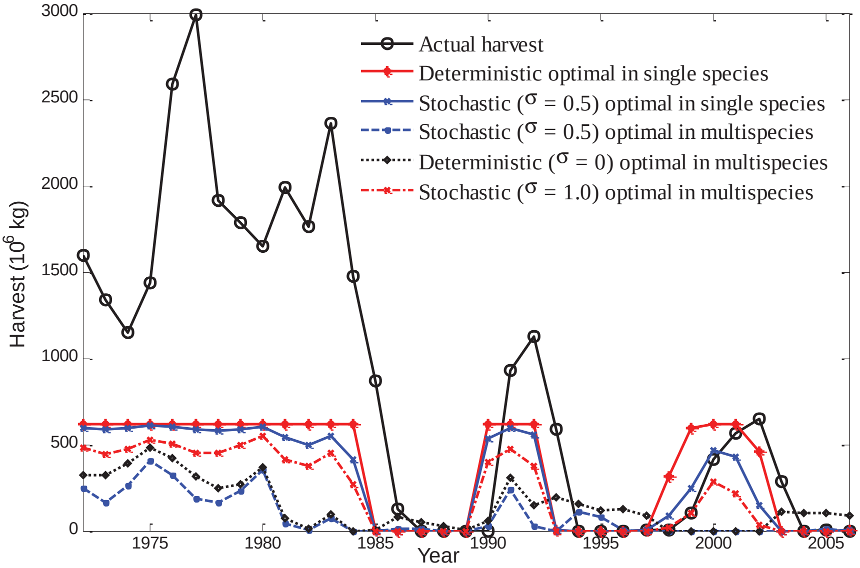

Figure 7. Actual versus optimal harvest of capelin species in different modeling approaches.

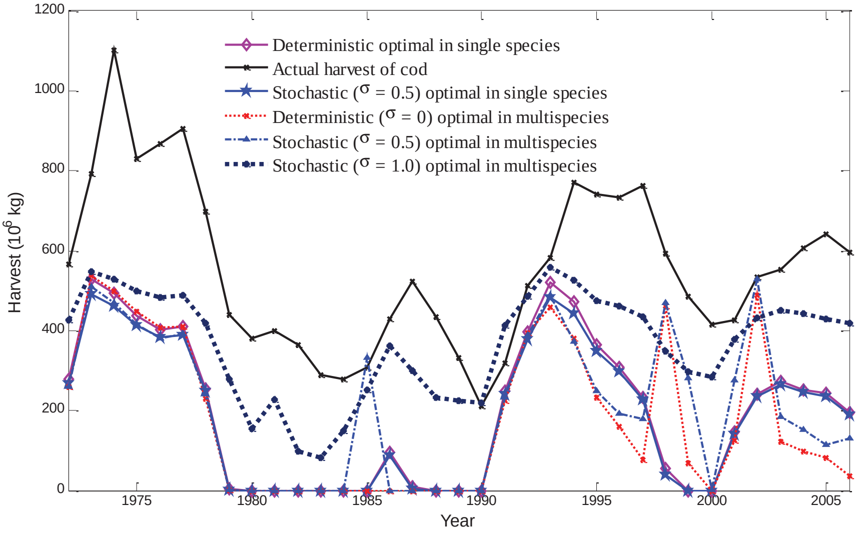

Figure 8. Actual versus optimal harvest of cod species in different modeling approaches.

Concluding remarks

We have demonstrated the applicability and usefulness of dynamic programming to multispecies management under stochasticity. While biological predator-prey interactions may have been well understood, much remains to be done for a comprehensive understanding of their economic consequences. Our approach sheds light on the interaction of economics with a complex biology.

Despite that the eco-biological system is relatively complex, the nature of our findings is intuitive. It is relatively more profitable to manage the ecosystem in a stochastic multispecies fishery than individual stocks separately. Our results show fundamentally different conclusions with and without stochasticity. We conclude that ignoring biological interactions never makes sense and that the level of stochasticity will have an impact on the optimal strategy for management.

We believe that policy makers would benefit from an increased appreciation of the effects of stochasticity and the consequences of ignorance. The ignorance becomes apparent when we compare our solutions to historical catches. In some periods, landings lie closer to the myopic policy than to the optimal dynamic feedback policy both in the stochastic and deterministic cases.

Although the general result might not reflect the best management due to the exogenous parameters adopted in the work, our paper is a breakthrough in complicated, stochastic multispecies modeling which combines both biology and economics and provides a basis to answer what the optimal response is for a bioeconomic manager of renewable resources.

The estimation of stochastic parameters in a multispecies model is a very difficult and complex task. But at the same time its accurate estimation is crucial for further improvement of the model. The stochastic parameter estimation, therefore, remains to be a future scope for the identification of optimal policy for stochastic multispecies fisheries. Similarly, the relaxation of the assumption of uncorrelated stochasticity or the noise among the interacting species could improve the model considerably.

Finally, we submit that real world fisheries management is concerned with a limited number of fish stocks that are only a part of a larger ecosystem. To analyze the effect of stochasticity and to assess risk of stock collapse in an ecosystem framework is beyond our scope here. We are only on one of the first steps on a long ladder toward the ultimate goal of ecosystem management under uncertainty. But, it is an important step in the right direction.

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Appendix A: Inner optimum with respect to controls for a multispecies model

The inner optimum can be obtained from the HJB equation (A1) with respect to control (harvest):

$$(A1)\qquad \delta V = \max_{h_x, h_y \geq 0} \left\{ p_1 h_x - q_1 h_x^{\alpha_1} + p_2 h_y - \frac{q_2}{y} h_y^{\alpha_2} - p_3 h_y^2 - h_x V_x - h_y V_y \right\}$$
$$+ (a_1 x^2 - a_2 x^3 - a_3 xy) V_x + (b_1 y^2 - b_2 y^4 + b_3 xy) V_y + \frac{1}{2}(\sigma_1 x)^2 V_{xx} + \frac{1}{2}(\sigma_2 y)^2 V_{yy}$$

The inner optimum for prey species is

$0 = p_1 - \alpha_1 q_1 h_x^{\alpha_1 - 1} - V_x$ or

$$(A2)\qquad h_x = \left[\frac{p_1 - V_x}{\alpha_1 q_1} \right]^{\frac{1}{\alpha_1 - 1}}$$

And the inner optimum for predator species is found by solving the following first order condition (FOC) equation numerically.

$$p_2 - \frac{q_2 \alpha_2 h_y^{\alpha_2 - 1}}{y} - 2 p_3 h_y - V_y = 0$$

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Ecosystem-Based Fisheries Management in the Shiretoko World Natural Heritage, Japan

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Abstract

People have been living in the Shiretoko Peninsula for thousands of years, and a main local industry in this area has been fishing. Humans are an integral component of the local ecosystem, rather than unwanted extras to be eliminated from the "original ecosystem." Therefore, unless the objective of ecosystem conservation is to go back to the original state dozens of centuries ago, sustainable harvesting of a wide range of species is an integral part of ecosystem structure in this area.

In the Heritage Integrated Plan for the Shiretoko Peninsula, local fishers have core roles in ecosystem conservation, especially in ecosystem monitoring. Also, to facilitate cross-sector coordination (e.g., between the fishery and tourism sector), new management organizations were established by the national government. Key components to the development of effective management were participation by the local fishery sector from the beginning of the planning process, scientific support from researchers, and accountability of administrators. With this process, total administrative cost for implementing ecosystem conservation measures was considerably decreased. Experiences from this ecosystem-based management approach may benefit managers in other countries where large numbers of small-scale fishers take a wide range of species under a fisheries co-management regime.

Introduction

Objective

Japan (Fig. 1) is one of the world's largest fish-consuming countries with a long history, and has developed its own customs and values in

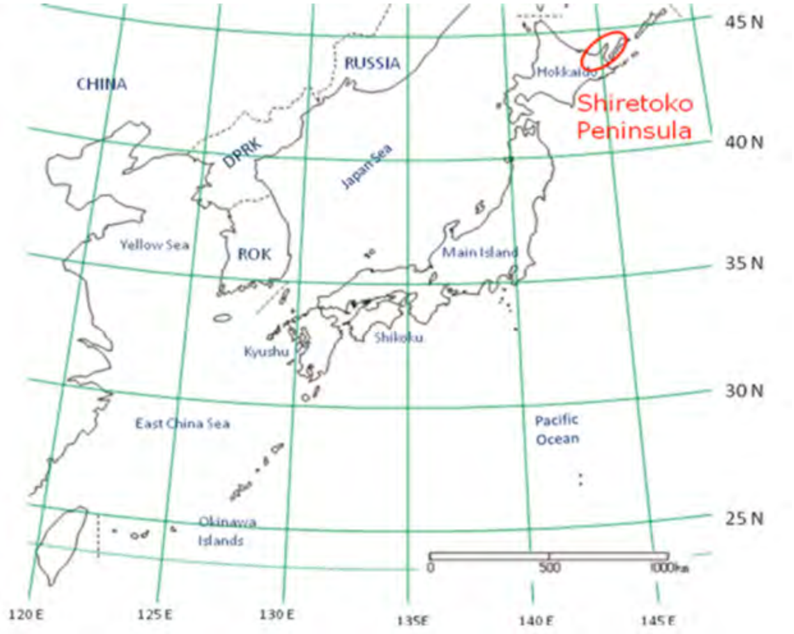

Figure 1. Location of Shiretoko Peninsula in Hokkaido, Japan.

terms of managing fisheries resources (Makino 2011). This study first introduces an outline of Japanese fisheries, including production, consumption, and institutional structure for the fisheries management. Then, as a case study of coastal fisheries management, the Shiretoko area experience is introduced. Finally, new activities after inscription to the UNESCO World Natural Heritage List are analyzed, with a brief estimation of additional costs for required new commitments.

People have been living in Japan for thousands of years, and local fishers have implemented various autonomous measures for sustainable fisheries. This existing institutional structure can be the base for ecosystem-based fisheries management. The objective of this study is to show that the ecosystem-based fisheries management framework established in the Shiretoko area is not a revolution, but an expansion from an existing fisheries management framework. This analysis shows that such an approach can considerably save cost in ecosystem conservation.

Fisheries production

Japan is one of the world's largest fishing nations based on both production and consumption. In 2006, fisheries produced 5.6 million tons, in

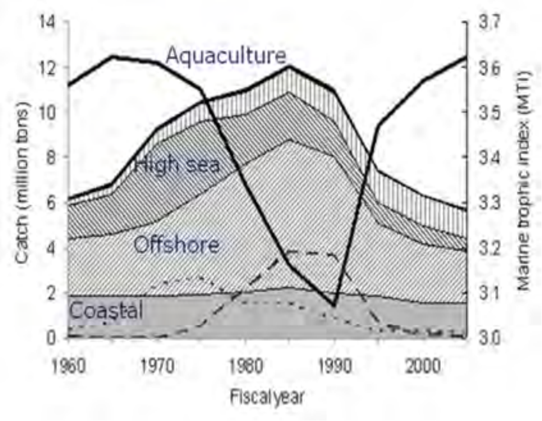

Figure 2. Marine trophic index (solid line) and the total production of Japanese marine fisheries from fiscal year 1960 to 2005, divided into five categories: coastal fisheries, offshore fisheries, far sea fisheries, marine aquaculture, and inland aquaculture from bottom to top. Broken and dotted lines represent catch of sardine and walleye pollock, respectively. (Modified from Matsuda et al. 2010.)

terms of both capture and aquaculture. It was ranked 5th in the world, following China, Peru, USA, and Indonesia (FAO 2008). Another feature of Japanese fisheries is that many species make up most of the catch. According to the five year average (2002-2006) of capture fisheries production in the FAO FISH STAT data, 33 species account for 90% of the total catch volume (4.4 million tons) in Japan. In comparison, 46 species constitute 90% of the 0.9 million ton catch in Spain. At the other end of the spectrum, in high-latitude northern European countries such as Norway, 90% of the total catch (2.7 million tons) consists of only eight species: blue whiting, Atlantic herring, Atlantic cod, coley, capelin, brown seaweed, Atlantic mackerel, and haddock (Makino and Matsuda 2011).

In the late 1960s and early 1970s, far sea fisheries were the most important fishery sector in Japan (Fig. 2). However, after establishment of 200 nautical mile Exclusive Economic Zones (EEZs) worldwide, the importance of this type of fishery drastically declined. To partially compensate, in the late 1970s and 1980s offshore fisheries were developed. The total volume of Japanese catch peaked in 1984 at 12.8 million tons. The main species in volume for this period was the Japanese sardine *(Sardinops melanostictus*) but because of natural fluctuations in biomass, this species' landing declined severely in the early 1990s (Watanabe et al. 1995, Yatsu et al. 2005). On the other hand, coastal fisheries have shown relatively stable production since the 1960s, with a slight decline over the last 20 years, and they have occupied the most important sector in Japanese production since the late 1980s. During this period, marine aquaculture developed considerably and now it is the second largest sector in terms of production value.

Fishing employment

The number of people engaged in fisheries production in 2007 was 204,000. This number has continuously decreased since 1953, when it was about 800,000. The aging of fishers is also a serious problem with currently 48% of them being over 60 years old. According to the Norinchukin Research Institute (2008), only 24.3% of fishers have a replacement to take over their business, while 59.5% have no one. Because the number of new fishers is very small (about 1,200-1,500 people per year), both a decrease in the total number of fishers and an increase of their average age will continue in the near future.

The total number of fishing vessels registered in 2007 was 313,000, of which 9,000 were non-powered and 267,000 were of less than five gross tons (Fisheries Agency 2009). Compared to other developed countries such as Iceland, New Zealand, Canada, and USA, most Japanese fishers are small-scale operators in coastal areas (Makino and Matsuda 2011).

Seafood consumption

Japanese annually consume large quantity of seafood; the average Japanese eats 80.2 grams of protein per day, of which 43.9 g are fish products. Fish products are the second largest source of total protein intake, and the largest source of animal protein intake in Japan.

Fish products are mainly served five ways in Japan: sashimi (raw fish), yaki-zakana (grilled fish), ni-zakana (boiled fish), himono (dried fish), and tempura (deep-fried fish)—sashimi being the favorite. Japanese sushi is a combination of sashimi or vegetables with vinegar rice. The most important factor in the quality of sashimi is its freshness. Yaki-zakana (grilled fish) also requires fresh fish, although it is said that freshness is less important for ni-zakana (boiled fish). Therefore, in the Japanese domestic market, prices for similar size and species differ considerably according to fish freshness. Japanese eat fish and rice almost every day. The average household spent ¥88,593 on fish products in 2008, with 59% of them fresh fish (Ministry of Internal Affairs and Communications 2009).

Institutional framework for fisheries co-management

Marine fisheries are classified into three categories in Japan: (1) rights-based fisheries for coastal fisheries; (2) license-based fisheries for offshore and distant water fisheries; and (3) free fisheries. Although the expiration period is fixed in law, fishing rights are regarded as a real right, not a privilege, and provisions of the territorial rights law are applied mutatis mutandis. However, they do not include the right to privatize sections of the sea surface into portions. Fishing rights are rather similar to use rights in their attributions, i.e., the right to conduct fishery operations exclusively in specified areas by specified methods. By contrast, fishery licenses are privileges, but taking into account the large capital investments of the license holders, they are also strongly protected under law.

The fundamental concept of fishery management in Japan is “the holistic utilization of the sea surfaces” by the resource users themselves, as stated in Section 1 of the current Fishery Law of 1949, which remains in force (Makino and Matsuda 2005). Under this concept, a wide range of fishing operations within a given area is to be managed as a whole, not simply in terms of each individual economic unit. As a result, coordinating organizations at various levels have been created to facilitate a holistic fisheries coordination, including the Fishery Policy Council at the national level, Wide-Area Fisheries Coordinating Committees (WFCCs) at the multi-jurisdictional level, Area Fishery Coordinating Committees (AFCCs) at the prefectural level, and local Fisheries Cooperative Associations (FCAs) at the local level.

In addition to these formal coordinating organizations, a number of new operational ideas have been developed since the late 1970s, largely on the initiative of fishers themselves. These developments include what is known as Resource Management-type Fisheries, or “Shigen Kanri-gata Gyogyo.” More specifically, to maintain and improve their incomes, as well as to sustain resources, autonomous bodies of local fishers known as Fishery Management Organizations (FMOs) have initiated various management measures. FMOs are often formed by a group of fishers within an FCA. According to the biological nature of the target species, FMOs exploiting particular species are sometimes organized by members from several neighboring FCAs, or even by members of FCAs from several prefectures.

Within such a framework, the principal decision-makers with regard to management are the local fishers. The Fishery Law provides a framework for fishery management through a system of fishing rights and licenses. To achieve the holistic utilization of the ocean surfaces, these coordinating organizations have been granted wide-ranging authority and power. For example, the AFCCs, which consist mainly of local fishers, may determine the allocation of and restrict applications for fishing rights and licenses by means of their Fishery Ground Plan and Committee Directions. A variety of fishing restrictions has been stipulated by Prefectural Fishery Coordinating Regulations, FCA regulations, and FMO rules. Prefectural Fishery Coordinating Regulations broadly stipulate fishing restrictions, and these regulations apply throughout the prefecture. FCA regulations stipulate fishing restrictions in more detail, and these are only applicable locally. In particular, FCA regulations consider the restrictions set out in the Prefectural Fishery Coordinating Regulations and make additions to them. Similarly, the FMO rules constitute a further refinement of the FCA regulations.

The government also plays a vital role in fishery resource management. In fact, the co-management literature makes it clear that neither local fishers, nor the organizations to which they belong, can function efficiently without government cooperation or intervention (Pomeroy and Berks 1997). This is also the case for the Japanese institutional framework. For example, the Prefectural Fisheries Division is responsible for the issuance and renewal of fishing rights and licenses, and bases its decisions on advice from the AFCC. Scientific information or administrative guidelines presented by the prefecture often form the basis for the regulations and rules devised by local fishers. Furthermore, the “Resource Management Agreement System” described in the Marine Fisheries Resource Development Promotion Law of 1971 legislatively encouraged autonomous fisheries management among fishers. When a local agreement between fishers prevails at a certain level within an area, the government can affirm the agreement, and then it becomes an official rule. Therefore, it constitutes an official support system for autonomous resource management by the fishers. Also, in relation to the United Nations Convention on the Law of the Sea, the Law Regarding Preservation and Management of Living Marine Resources was enacted in 1996. Under this law, a total allowable catch (TAC) and a total allowable effort (TAE) system were introduced. Based on advice from the Fishery Policy Council or WFCCs, the central government sets the TAC and TAE, and controls total fishing pressures. However, the allocation of quotas and the determination of access rules are basically the responsibility of fishers’ organizations.

Hence, the Japanese fisheries management is co-management by fishers and the government, rather than compulsory, top-down regulation by the government, or a market-oriented management based on property rights and their efficient utilization by economically rational resource users.

Coastal fisheries management in the Shiretoko World Natural Heritage Area

Overview of the ecosystem and coastal fishery

The Shiretoko Peninsula and its adjacent marine areas, i.e., the Shiretoko World Natural Heritage (WNH) area, are considered to be the southernmost limit of seasonal ice floes in the Northern Hemisphere, and are affected by both the East Sakhalin cold current and the Soya warm current. In addition, the area is also influenced by intermediate cold water from the Sea of Okhotsk, creating a complex and rich marine ecosystem with both migrating and permanent species (Ministry of the Environment and Hokkaido Prefectural Government 2007).

In early spring, the Shiretoko ecosystem is characterized by rich low trophic level activity created by algal blooms following the melting sea ice. The area's high primary production supports a wide range of species, including marine mammals, seabirds, and commercially important species. In summer, sperm whales (*Physeter macrocephalus*) feed on squid in this area, and attract many tourists for whale-watching. In winter, numerous rafts of Steller sea lions (*Eumetopias jubatus*) are observed around the peninsula. Killer whales (*Orcinus orca*) also are present in the area throughout the year.

A distinguishing characteristic of this site is the interrelationship between its marine and terrestrial ecosystems. Large numbers of anadromous salmonids, such as chum salmon (*Oncorhynchus keta*), pink salmon (*O. gorbuscha*), masu salmon (*O. masou masou*), and Dolly Varden (*Salvelinus malma*), migrate up the rivers to spawn. They serve as an important source of food for terrestrial species such as brown bears (*Ursus arctos*), Blakiston's fish-owls (*Ketupa blakistoni blakistoni*), Steller's sea eagles (*Haliaeetus pelagicus*), and white-tailed eagles (*H. albicilla*). The brown bear is the largest land animal in Japan, and the top predator in the ecosystem. The peninsula is also internationally important as a stopover point for migratory birds (IUCN 2005). Steller's sea eagles and white-tailed eagles migrate from Russia to this area in winter, although some white-tailed eagles live permanently on the peninsula.

According to archaeological studies, people participating in fishing activities have lived in this area for thousands of years. Commercial fisheries in Shiretoko began in 1790 with the establishment of a fishery market by the rulers of mainland Japan, with the main products at that time being dried or salt-cured salmon, trout, and herring (Shari Fisheries History Editing Committee 1979). After the Meiji revolution of 1868, offshore fisheries targeting halibut and cod started. After the Second World War, a fisheries sector rapidly developed and the number of fishers in Shiretoko increased (Shiretoko Museum 2001). Today,

Figure 3. Autonomous marine protected areas introduced by walleye pollock gillnet fishers.

marine areas around the peninsula are among the most productive fisheries in Japan. In 2008, Shiretoko fishers caught 63,703 tons of fish, worth ¥23,525 million. Their main target species and gear types are salmonids using set nets, common squid (*Todarodes pacificus*) by jigging, and walleye pollock (*Theragra chalcogramma*), Pacific cod (*Gadus macrocephalus*), and Atka mackerel (*Pleurogrammus azonus*) by gillnetting. Fish processing industries are also very active on the peninsula, and the dried kelp (*Laminaria diabolica*) produced in this area is highly prized and fetches one of the best prices on the Japanese market.

Autonomous measures for resource management

Operations for walleye pollock are managed by licenses from the prefectural governor and fisheries coordinating regulations based on the Fisheries Law of 1949 and the Fisheries Resources Protection Law of 1951. This stock is also managed by the national government under the total allowable catch (TAC) system, based on the Law Concerning the Conservation and Management of Marine Life Resources of 1996. In addition to these official management measures, various autonomous management measures have been implemented. Local fishers compile data on catch size, time, location, body size, maturity, etc. These data are provided to the prefectural research station for analysis, with analyses returned to walleye pollock fishers, who then discuss management measures. For example, local fishers voluntarily enlarged the mesh size of their pollock gillnets from 91 to 95 mm in the 1990s, in accordance with research analyses provided by scientists.

Gillnet fishers divide the fishery ground into 34 areas based on their local knowledge and experience. Since 1995, they declared seven of these areas protected to conserve resources, including a portion of the walleye pollock spawning ground. The protected areas are re-examined every year on the basis of the previous year's performance and scientific advice from researchers. After nomination for World Heritage Listing, another six areas were designated as protected (Fig. 3).

Another example of an autonomous measure to conserve walleye pollock resources is the reduction in fishing capacity. The number of gillnet vessels in the late 1980s was 193 and to reduce the fishing capacity to better balance stock status, local fishers have decommissioned more than half of their vessels since 1996. Compensation for this decommissioning, about ¥1.1 billion, was jointly funded by the remaining fishers and the FCAs. The government paid interest costs for borrowing compensation money from the bank. In 2002, fishers introduced a joint operation system to reduce fishing pressure by 20% and further reduce operation costs: five boats form each group, with each boat sequentially suspending operations during the fishing season.

The recruitment, age structure, and catch per unit effort (CPUE) of walleye pollock has increased in recent years (Fisheries Agency and Fisheries Research Agency 2011), presumably as a result of these measures. Local fishers are thus engaged in both fishery operations and resource management, with core fishers attending about 15-20 meetings per month for their autonomous management.

New activities after inscription to the UNESCO Heritage List

New organizations for cross-sector coordination

Marine ecosystem conservation is by its nature a suite of activities across a wide range of related sectors such as fisheries, transport, tourism, etc. However, there is no domestic law specific to World Heritage programs, and conservation measures have been implemented by more than one authority based on separate laws. Table 1 shows the legal framework relating to the Shiretoko WNH area. As in many other countries, administrative procedures in Japan are structured in vertical silos. This often hinders horizontal cooperation and coordination across ministries and departments. For example, the Natural Park Law of 1957

Table 1. Major legal basis and administrative authorities for the Shiretoko World Natural Heritage area.

Public services	Legal basis	Administrative authority
Fisheries management	Fisheries Law of 1949 Fisheries Resource Protection Law of 1951 Law Concerning the Conservation and Management of Marine Life Resources of 1996	Fisheries Agency (Ministry of Agriculture, Forestry and Fisheries)
Pollution control	Law Relating to the Prevention of Marine and Air Pollution from Ships and Maritime Disasters of 1970 Waste Management and Public Cleansing Law of 1970 Water Pollution Control Law of 1970	Coast Guard (Ministry of Land, Infrastructure, Transport and Tourism) Ministry of the Environment
Landscape conservation and material circulation	Law on the Administration and Management of National Forests of 1951 Natural Parks Law of 1957 Nature Conservation Law of 1972	Ministry of the Environment Forestry Agency (Ministry of Agriculture, Forestry and Fisheries)
Species protection	Law for the Protection of Cultural Properties of 1950 Law for Conservation of Endangered Species of Wild Fauna and Flora of 1992 Wildlife Protection and Appropriate Hunting Law of 2002	Ministry of the Environment Ministry of Education, Culture, Sports, Science and Technology

does not allow the Ministry of the Environment sufficient authority to regulate adverse effects from fisheries activities on marine ecosystems. Because the fisheries sector has a long history as the foundation of the regional economy, coordination with the fisheries sector is especially important. The tourism sector, which is another important sector in the regional economy, experiences the same conditions. Therefore, to address this problem, a new system for cross-sector coordination was established for management of the Shiretoko WNH area (Fig. 4).

In October 2003, the Shiretoko WNH Site Regional Liaison Committee was established with officers from different ministries and departments in central and local government. They discuss what is proper management of the site, exchange information, and coordinate various interests among sectors. Local Fisheries Cooperative Associations (FCAs), the tourism sector, the Scientific Council (see below), and nongovernment

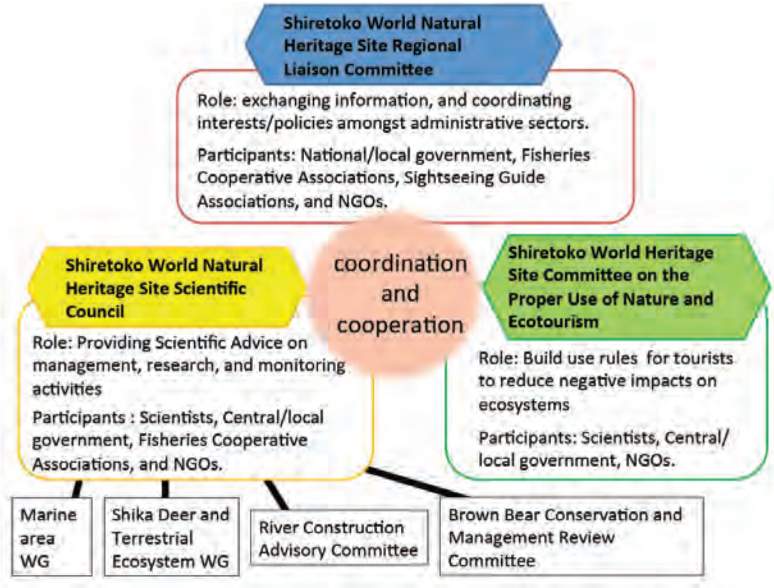

Figure 4. New organizations for cross-sector coordination in the Shiretoko World Natural Heritage area.

organizations (NGOs) are also members of this liaison committee, and the committee serves as the core venue for policy coordination among administrative bodies.

The Shiretoko WNH Site Scientific Council was established in July 2004. It provides scientific advice on the establishment of an Integrated Plan (explained in the next subsection) and on supporting research and monitoring activities. The council has two working groups: the Marine Working Group and the Yezo Deer Working Group for managing Yezo deer; and two committees: the River Construction Advisory Committee for river constructions such as mudslide control dams, and the Brown Bear Conservation and Management Review committee. The Scientific Council and working groups are composed of ecologists, social scientists, and representatives of ministries and departments in central and local government, FCAs, and NGOs.

Every year about two million tourists visit the area. The Shiretoko National Park Committee for the Review of Proper Use was founded in 2001. In April 2010, it was extended and renamed as the Shiretoko World Natural heritage Site Committee on the Proper Use of Nature and Ecotourism.

Through these organizations and their interrelationships, stakeholder participation is ensured, information and opinions are exchanged, and consensus among the wide-ranging interests of multiple users of ecosystem services is achieved, thus increasing the legitimacy of the Integrated Plans and related rules. This is the core institutional framework for integrated management under the "Shiretoko Approach."

The Integrated Plan

The Multiple Use Integrated Marine Management Plan (hereafter the Integrated Plan) was developed by the Marine Working Group of the Scientific Council, and approved by both the Ministry of the Environment of the Government of Japan and the Hokkaido Prefectural Government in December 2007. The Integrated Plan defines measures to conserve the marine ecosystem, strategies for maintaining major species, monitoring methods, and policies for marine recreational activities. The objective of the Integrated Plan is "to satisfy both conservation of the marine ecosystem and stable fisheries through the sustainable use of marine living resources in the marine area of the heritage site." The fisheries sector participated from the beginning of the drafting process.

Monitoring is the key component in an adaptive management. To monitor the Shiretoko marine ecosystem, the Marine Working Group outlined a food web (Fig. 5) and identified indicator species. The indicator species were salmonids (e.g., chum, pink, and masu salmon), walleye pollock, Pacific cod, Steller sea lions, seabirds including sea eagles, etc. They were selected from keystone species, predators at higher trophic levels that are likely to have a great impact on ecosystems, and endangered species in the waters surrounding Shiretoko.

Under the Shiretoko Approach, local fishers are identified as an integral part of the ecosystem and the data they provide are used to cost-effectively monitor the ecosystem. Local FCAs have been collecting and compiling catch data for more than 60 years. These data cover many of the indicator species and other major marine species in the food web. For some species, such as walleye pollock, detailed biological information such as size, maturity information, time and location of catch has been recorded on an autonomous basis by gillnet fishers. This information has established an important baseline for monitoring change against in the function and structure of the Shiretoko marine ecosystem.

Finally, since fishers are biased by their behavior, i.e., fishing activity takes place in an economic context, catch data alone are not enough to monitor the entire marine ecosystem and so additional data are needed. Therefore, the Integrated Plan specifies monitoring of noncommercial species, as well as basic environmental indices such as weather, water quality, sea ice, and plankton, by government agencies. Thus, the

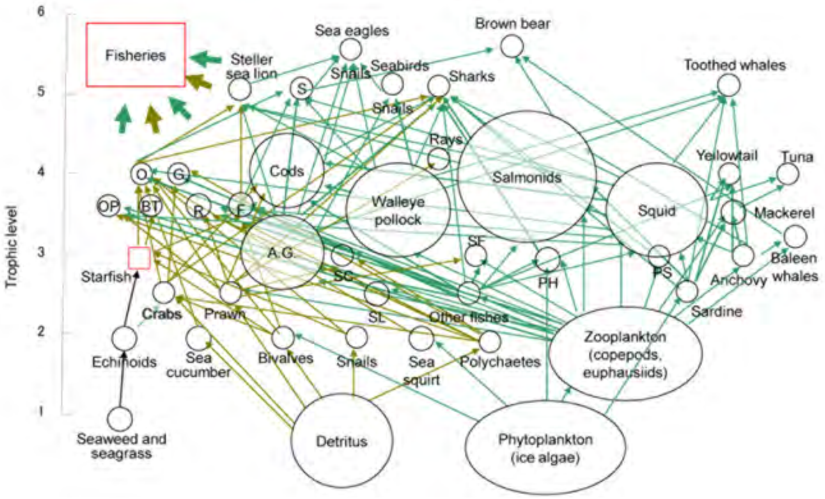

Figure 5. Coastal food web for the Shiretoko World Natural Heritage area.

Shiretoko Approach has initiated a system of role-sharing between government and ecosystem service users for sustainable and economically efficient ecosystem monitoring.

Steller sea lion

After the field evaluation in July 2004, the IUCN (International Union for Conservation of Nature and Natural Resources), an advisory body for UNESCO, expressed concerns about the conservation of Steller sea lions. The Okhotsk and Kuril population of Steller sea lions migrates from their breeding and landing grounds in Russian waters to the Shiretoko WNH area for overwintering and foraging. Because the Asian population of Steller sea lions sharply declined until the 1980s, this species has been classified as "endangered" and is on the IUCN Red List. Fortunately, its population has been gradually increasing at 1.2% per year since the early 1990s (Burkanov and Loughlin 2005). The entire population, which extends throughout the Sea of Okhotsk, the western part of the Bering Sea, and the Komandorskie Islands, was estimated at 15,676 in 2005 based on surveys from reproductive colonies. Hence, the Ministry of the Environment of Japan ranked the sea lion as "vulnerable category II" in 2002, which corresponds to "Vulnerable" in the current IUCN category.

Fishers in the Shiretoko WNH area consider the Steller sea lion as a destructive animal. Sea lions sometimes swim inside nets to eat the fish and then break the nets to escape. This damage is increasing, and estimated damage to fishing nets now costs more than ¥1 billion in Hokkaido (Fisheries Agency and Fisheries Research Agency 2010). Therefore, to mitigate the damage, 116 Steller sea lions are now culled each year under the Fisheries Law. However, since this cull level had no strong scientific foundation, in 2007 the Fisheries Agency of Japan revised the procedure for setting a cull level limit. It is now calculated based on potential biological removal (PBR) theory (Wade 1998), which is used under the U.S. Marine Mammal Protection Act. Using the number of sea lions migrating to Japanese waters and the life history parameters used for the eastern Aleutian population, the estimated PBR is now set at 227. Sea lions are also often observed as bycatch in bottom set nets, gillnets, and set-net fisheries in Hokkaido, but there are no official statistics on the number of sea lions killed by these fisheries. The estimated number of kills was between 55 and 107, and so the highest number has been subtracted from the PBR. Thus, the revised cull limit is now set at 120 individuals. If information on sea lion bycatch improves, the margin of error may be narrowed and the cull limit may be increased. Note that the culled sea lions are consumed locally as food, and so they too are considered to be an exploitable marine resource for Japanese.

Interconnection between land and sea

Many anadromous salmonids, including hatchery-derived chum and pink salmon, return to rivers in Shiretoko to spawn. Upstream they serve as an important source of food for terrestrial mammals and birds of prey, and contribute to biodiversity and nutrient circulation. Also, salmonids are the most important fisheries resource for set-net fishers operating in the Okhotsk Sea. Under the Fisheries Law and Fisheries Resource Protection Law, fishers with set-net fishery rights in marine areas are allowed to fish, while capture is prohibited in all rivers and near the mouths of certain rivers.

Man-made constructions such as dams could interfere with wild salmonids during their migration and prevent natural spawning. To maintain and facilitate interactions between marine and terrestrial ecosystems, man-made constructions along rivers in the Shiretoko WNH area have been modified since 2005. The River Construction Working Group under the Scientific Council (Fig. 4) has conducted a thorough survey on all the rivers in the area, and identified 118 man-made constructions in the Shiretoko WNH area. The working group then evaluated their impact on salmonids and investigated possible structural modifications, taking into account the probability of disaster risks on local residents. In some cases, modifications would have increased risk of disasters in densely populated areas, and so some constructions were not modified. The other structures have either been modified or were under modification at the end of May 2010. To assess the effects of these measures, a three-year program is in progress to monitor upstream salmonid runs, number and distribution of spawning beds, substrate composition, current velocities, and river discharge rates. Preliminary results have shown an increase in the number, size, and distribution of salmonid spawning beds.

Marine recreational activities

The Shiretoko Peninsula is a popular tourist destination in Japan, and tourism is an important contributor to the regional economy. Since its addition to the World Heritage List, the number of tourists visiting has increased considerably. Tourists use marine areas for sightseeing, sea kayaking, private boating, scuba diving, and recreational fishing, among other uses.

However, there has been growing concern that unregulated recreational use of marine areas may have adverse effects on the ecosystem. For example, passage by boats and unregulated feeding and watching at close range may affect the survival of seabirds and marine mammals. Many local fishers also complain of obstruction by tourists.

To prevent these negative impacts on the marine ecosystem and local fisheries, the Integrated Plan prescribes that recreational activities are to be managed under rules established by the Shiretoko National Park Committee for the Review of Proper Use (Fig. 4). This committee is composed of academics, tourism and guide representatives, environmental NGOs, and officers representing forestry, coast guard, environment, and local government. The committee prescribes patrols and activities to monitor tourist use, formulates rules for tourists, and promotes ecotourism. In addition, a new working group for sustainable tourism, the Working Group for Ecotourism, was formed under the Scientific Council in April 2010.

Costs for conservation measures

The lack of sufficient financial resources is one of the most challenging barriers to implementing ecosystem-based management. Table 2 shows the estimated administrative costs for the ecosystem conservation measures in the Shiretoko WNH area in 2006, based on information provided by the Ministry of the Environment, the Forestry Agency, and Hokkaido Prefecture. These costs are additional expenses that result from its addition to the Heritage List; they do not include conventional fisheries management costs. Also, the personnel are engaged mainly in Shiretoko WNH affairs: five full-time and four part-time workers at the Ministry of the Environment, one full-time at the Forestry Agency, and seven full-time for Hokkaido Prefecture. The average wage of government officers was used to calculate these personnel costs.

Table 2. Administrative costs for the Shiretoko World Natural Heritage area in 2006.

Cost item	Amount (¥ million)	Purpose
Running costs for Scientific Council and Working Groups	17.5	Giving scientific advice on the Integrated Plan
Running costs for the Committee for the Review of Proper Use and Shiretoko Ecotourism Association	15.1	Development of strategies for suitable tourism
Research and monitoring activities	54.7	Monitoring and research into adaptive management
River improvement	284.9	Modification of river constructions
Personnel	101.8	Administrative staff at the Ministry of the Environment and Hokkaido Prefecture
Total	473.5	

Source: Makino et al. (2009)

In 2006, fisheries production was ¥22,966 million, and tourists spent an estimated ¥36,617 million in the area. The total administrative cost thus corresponds to 0.8% of the amount returned by the two main industries that depend on services from the marine ecosystem. The total cost of effecting ecosystem-based management seems small in comparison.

Discussion and future challenges

The Shiretoko Approach is based on the Japanese fisheries management framework. At the beginning of the listing process, the IUCN mentioned that the Japanese fisheries management system was too complicated. The Japanese management framework might have been interpreted by the IUCN representatives as "setting the wolf to guard the sheep" (or from a Japanese proverb, "to set the cat to guard the dried bonito"). Therefore, as a member of the Marine Working Group of the Scientific Council, I made considerable efforts to explain the Japanese institutional framework and its relationship with ecosystem conservation in an internationally understandable manner. According to Copes and Charles (2004), the Japanese fisheries management system can be categorized as a type of "community-based co-management," which acknowledges local fishers as the primary participants in management and that the involvement and support of the broader communities is essential. It is open to considering a wide range of human needs in the community, and therefore lends itself to the implementation of a balanced mix of biological, social, and economic objectives. This fisheries institutional background in Japan naturally leads to a different ecosystem-based management framework than, for example, that of Iceland or New Zealand, where market-based individual transferable quotas are the central policy tool. There is no unique framework toward conserving marine ecosystems and sustaining livelihoods. What is required is a careful assessment of both the existing institutional framework and the potential role of ecosystem service users in marine ecosystem management.

At the UNESCO/IUCN Report on the Reactive Monitoring Mission held in February 2008, the mission team characterized the Shiretoko Approach as “an excellent model for the management of natural World Heritage Sites elsewhere” (UNESCO/IUCN 2008). We hope that the knowledge gained in setting up the Shiretoko WNH can contribute to future ecosystem-based management in other regions where large numbers of small-scale fishers use a wide range of species under a fisheries co-management regime.

In the Shiretoko WNH area, territorial disputes with Russia have encouraged participation by the fisheries sector. Russian trawlers are much bigger (700-4,000 gross tons) than Japanese gillnet vessels (10-19 gross tons) and they reportedly catch smaller individuals of walleye pollock (Fisheries Agency and Fisheries Research Agency 2007). As yet, there is no coordination between Japan and Russia to deal with this conflict. Shiretoko fishers hope that the World Heritage Listing will attract international attention to this situation and lead to some form of more effective management of walleye pollock in the near future. Because ecosystem boundaries do not necessarily reflect jurisdictional ones and are usually closely linked with adjacent areas, ecosystem management measures should be coordinated internationally where needed. Although there are serious territorial disputes over this boundary, dialog between scientific groups can be the first step to a resolution (Crosby 2007). Resolving this cross-boundary management (Ostrom et al. 2002) is important at an ecosystem level and thus for the Japanese government.

In 2008, a meeting between the Russian president and the Japanese prime minister was held in parallel to the 34th G8 Summit in Hokkaido to address the territorial dispute between the two countries. At this meeting, the Cooperative Program was signed with respect to Japan-Russia cooperation in the fields of conservation and sustainable use of neighboring areas to Japan and Russia, such as the Sea of Okhotsk. This program includes the conservation and rational use of the marine and onshore sections of neighboring areas, the use of information regarding the ecosystems, evaluations of marine environments, surveys of the effects of climate change on the ecosystems, and the expansion of exchanges between relevant institutions and experts in Japan and

Russia. Also, the Amur Okhotsk Consortium was established among Chinese, Japanese, and Russian researchers in 2009 (http://www.chikyu.ac.jp/AMOC/history.html) as a first step toward cooperation in environmental conservation of the Sea of Okhotsk and the Amur River Basin.

Another looming problem facing ecosystem conservation in the Shiretoko WNH area is climate change. The outstanding value of Shiretoko is strongly related to the presence of seasonal sea ice. This influences productivity of the marine ecosystem, which in turn influences productivity and diversity of the terrestrial ecosystem. The effects of long-term climate change could have a significant impact on these ecosystems. Local fishers, researchers, and residents agree that the amount and thickness of sea ice have been rapidly decreasing in recent years.

The Report on the Reactive Monitoring Mission by UNESCO and IUCN held in February 2008 pointed out the need to develop a climate change strategy that includes the following activities: (a) development of a monitoring program that identifies both long- and short-term impacts of climate change and specifically monitors parameters such as the extent of sea ice and the impacts on populations of key indicator species; and (b) adaptive management strategies that could be applied to minimize any impacts of climate change on the value of the site (UNESCO/IUCN 2008). The Scientific Committee of the Shiretoko WNH is now preparing a management strategy in response to this report (Makino and Sakurai 2012).

These new challenges cross sectors, and cannot be tackled effectively by isolated measures by individual ministries and agencies. In this regard, several initiatives to coordinate and integrate a wide range of measures have been established by the Japanese government. For example, it has developed a National Biodiversity Strategy of Japan, in accordance with the Convention on Biological Diversity, which provides targets and directions for measures to ensure conservation and the sustainable use of biological diversity. In the Third National Biological Diversity Strategy (Government of Japan 2008), the Shiretoko WNH was cited as a successful case of marine biodiversity conservation (Government of Japan 2008). In addition, the Ministry of the Environment is now formulating their first strategy specifically for the conservation of marine biodiversity. As for marine policy coordination, which includes resource conservation, biodiversity conservation, and international coordination, the Basic Plan of the Integrated Ocean Policy was formulated in March 2008. It is expected that these strategies and the basic plan will facilitate the formulation of integrated measures for the future challenges described above.

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Overexploitation of Fish Resources and Small-Scale Fisheries in the Northeastern Baltic Sea: Social Aspects of Ecosystem-Based Fisheries Management in the West Estonian Archipelago

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Abstract

The paper describes the outcome of an Estonian-Finnish INTERREG IIIA project (2006-2008) aiming to find the conceptual strategy for development of a coastal fishery in the West Estonian archipelago sea region in conditions when economically valuable local fish stocks have been seriously decreased in the two recent decades.

The opening of western European fish markets for Estonian fish trade in the 1990s, in conjunction with a relatively high price offered for economically key species like pikeperch (*Sander lucioperca* L.) and perch (*Perca fluviatilis* L.), resulted in an increase of fishing effort in the early 1990s. Increasing cormorant (*Phalacrocorax carbo*) and gray seal (*Halichoerus grypus*) populations have added additional pressure on the fish stocks in the area since the late 1980s.

The combined effects of the factors mentioned caused a drastic and persisting drop both in stock size and landings since the second half of the 1990s. As a result, the income and social well-being of coastal communities in the region suffered substantially, generating acute social problems such as high unemployment, decrease of coastal population, and impoverishing of structure and overall survival potential of coastal communities. The possible ways of mitigation of those problems, proposed as a project outcome, are discussed in the paper.

Introduction

The coastal fishery has historically been one of the most important ways of living in the western Estonian coastal region and the islands (Fig. 1). In addition to fishing, primary fish processing like salting and smoking took place in coastal communities up to the early twentieth century. Mainly gillnets, fyke nets, and row/sailing boats were used, and most inhabitants were involved in the fishery sector. Additionally, livestock breeding on the coast has been widespread. The combination of fisheries and livestock breeding in coastal areas lasted for centuries and produced unique semi-natural environments and architectural heritage (e.g., coastal and woodland meadows), which have became of high importance in EU Natura 2000 policy (http://www.natura.org/).

With the technological developments of the early twentieth century, fish processing and gear manufacturing began to concentrate into a number of specialized factories and the row and sailing boats were equipped with engines. As a result, the number of employees started to decrease in the fishery sector in the 1930s. However, approximately 10,000-12,000 coastal inhabitants were still fully or partly involved, mainly as inshore fishermen, before World War II (Anon. 1987, Pettai 1991).

In the 1950-1960s, the coastal fishery was changed from small cooperatives to large fishery collective farms, which operated in other fields of the fisheries sector besides fishing such as fish processing and boat and shipbuilding. The collective farms were mostly concentrated in the larger settlements, while the most of the coastline and the historical small harbors were closed for civil use. This led to the deterioration of the historical lifestyle of the coastal communities. However, the role of coastal fishing integrated with agricultural activities remained an important part of life in western Estonian coastal villages and the historically settled coastal landscapes retained their historical shape.

After regaining independence in 1991, the restrictions on going out to sea were abolished in Estonia and the number of fishers, both professional and nonprofessional, began to grow and the fishing effort increased. At the same time, the control and enforcement over coastal fisheries was clearly not sufficient and the actual catches of the early 1990s remain virtually unknown (Anon. 2005, Kangur 2006, Vetemaa et al. 2006).

In 1995 the new Estonian Fishery Act was adopted. It granted fishing rights to all coastal inhabitants, considerably raising the number of people having rights to fish with commercial gears (Anon. 1995). As a matter of fact, the implementation of a minimum landing size (L_{min} = 19 cm) in 1997 was the first regulatory measure since the beginning of the 1990s, reducing the fishing pressure on perch stock in the area. In 1999, the minimum gillnet mesh size A_{min} = 92 mm was implemented

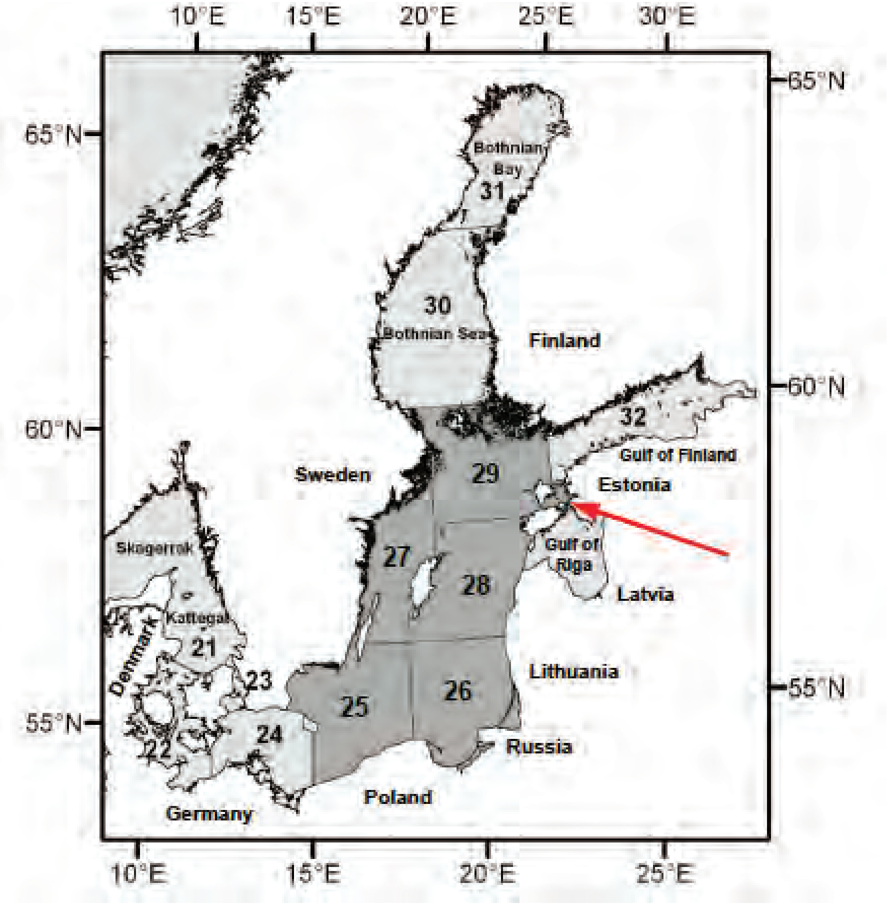

Figure 1. **Location of the West Estonian archipelago in the Baltic Sea.**

for this region and the total ban on fishing (except herring pound nets and eel traps) was established for the period from 01 April to 15 May (Järv et al. 2005, Järv and Järvik 2009). However, the annual landings of perch decreased until 2005-2006, and stabilized at a low level afterward (Järv and Järvik 2009, Saat 2008). The results of special fish monitoring within the Western Estonian archipelago indicated the absence of strong or even medium year-classes of perch in 1995-1998. Only medium size year-classes of perch appeared in 1999, while only the poor year-classes of perch emerged in 2000-2004 (Saat 2008). Additionally, changes in natural conditions such as eutrophication of the coastal zone in the 1970-1980s, plus a rapid increase in abundance of cormorants and seals since the 1990s, have had negative impacts on fish stocks in the Estonian coastal sea (Anon. 2007, Eschbaum et al. 2003, Vetemaa et al. 2006). The combination of unfavorable environmental conditions and overexploitation in the early 1990s led to a significant drop in catches in the commercial coastal fishery and, as result, the income from fisheries for coastal fishermen suffered substantially. The threat to the survival of coastal professional fishing arose particularly for the West Estonian archipelago sea region, which was formerly one of the most important areas of coastal fishing in Estonia with annual catches above 5,000 t (Fig. 2; Saat 2008).

The problem became even more acute after the accession of Estonia to the European Union in 2004, and the adoption of principles of the Common Fisheries Policy, when the feasibility of possible financial support to the coastal fishery in the West Estonian archipelago area needed to be evaluated.

In order to explore which measures could be taken to reverse the negative trends, the EU INTERREG IIIA Finnish-Estonian project was launched for 2006-2007. Three counties—Saaremaa, Hiiumaa, and Läänemaa, with 5, 5, and 8 local municipalities respectively—were covered by the project (Järvik et al. 2008).

The main goals of the project were to:

- Propose a plan for development of a professional fishery.
- Evaluate the perspectives of fish processing and trading.
- Elucidate the needs of professional education for fishers.
- Assess the current state and future prospects of fishing tourism.
- Evaluate the possibilities for sustainable management of local fish stocks.
- Propose ways to integrate development of the coastal zone.

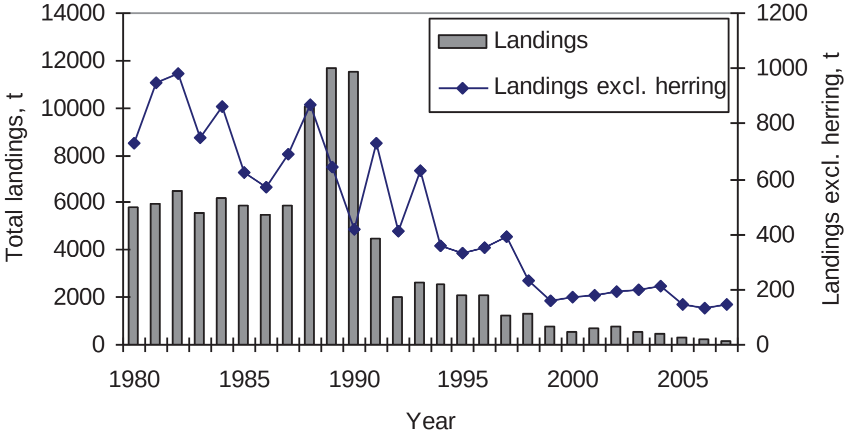

Figure 2. Annual landings (t) from the West Estonian archipelago sea area in 1980-2007. Bars represent total landings and line landings other than herring.

The project targeted the following groups:

- Professional fishers.
- Fish processing enterprises.
- Aquaculture enterprises.
- Municipalities and county authorities.
- Tourist enterprises.
- Nature conservation institutions, including the Natura 2000 managers.

Material and methods

The general framework of the project is described in Fig. 3. The project is based methodologically on a bottom-up strategy aiming for as broad as possible representation from local communities. The data were collected from relevant government statistical sources and by interviewing professional coastal fishermen.

In order to obtain the qualified data on the peculiarities of coastal professional fisherman communities, quantitative analysis was needed (Aimre 2006). A written questionnaire of 20 questions was used in order

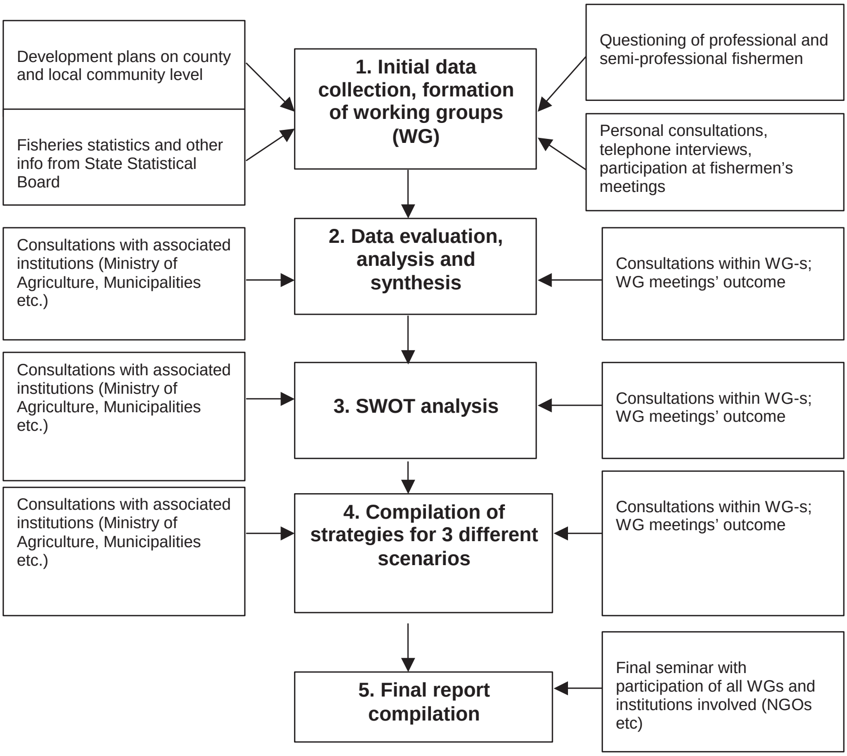

Figure 3. The general framework of the project.

to be less intrusive compared to telephone or face-to-face interviewing. Additionally, when using the method of a mailed questionnaire, the respondents are free to complete responses according to their own time schedule (Walonick 1993). The interviewing was a combination of quantitative and qualitative studies. Altogether, 180 (of 370 total) randomly chosen local professional fishermen were interviewed in 2007 winter by mail. The interviews resulted in 72 responses (40%). The interviewees remained anonymous.

To achieve the main goals of the project, ad hoc working groups were established, consisting of representatives of local communities and project managers. The working groups carried out 1-3 workshops (11 in total) during the project period. Additionally, the project managers had a number of meetings with professional fishermen in all three counties of the West Estonian archipelago to introduce the main results in interviews. The outcome of interviews was then used in S.W.O.T.

Table 1. Short overview of the main aspects of S.W.O.T. analysis: strengths and weaknesses.

Strengths	**Weaknesses**
West Estonian archipelago sea area (>2,200 km2) with shallow brackish water and with rich bottom communities offers excellent conditions for development of diverse and abundant fish fauna.	The stocks of main commercial fishes (perch, pikeperch, pike, whitefish) have been in poor condition since the mid 1990s because of overfishing and a drastic increase in the abundance of predators—cormorants and seals.
The quality of the coastal sea environment is relatively good due to absence of major pollution sources in West Estonian archipelago sea area.	Some spawning areas of freshwater and brackish water species are not recovered yet from eutrophication caused by intensive agriculture in the 1960-1980s.
Historically developed unique semi-natural environments and architectural heritage have been preserved in the area. The area is protected as an EU Natura 2000 site.	The coastal fishery often has a seasonal character and can't provide year-round income for fishermen.
Long traditions of fishing and professional knowledge of fishermen have been historically propagated from one generation to another.	The average age of coastal fishermen has increased recently.
Implementation of an adequate system of licensing of coastal fishermen.	Most fishing boats originate from the Soviet era, equipped with wasteful and high environmental risk engines.
Centuries-long history has resulted in the optimal localization of small fishing harbors and landing sites.	The selectivity of traps and fyke nets are not at an adequate level.
A number of commercial fish species in the region have good markets, both in Estonia and abroad.	There are almost no active fish processing facilities left in the region.
A traditional system of professional education at the vocational, sub professional, and college levels exists.	Landing sites are not equipped with ice machines or freezing and lifter mechanisms, which causes a high risk that landed fish may become spoiled.
There is a well developed system of fisheries investigations.	Due to the limited demand of local markets and the absence of possibilities for preservation of fish on the spot, problems with selling landed fish often arise, particularly on islands.
The availability of financing from EU Fisheries Fund (EFF) in 2007-2013 of up to 85.4 million Euros.	The prices offered to fishermen by first-buyers are often extremely low.
Since 2007 Estonia has its own fisheries strategy, formulating inter alia the principles of the development of coastal fisheries.	Coastal fishermen often face difficulties in getting bank loans, and private investors are reluctant to invest in fishing harbors.

Table 2. Short overview of the main aspects of S.W.O.T. analysis: opportunities and threats.

Opportunities	Threats
Regulation of fishing effort in order to achieve a balance between effort and available resource and thus increasing the economic payback of the coastal fishery.	Deterioration of the ecological situation in West Estonian archipelago sea area, including the threat of potential navigation disasters, pollution by river inflow, etc.
Financial and political support for renovation of a number of key local fishing ports.	The continuous lack of adequate investments in the regional fishery sector jeopardizes recruitment of younger generations into the coastal fishery.
Establishment of a system supporting small-scale fish processing using EFF co-financing.	Further strengthening of health security demands on fish products, under conditions lacking adequate financial support would seriously damage the development of the fish processing industry in the region.
Modernization of fishing boats.	Further increase in abundance of cormorants and seals without adequate mitigation measures can be envisioned in the region.
Establishment of a system for supporting recruitment to the fishery sector.	The absence of small fishing ports owned by municipalities and/or nonprofit organizations does not allow applying for available funding from EFF.
Establishment of the state system of training coastal fishermen and small-scale fish processing.	Drastic increase in prices of oil products.
Implementation of fishing methods and gears of higher selectivity.	Substantial decrease of fishing possibilities due to implementation of new extraordinary environmental protection measures in the region.
Implementation of seal-proof fishing gears.	Possible ongoing pressure of first-buyers to keep prices low
Establishment of the state program of regulating the abundance of cormorants.	Worsening of the employment situation as a result of possible general economical slowdown, would force inhabitants to leave the area, causing degradation of local historical heritages and also several Natura 2000 objects and habitats.
State/municipal regulation of the ownership of fishing ports; some small fishing ports in the region should be operated as non-private, which would allow applying for funding from EFF.	
Restoration of fish spawning grounds and the possibility of fish stocking (whitefish, pikeperch, perch).	
Support at state and municipality level of maintenance and development of historical-architectural heritages in coastal villages, including EU Natura 2000 objects and habitats.	

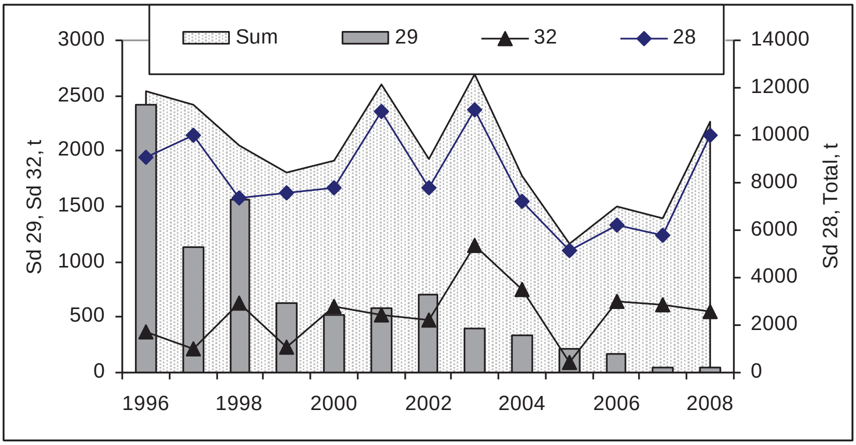

Figure 4. Herring landings (t) in the Estonian coastal fishery, by ICES subdivisions in 1996-2008.

(Strengths, Weaknesses, Opportunities and Threats) analysis to address the main goals of the project (Järvik et al. 2008). A short overview of the S.W.O.T. analysis can be found in the Tables 1 and 2).

Results and discussion

The poor state of many local fresh- and brackish-water fish stocks has been a fundamental problem in the coastal fishery of the West Estonian archipelago as reflected in the dramatic drop in commercial catches since the 1990s, when the total catch of fish other than herring decreased 3- to 4-fold (Fig. 2). The fast decrease can be observed also in herring landings (Fig. 4). In this case, however, the substantial decrease in fishing effort in the coastal herring fishery was at least one of the factors causing landings to decrease. Approximately 200 herring pound nets were deployed in the area contributing to 90-95% of annual herring landings in 1970-1980s (Figs. 2 and 4). In the 2000s, the total number of deployed pound nets has become negligible, particularly because of damage to the gears and catch by seals (Järvik and Raid 2008). The interviewed fishers, who were asked to list the possible causes for the decrease in coastal fisheries, pointed at the poor state of fish stocks due to bad environmental conditions (60-81%), inadequate administration of the fishery (31-41%), low fish price at the local market (22-33%) as main reasons behind the situation. The possible overexploitation of fish resources got higher ranking in one county only (Saaremaa, 33%,

Table 3. Responses of fishers (percent) to the question "What has caused the poor situation in coastal fishery?" The interviewees were asked to list the possible causes.

Possible cause	Saaremaa County	Hiiumaa County	Läänemaa County
Bad environment	60	81	62
Overexploitation	33	22	16
Administrative mistakes	37	31	41
Accession to the EU	7	18	0
Low fish price	33	22	33
There is no problem and stocks will recover	7	4	4

Table 3). As to the effect of environmental conditions on stocks, the fishers mostly referred to the excess abundance of cormorants. The results indicate that fishers clearly acknowledge the existence of the problems and remain pessimistic with respect to future developments in the coastal fishery. However, the fishers were quite reluctant to point at overexploitation of fish resources in the area as the main reason for the current situation. This is in stark contrast to the general opinion that the rapid increase in the number of fishers after coastal habitants were granted fishing licenses in the early 1990s, and virtually uncontrolled fishing, are the main reasons for the drastic decline in fish stocks and fisheries (e.g., Saat 2008). The opening of Western markets for perch and pikeperch filets in the generally poor economic situation in the early 1990s presented a strong incentive to start fishing businesses and the number of fishers soared (Järvik et al. 2008). Unfortunately, at the same time several existing fishing restrictions were eased, particularly in Matsalu Bay, which was known as the main spawning area for several fish species in the West Estonian archipelago (Saat 2008). As a result the resources of valuable species of that bay (whitefish, pike, eel, pikeperch, several cyprinids, and particularly perch) collapsed (Vetemaa et al. 2002, 2010; Saat 2008). Even at present when commercial catches have fallen to close to historic lows, and the control and enforcement has clearly improved, the excess fishing effort (in the gillnet fishery, in particular) still seems to hamper the recovery of a number of fish stocks in the area.

Other factors having a negative effect on fisheries can be related to ecosystem changes. For example, the number of nesting cormorants

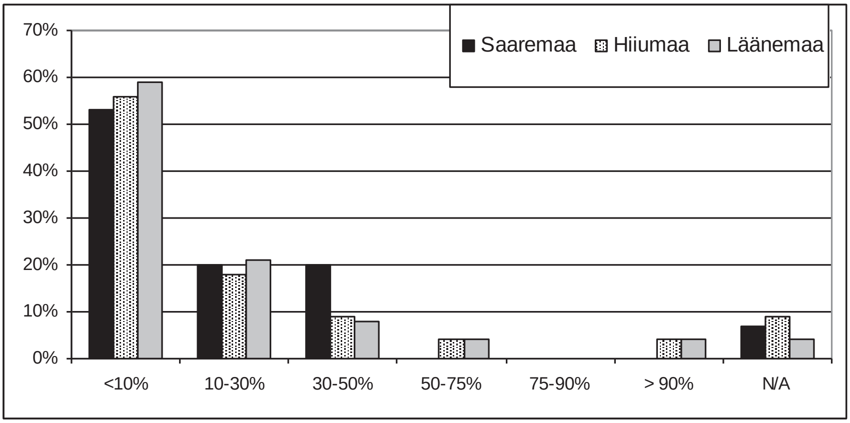

Figure 5. Share of fishing in the total income of interviewees by counties. N/A = did not respond.

has dramatically increased, although they were virtually missing in the area up to the end of the 1980s. Approximately 1,000 nesting pairs were counted in the area in 1990s, and 40,000-50,000 pairs have been encountered in the 2000s (Eschbaum et al. 2003, Vetemaa et al. 2010). Due to the shallow water, mostly less than 10 m, cormorants are able to reach all water depths in the area. According to a pilot study performed in the early 2000s, the cormorants catch more fish annually in this area than professional and recreational fishery and poaching do together (Eschbaum and Veber 2002). Additionally, the numbers of another predator, the gray seal, have increased significantly in past decades, from 1,500 in 1999 to more than 3,000 by the end of 2000s (I. Jüssi, Estonian Fund for Nature, pers. comm.).

The observed changes in the ecosystem in conjunction with excess fishing effort have contributed to total mortality and decrease of fish stocks in the area. The collapse of fish stocks has its obvious social effect: the decrease of fish catches in the West Estonian archipelago by more than a factor of 10 during a short period had a substantial effect on incomes of coastal fishers. The results of the interviews showed that only 10% of coastal fishermen fishing in the area got their main income from fishing, whereas over 50% of responding fishers said that their income from the fishery was less than 10% (Järvik et al. 2008; Fig. 5).

Out of approximately 370 fishermen who have rights to professional fishing in region, more than 185 can be regarded as semiprofessional or household fishermen, for whom the use of large professional fish-

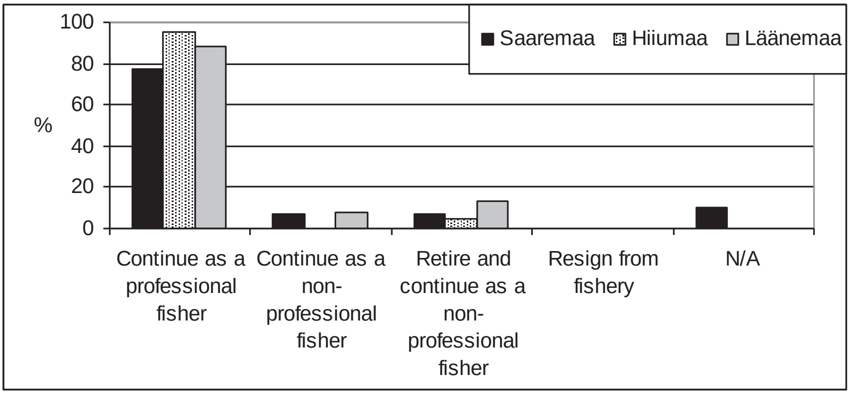

Figure 6. Future plans of interviewed coastal fishermen by counties. N/A = did not respond.

ing gears, as herring pound nets and large fyke nets, have became too expensive, particularly in the conditions of massive seal damage (Järvik et al. 2008).

In spite of the generally poor situation in the fishery, surprisingly high percentages were still planning to carry on as professional (or semiprofessional) fishermen: 77% of the interviewed fishers from Saaremaa, 94% from Hiiumaa, and 88% from Läänemaa counties (Fig. 6). The main reason for the majority is that professional fishing has been a traditional activity for centuries in this region; it was an essential part of the way of life for their ancestors and therefore should be the same for future generations (Järvik et al. 2008).

At the same time, they admit that other income sources are needed in order to subsist in the future. Depending on the county, 25-48% of fishers' responses said fishing tourism is one of the main sources for potential additional income, and 15-20% also responded that small-scale fish processing as a family business might be profitable as well. Traditional agricultural activities are seen as an income sources by 10-30% of fishers (Järvik et al. 2008; Fig. 7).

Consequently, the developing of fishing (and fish-farm) tourism and small-scale fish processing, accompanied by the renovation of fishing ports, may result in sufficient increases of the incomes of local permanent habitants in the fishery sector that could slow down or even reverse the current migration of people from coastal villages to the mainland and cities.

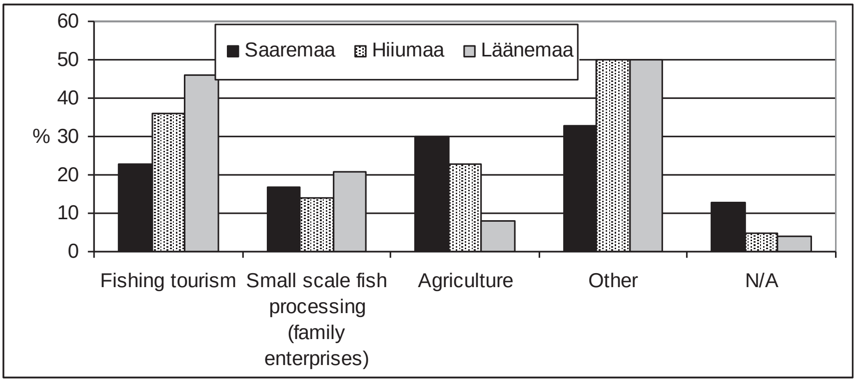

Figure 7. Possible sources of supplementary income for coastal inhabitants as proposed by interviewees. N/A = did not respond.

The potential increase of landings, the creation of fishing tourism, and restoring profitability through a small-scale fish processing industry, however, can be achieved only when the local fish stocks that are economically important and potentially attractive for game fishing are in fair condition. In that context, pike, pikeperch, perch, and whitefish stocks of the West Estonian archipelago should deserve particular attention. Given the present poor state of fish stocks, however, positive effects cannot be expected without additional effort on local and state levels. The following measures are outlined as fist steps towards initiating the process:

- Map spawning grounds of valuable fish species in order to evaluate their reproduction potential and present reproduction success;
- Restock valuable fish species in the area (pike, whitefish)
- Mitigate cormorant and seal effects on fish stocks and fishery (e.g., “oil” cormorant eggs in main colonies and introduce seal-proof gears);
- Improve selectivity of coastal fishing; and
- Increase the role of local municipalities in fishing regulation.

Strengthening enforcement and regulating the abundance of cormorants might be the first steps allowing the fish stocks recover. However, landings would probably never reach the level of pre-1990s due to altered marketing conditions (particularly that of herring) and due to ecosystem changes (Järvik et al. 2008, Saat 2008).

In order to effectively carry out the proposed steps the need for special financial support to coastal fishery in the region from the European Fisheries Fund was especially pinpointed in the outcomes of the project. According to Operational Program of the European Fisheries Fund for 2007-2013, the mentioned financing started in 2008 including implementation of more selective and seal-secure trap nets (Anon. 2008). Additionally, the renovation of around 10 small fishing harbors within the area is planned to start in 2011 (Anon. 2009).

Conclusions

The following main conclusions can be made from the project:

- The recently introduced fishing restriction measures (including closed periods and areas, gear restrictions, etc.) have not resulted in significant recovery of overexploited fish stocks (pikeperch, perch, pike), possibly due to the combined effect of still high fishing effort and unfavorable ecosystem changes (high number of cormorants and seals).

- The drastic decrease in fish landings in the West Estonian archipelago has seriously affected the income of professional fishermen, which has forced increasing numbers of inhabitants of coastal communities to leave the area. While fishers believe the fishery will remain one of the most important income sources for coastal village inhabitants in the West Estonian archipelago, it has become clear that fishing itself without supplementary sources of income for local habitants would not be enough to sustain development of coastal communities.

- Fishing tourism and small-scale fish processing in local fishing harbors should be considered as prospective additional income sources for coastal inhabitants. Further, other traditional activities such as agriculture and forestry should be continued to maintain the existence of viable coastal communities, as a prerequisite for safeguarding the Natura 2000 sites within the region.

- The integrated coastal area management strategy can be successful only when fully implemented and endorsed (both politically and financially) on state, county, and municipality levels.

- External financial support (e.g., from the European Fisheries Fund) would be vital in order to maintain sustainable development of the coastal fishery in the West Estonian archipelago area.

- The output of the project served as the basis for development of Fisheries Strategies for all three surrounding the West Estonian archipelago counties within the framework of the European Fisheries Fund application plan 2007-2013.

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A Bio-Economic Model for the Lobster (*Homarus americanus*) Fishery in Canada

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Abstract

A bio-economic model was developed for a lobster (*Homarus americanus*) fishery in Canada to assess the biological and economic benefits associated with an increase of the minimum legal size (MLS) from 70 to 72 mm carapace length, either over two years (scenario 1) or in one year (scenario 2), and an effort reduction from 250 to 200 traps per harvester. The model has three components: a lobster population dynamics module, a socioeconomic module, and a link module. For scenario 1, landings would increase 1.7% versus the status quo, with a 2.4% increase in revenues, and a 10.8% increase in the egg production. Scenario 2 was better with a 2.7% increase in landings, a 3.9% increase in revenues, and an 18.0% increase in egg production versus the status quo. The 20% trap reduction, without affecting landings as the removed traps were considered empty of commercial-size lobsters, will generate an increase of net after-tax revenues for fishing enterprises of 35.4%, with a 5% reduction in employment, 11.6% and 19.0% increases in gross domestic product and tax revenues, and finally a reduction of 10% for both the energy consumed and greenhouse gas emission. Hence, increasing MLS and reducing the number of traps had both biological and economic benefits, demonstrating that an integrated approach for a successful ecosystem-based fisheries management of the lobster fishery should include economic objectives in addition to the parameters traditionally included in fishery assessment models.

Introduction

The lobster (*Homarus americanus*) fishery in the southern Gulf of St. Lawrence (sGSL) has developed over more than a century as a nearshore small-boat fishery, involving a large number of harvesters using only lobster traps as fishing gear (DeWolf 1974). The lobster fishery is presently the most important resource in Eastern Canada with total landings of 57,048 t in 2008 valued at over $600 million CDN (DFO 2010a). This fishery employs thousands of harvesters, plant workers, and employees of various suppliers in small coastal communities.

The Department of Fisheries and Oceans (DFO) Canada manages the sGSL lobster fishery based entirely on effort control (i.e., no quota) and measures to protect key components of the lobster population (Comeau et al. 2008). Effort is controlled by a fixed number of fishing licenses, an individual trap allocation, restrictions on gear characteristics, and a 2-month fishing season. The only summer fishing season in the sGSL, which was the focus of this paper, operates from 10 August to 10 October with up to 80% of landings observed in the first four weeks (Comeau et al. 2004, 2008). In addition to effort controls, only harvested animals with a carapace length (CL) above the minimal legal size (MLS) of 70 mm, and females not carrying eggs and smaller than 115 mm CL, can be retained.

In sGSL, numerous changes in MLS were observed since its establishment at 63.5 mm CL in 1952 (DeWolf 1974, Comeau et al. 2008). Between 1990 and 2004, the MLS in the summer fishery was raised from 63.5 to 70 mm CL mostly to increase egg production (Lanteigne at al. 1998, 2004; Comeau et al. 2004, 2008), as recommended by the Fisheries Resource Conservation Council (FRCC 1995). The FRCC is a nongovernmental committee mandated, for the first time in 1994 by the federal minister responsible for DFO, to review approaches to conservation and recommend strategies for sustainable exploitation of Canadian lobster stocks. Based on estimates from an egg-per-recruit model, the FRCC first concluded that the lobster fisheries were primarily harvesting immature animals, and not allowing for adequate egg production (FRCC 1995). In their recommendations, they suggested that MLS could be increased to increase egg production. In their second report, the FRCC further recommended that the MLS should be set at the size at onset of 50% sexual maturity (SOM_{50}) allowing for more primiparous females (i.e., first time spawners) to mature before becoming available to the fishery (FRCC 2007). Thus, the MLS should be further increased to the SOM_{50}, which is 72 mm CL for the summer fishery (Comeau and Savoie 2002; Comeau et al. 2004, 2008).

In both reports, the FRCC (1995, 2007) also concluded that the exploitation rate was too high and recommended to reduce fishing effort by reducing the number of licenses, traps per licenses, trap hauls, or length of the fishing season (FRCC 1995). Although measures aimed at increasing egg production were implemented in two multiyear conservation plans after extensive consultations with the fishing industry (Comeau et al. 2008), no measure was put in place to reduce effort. Fisheries managers were unable to convince the industry to reduce the number of traps per harvester, set at 250 since 1966 (DeWolf 1974), although it was shown that over 50% of traps are empty (i.e., no legal-size lobster) in the first four weeks of the summer fishery (Comeau et al. 2004, 2008).

DFO recently announced the Atlantic Lobster Sustainability Measures (ALSM) contribution program, created to provide financial aid for the Canadian lobster industry, mainly harvesters (DFO 2010b). The establishment of the ALSM was triggered by substantial price declines in 2008 and 2009, in comparison with 2007, caused by the global economic and financial crisis. The objective was to provide support for the development and implementation of lobster sustainability plans, to help the fishery make changes that will enhance its economic prosperity through self-rationalization, i.e., effort reduction, and long-term sustainability. To access funding, harvester associations must first submit a sustainability plan that improves biological productivity (MLS reaching SOM_{50} or equivalent), provides reliable reporting of landings, and reduces ecosystem impacts of lobster fishing activities in terms of ghost fishing, entanglements of nondirected species, or recording bycatches. Harvester associations may then submit a request to DFO for funding for specific projects, which mainly consist of effort reduction through buyback or reduction of the number of traps.

The purpose of this paper is to describe a bio-economic model for the lobster industry that includes a population dynamics module linked to a socioeconomic module to provide biological (i.e., landings and egg production) and economic (i.e., expenditures and net benefits) outputs associated with the increase of the MLS from 70 to 72 mm CL, and a reduction in the number of traps per harvester from 250 to 200 for the summer fishery in the sGSL. Our case study investigated economic objectives, in addition to the parameters traditionally included in fishery assessment models, which could be used to guide the decision-making process for fishery managers attempting to implement an ecosystem-based fisheries management strategy for lobster fisheries as opposed to the traditional single species approach.

Methods and models structure

The biological module is a lobster population dynamics (LPD) module (Fig. 1) providing outputs for the number of lobsters landed, their size and weight, by sex and by time period (20 time periods per year), and egg production that are similar to those from a yield- and egg-

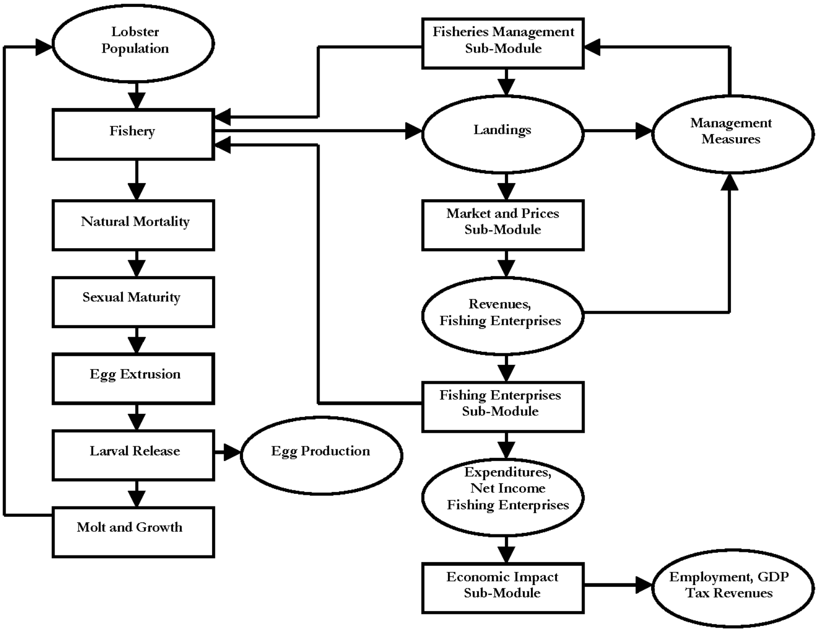

Figure 1. Flow chart of the lobster (*Homarus americanus*) bio-economic model composed of the population dynamics and the socioeconomic module. Rectangles and ovals represent processes and outputs, respectively. Landings are in number of lobsters converted to weight.

per-recruit model (see Fogarty and Idoine 1988 for models adapted to lobster). There are six main events in the LPD module, five of which are biological and one human intervention (i.e., the fishery). The sequence of these events (Fig. 1) is important to properly estimate the number of lobsters that survive and are able to reproduce from year to year.

Linkages between the LPD and socioeconomic modules are bidirectional (Fig. 1). From the LPD module output (i.e., landings in number of lobsters converted to weight), the market and prices sub-module of the socioeconomic module calculates total landed weight and revenues for lobster fishing enterprises, and generates a vector of revenues on a year-by-year basis. This vector of revenues is then used as an input by the fishing enterprises sub-module to calculate the net income of harvesters. Both landings and the revenues of fishing enterprises will have an impact on management measures determined by the fisheries management sub-module. The economic impact sub-module takes the revenues on a year-by-year basis and calculates both economic benefits and cost-benefit statistics (for 10 and 20 year periods).

Biological parameters for the lobster population dynamics module

Simulations on males and females were done separately due to differences in their growth rates and fishery regulations (i.e., the landing of berried females is prohibited). The model was divided into 20 time periods. The first time period from January to the end of June was followed by 18 periods representing one week each, from the beginning of July to mid-November, and the last time period ran from mid-November through the end of December. The model used input variables to model growth, natural and fishing mortalities, and egg production. Three management measures were included in the model. The first is that berried females must be released if caught. The second and third measures are the release of animals smaller than the MLS, currently at 70 mm CL, and the release of females larger than the maximum size set at 114 mm CL. All data related to lobsters were contained in matrices with 300 lines (one line per millimeter in size) and 500 columns (20 time periods per years × 25 years).

Fishing mortality

Estimates of the annual exploitation rate (percent caught) based on catch curve and the Leslie Analysis methods vary from 57% to 84% in the sGSL (Lanteigne et al. 1998). More recently, Comeau and Mallet (2001) estimated an exploitation rate of 69% with small confidence intervals based on the catch-effort model. Based on that information, the mean annual exploitation rate was set at 70% for lobsters from the MLS to 99 mm CL. For larger animals, the exploitation rate was decreased based on trawl survey observations (Comeau et al. 2008) to 20% for animals between 100 and 113 mm CL; 15% between 114 and 130 mm CL; 10% between 131 and 142 mm CL; 5% between 143 and 159 mm CL; and zero for lobsters 160 mm CL and above (animals rarely seen in the catch; Mallet et al. 2006).

Natural mortality

There is no accurate estimate for the lobster natural mortality (M). The only fishery-based estimates of M from the sGSL show that it varies from 0.0057 to 0.02 based on tagging studies and between 0.002 and 0.01 based on simulations, with wide confidence intervals in both cases (Comeau and Mallet 2001). Comeau and Mallet (2001) indicated that the inability to accurately estimate M from existing methods is probably due to an actually low value of M; otherwise its estimate would be possible notwithstanding the accuracy of the methods (i.e., the accuracy of models overshadow the estimate of *M*). Indeed, this is supported by predator-prey relationship projects carried out in the sGSL (Hanson and Lanteigne 2000, Hanson 2009). Nevertheless, with wide variations in the *M* estimates conservative values of 10% and 5% were used for molting and non-molting animals, respectively. A higher *M* for recently molting animals is used because they should be more vulnerable to predation due to their very soft shell condition.

Egg production

The number of eggs at size was based on Campbell and Robinson (1983) fecundity curve:

$$y = 0.00256x^{3.409}$$

where y represents egg production and x is CL (mm).

The SOM_{50} of female lobsters in the sGSL is typically reached at 72 mm CL (Comeau and Savoie 2002, Comeau 2003). The proportion of mature females (*P*) within each size class of 1 mm CL was modeled by the logistic equation:

$$P = \frac{1}{1 + \exp^{[-(-20.2231 + 0.2802\,CL)]}}$$

About 97% of all females reach sexual maturity over three years, or three molt groups (Comeau and Savoie 2002). Three cohorts of mature females were created in the module based on the year sexual maturity was reached where 14% reach maturity in year 6, 58% in year 7, and 28% in year 8 to match the maturity schedule with the growth schedule.

The schedule of larval release starts during the second week of July at 16.7% followed by a 20% release in each of the following weeks. By the third week of August, 100% of females have released their larvae, which was corroborated by direct observation using scuba (Michel Comeau, pers. comm.) and at-sea sampling (Mallet et al. 2006).

Growth

Based on Comeau and Savoie (2001), three different growth rates were used in the model. A growth increment of 16.8% per molt was used for males, while 15.2% and 12.0% were used for immature and mature females, respectively. The molt frequency for males between 50 and 120 mm CL and immature females in the sGSL is annual with the main molting period from early July to early September (Comeau and Savoie 2001). The proportion of animals available to the fishery for each time period used in the model is presented in Table 1. The molt frequency for mature females, with a two-year reproductive cycle with molting and spawning in alternating years (Comeau and Savoie 2002), was decreased

Table 1. Proportion of males and non–egg bearing females available to the fishery for the first time (i.e., molting above the minimal legal size; Comeau and Savoie 2001) and the proportion of legal-size first-time-matured females available to the fishery during their first spawning year (year 1) and larval releasing year (year 2) (scuba observations, M. Comeau pers. comm.; Comeau and Savoie 2002; Comeau et al. 2004, 2008) for each of the 20 time periods.

	Time period	Proportion of males and non–egg bearing females	Year 1	Year 2
January-June	1	0.00	1.00	0.00
July	2	0.11	1.00	0.17
	3	0.13	1.00	0.17
	4	0.14	0.99	0.20
	5	0.17	0.97	0.25
August	6	0.20	0.90	0.33
	7	0.25	0.79	0.50
	8	0.33	0.68	1.00
	9	0.50	0.47	1.00
September-December	10-20	1.00	0.00	1.00

to a molt every two years, and a molt every three years for females larger than 120 mm CL (Waddy and Aiken 1986). There is little information available on molt frequency for males larger than 100 mm in the sGSL but there should be longer intervals for larger animals (Campbell 1983). Thus, the molt frequency for males larger than 120 mm CL was decreased to a molt every two years.

The CL (mm) was converted to corresponding weight (g) for each sex with the length-weight relationships:

females, $Weight = 0.0015CL^{2.857}$

males, $Weight = 0.0010CL^{2.9504}$

taken from Mallet et al. (2006) to estimate growth in weight.

Socioeconomic module

The socioeconomic module is made of four sub-modules (i.e., markets and prices, fishing enterprises, fishery management, and economic impact) that represent the human component in the lobster fishery (Fig. 1). The module is dynamic and has the ability to track the distribution of impacts for up to 20 years; hence, one can calculate, for example, the return on investment (ROI) and internal rate of return (IRR).

Markets and prices sub-module

The market and prices sub-module uses log-log equations to assess landing prices. The general equations for canner and market lobsters were as follows:

Canner: $Log\ P_c = a_0 + a_1 LogEG + a_2 LogW + a_3 LogWD$

Market: $Log\ P_m = a_0 + a_1 LogEG + a_2 LogW + a_3 LogWD$

where P_c was the landing price of canner-size lobster (i.e., lobster ranging from 70 to 80 mm CL), *EG* was the economic growth in Canada and the USA used as a proxy for the demand in North America, *W* was total lobster landings, *WD* was the world demand for lobster, P_m was the landing price of market-size lobster (i.e., lobster with a CL of 81 mm and over), and *a* represents parameters. The following parameters (elasticity coefficients) were used in the general equations:

Canner: $Log\ P_c = 0.98 + 1.19 LogEG - 1.95 LogW + 0.54 LogWD$

Market: $Log\ P_m = 3.16 + 0.86 LogEG - 1.71 LogW + 0.54 LogWD$

Hence, the price for canner and market lobsters (a_2) will decline by 1.95% and 1.71% respectively for every 1.00% increase in landings (measured in metric tons). The elasticities for demand (both for North America and the world) were set to zero for our case study as the example related to changes in supply (landings for all of Canada) and did not look at the demand. Hence, it was assumed that demand remained constant. Nonetheless, although the price elasticity for world demand (as measured by exports) is the same for both categories of lobsters (i.e., $a_3 = 0.54$), it is different with regard to economic growth (as measured by the Canadian gross domestic product). The elasticity for canner (a_1) is 1.19 which suggests that the buyers of canner lobsters are more sensitive to economic conditions than buyers of market-sized lobsters ($a_1 = 0.86$).

It should also be noted that the effect on lobster prices in our case study is minimal. The reasons are that we used Lobster Fishing Area 25, which represents only 7% of all landings in Canada, and raising the MLS from 70 to 72 mm CL results in lower landings for canner (i.e., higher prices) and higher landings for market-sized lobster (i.e., resulting in lower prices) with the price changes basically canceling each other.

Fishing enterprises sub-module

The fishing enterprises sub-module models both expenditures by main input categories and revenues. This sub-module makes it possible to assess the impact of measures to rationalize fishing fleets and reduce fishing effort on the net revenues of lobster fishing enterprises. The profits equation was defined as:

$$Profits = R - E$$

where R was the fishing revenues from landed lobsters, and E was the total expenditures. The equation for the fishing revenues was as follows:

$$R = (L_c \times P_c) + (L_m \times P_m)$$

where L_c was total landings of canner-size lobster in pounds, L_m was total landings of market-size lobster in pounds, P_c was the price per pound of canner-size lobster paid to lobster fishing enterprises, and P_m was the price per pound of market-size lobster paid to lobster fishing enterprises. The general equation for the total expenditures was as follows:

$$E = FC + LC + VC$$

where FC was the fixed costs established at $16,568 per fishing enterprise (DFO 2010a), LC was the labor cost, and VC was the variable cost (e.g., fuel, bait, repairs, fishing gear). The following coefficients were used in the total expenditures general equation:

$$LogLC = 3.44 + 0.24 LogNT$$

$$LogVC = 2.42 + 0.72 LogNT$$

where NT was the number of traps per fishing enterprise.

Fishery management sub-module

The fishery management sub-module is able to assess the impact of various management conservation measures (e.g., minimum and maximum legal size, window-size for females, changes to the length of the fishing season, number of traps) on the biomass and fishing enterprises revenues.

Economic impact sub-module

The economic impact sub-module is based on an input-output (I/O) core integrated with econometric modules (LeBreton 1985, 1986a,b). This approach combines the impacts by industry of the I/O models with the nonlinear estimation and the broad scope of econometric models. The I/O core is based on algorithms (Statistics Canada 1984), methodologies, and I/O data purchased from the National Accounts Division of Statistics Canada for 2005 (Statistics Canada 2010).

It operates with both open (indirect impacts) and closed (induced impacts: consumer expenditures) versions to calculate impacts. Statistics generated by the sub-module include: gross domestic product (GDP); federal and provincial tax revenues; employment; energy utilization (by source of energy, in gigajoules (Gj) and volume), and greenhouse gas production.

It includes a cost-benefit sub-sub-module (SSM). This SSM performs cost-benefit analysis, including the calculation of IRR and net present value (NPV) statistics over 10 and 20 years.

Input parameters for the lobster population dynamics module

The lobster size distribution used in the LPD module (Fig. 2) is based on size distribution presented at the 2004 and 2007 stock assessment reviews for animals in their first year in the fishery (Comeau et al. 2004, 2008). Three runs were done to simulate an increase in the MLS of lobsters from 70 to 72 mm in one year (scenario 1) or two years (i.e., 1 mm per year; scenario 2) versus the status quo. The LPD module was used as an egg-per-recruit model in this paper and all simulations were done with an initial total of 100,000 lobsters from 60 to 69 mm CL for the first time period (i.e., January–end of June), and no berried female.

There is a rather complex relation between mature females and fishing mortality. Simultaneously within a given size-group of mature females, both berried females not eligible for the fishery and females that have released their larvae and are eligible, could be observed. Hence, females that reach sexual maturity in a given year start to extrude eggs the third week of July (Table 1) of the following year. By the end of August–early September, all females are berried and therefore not eligible for the fishery. Hence, the value of the parameter becomes zero (Table 1). It will start to rise again the following year around the second week of July as larval release starts. Here, 17% of females of that cohort will be eligible for a fishery during the third week of July because they are no longer berried. By the third week of August, all females will have molted and mated and they are all eligible for the fishery, which is reflected in the value of the parameter (value of one; Table 1).

Input parameters for the socioeconomic module

A reduction of fishing effort was simulated by reducing the number of traps allowed per lobster fishing permit by 20% from a maximum number of traps presently allowed from 250 to 200. In order to identify

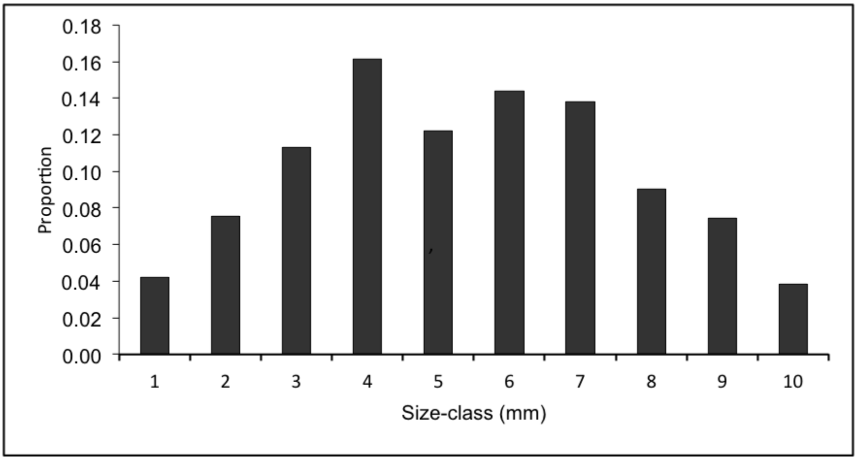

Figure 2. The number of lobsters (*Homarus americanus*) in the model is divided into ten size classes based on a flat normal distribution of lobsters of a given size class, where vertical axis represents shares of the total population for a given age group. (Based on information from Comeau et al. 2004 and 2008)

the direct impact of this 20% trap reduction on net incomes, benefits for both lobster fishing enterprises and the economy as a whole, and the impact on energy consumption and greenhouse gas production, simulations through the fishing enterprise and the economic impact sub-modules were calculated. The cost structure of lobster fishing enterprises for the summer fishery was obtained from DFO (2008) based on surveys conducted with owners of lobster fishing enterprises. The calculations are also based on a total number of 708 commercial fishing licenses and a value of $125,000 per license based on information from the Maritime Fishermen Union representing the majority of the harvesters from the summer fishery (Martin Mallet, Maritime Fishermen Union, Shediac, New Brunswick, pers. comm.). Information on the value of a license directly from the industry better reflects its true commercial value because lobster fishing licenses sold between harvesters include the value of the boat and fishing gear. The annual license fee paid to DFO for this limited entry fishery is only $100 for the summer fishery. Finally, both fixed and financial costs (on loans taken on to buy the boat and other major equipment) are assumed to remain unchanged, at least over the short term.

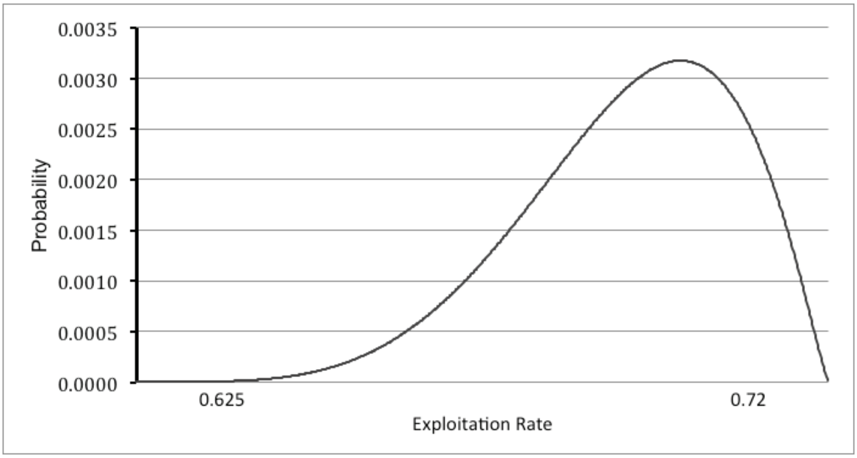

Figure 3. Beta distribution of the exploitation rates used in the Monte Carlo simulations.

Uncertainties

A Monte Carlo model was used to generate a range of potential outcomes in the LPD module to account for the variability of the exploitation and mortality rates. A total of 100,000 simulations were performed (50,000 for each rate). Values of the egg production and landings are in absolute numbers. The statistical distribution of the exploitation rate was assumed to be a Beta distribution (Fig. 3) with a mean value of 0.70. The statistical distribution for the natural mortality rate was assumed to be a Gamma distribution (Fig. 4) with a mean value of 0.10.

Monte Carlo simulations for landing prices (50,000 simulations) were allowed to vary between the maximum and minimum prices over the last 26 years. The statistical distribution used will follow a modified, rather flat, Beta distribution that dips toward both extremities (Fig. 5). Price for canner-size lobsters varied between $1.58 and $4.97 and price for market-size lobsters varied between $2.50 and $5.89 (shown in Fig. 5).

Results

Increasing the minimum legal size from 70 to 72 mm carapace length

Simulations showed that an increase in the weighted average landings from 42.0 t to 42.7 t and 43.1 t could be anticipated for the first and sec-

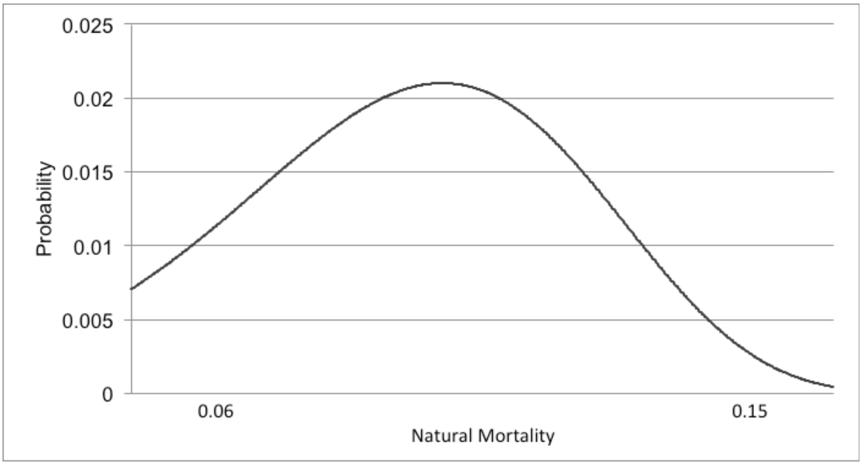

Figure 4. Gamma distribution of the molt-related mortality used in the Monte Carlo simulations.

ond scenario, respectively (Table 2). This increase in landings associated with both scenarios would occur over a 5 to 6 year period (Table 3, Fig. 6). For the first scenario, simulations indicated that landings decrease the first year by 383 kg, but increase by 361 kg the second year, followed by a further increase of 354 kg in year 3 (Table 3). Therefore after year 3, landings have increased. For the second scenario, the decline in landings is more important the first year with a drop of 1,357 kg (Table 3). However, similar to the first scenario, landings are positive after year 3 with an increase of 990 and 799 kg by the second and third year, respectively (Table 3). In both scenarios, significant increases in landings are found up to 6 years after initial size increase (Fig. 6).

The increase in egg production was more important than landings in our simulations. The weighted average number of eggs produced increases from 188.9 million for a MLS of 70 mm to 209.3 and 222.9 million in the first and second scenario, respectively (Table 2). The increase in egg production will take at least 3 years (Table 3) mainly because of the 2-year reproductive cycle.

Revenues for fishing enterprises based on simulations from the market and prices sub-module with landings input are expected to increase from $330,700 to $338,600 and $343,600 for the first and second scenario, respectively (Table 2). For the first scenario, revenues diminish the first year by $3,200 but increase by $3,600 the second year, followed by another increase of $3,300 in the third year (Table 3).

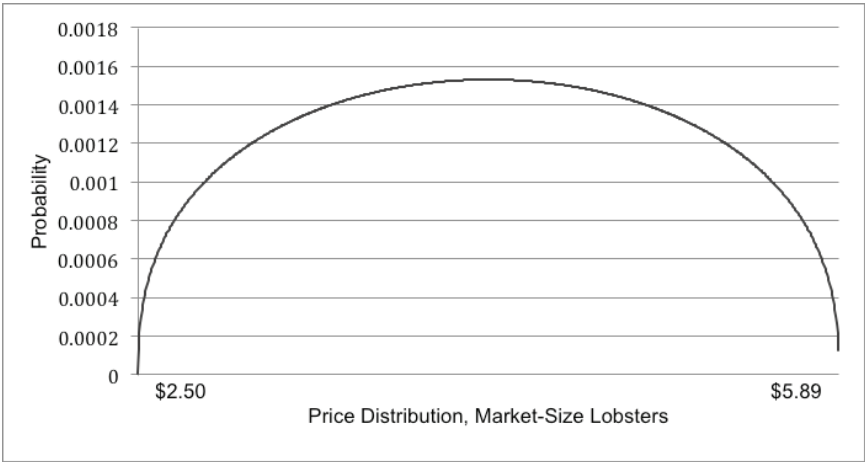

Figure 5. Beta distribution of lobster *(Homarus americanus)* landing prices used in the Monte Carlo simulations. The price distribution for market-size lobsters (i.e., lobsters 81 mm carapace length and over) is shown. The price distribution for canner-size lobsters (i.e., 70-80 mm carapace length) is similar and varies between $1.58 and $4.97.

Therefore, cumulative revenues become positive after year 2. For the second scenario, revenues drop by $9,700 the first year, but increase to $8,500 in year 2 and $7,400 in year 3 (Table 3). Hence, cumulative revenues become positive after year 3, one year later than scenario 1, but with a higher total increase (Table 2).

Reducing the number of lobster traps

Simulations showed that a reduction of the number of traps by 20% from 250 to 200 would increase the net after-tax revenues for lobster fishing enterprises in the summer fishery from $4.21 to $5.70 million (Table 4). To achieve a similar level of increased profitability, at least 5% to 6% of all fishing licenses would have to be retired, or about 40 licenses. At $125,000 per license, the cost would be $5 million.

The direct impacts are expected to involve lower employment levels from 443 to 421 (Table 4), a 5% reduction in terms of the number of hours worked by the deckhands. The GDP, because mostly of increased profits, would increase from $12.18 to $13.60 million (Table 4). Direct federal and provincial tax revenues would increase from $1.03 million to $1.23 million (Table 4), an increase generated by higher revenues

Table 2. Weighted average of landings (t), egg production (million), and revenues for fishing enterprises ($ thousand) for an increase of the minimum legal size of lobster (*Homarus americanus*) from 70 to 72 mm carapace length (CL) in two successive years (1 mm per year) or in one year for a summer fishery.

	Size and size increases		
	70 (mm CL)	71 and 72 (mm CL)	72 (mm CL)
Landings	42.0 (35.1-49.1)	42.7 (35.6-50.0)	43.1 (36.0-50.5)
Percent difference (vs. 70 mm)	—	1.7% (1.5-1.8)	2.7% (2.5-3.0)
Egg production	188.9 (153.0-226.0)	209.3 (169.1-250.9)	222.9 (179.8-267.9)
Percent difference (vs. 70 mm)	—	10.8% (10.5-11.0)	18.0% (17.5-18.5)
Revenues for fishing enterprises	$330.7 (215.4-445.8)	$338.6 (219.9-457.4)	$343.6 (223.1-464.5)
Percent difference (vs. 70 mm)	—	2.4% (2.1-2.6)	3.9% (3.6-4.2)

Landings and egg production values are based on simulations with an initial 100,000 animals ranging from 60 to 69 mm CL. The 75% probability interval estimated from Monte Carlo simulations is shown in parenthesis.

Table 3. Differential in landings (kg), egg production (million), and revenues for fishing enterprises ($ thousand) per year following the initial increased of the minimum legal size of lobster (*Homarus americanus*) from 70 to 72 mm carapace length (CL) in two successive years (1 mm per year) or in one year for a summer fishery.

	Years following size increases				
	1	2	3	4	5+
Landings					
70 to 71 to 72 mm CL	–383	361	354	197	171
70 to 72 mm CL	–1,357	990	799	345	323
Egg production					
70 to 71 to 72 mm CL	0.0	0.0	5.6	3.1	11.7
70 to 72 mm CL	0.0	0.0	9.8	5.0	19.2
Revenues for fishing enterprises					
70 to 71 to 72 mm CL	–3.2	3.6	3.3	1.9	2.3
70 to 72 mm CL	–9.7	8.5	7.4	3.4	3.3

Landings and egg production values are based on simulations with an initial 100,000 animals ranging in size from 60 to 69 mm CL.

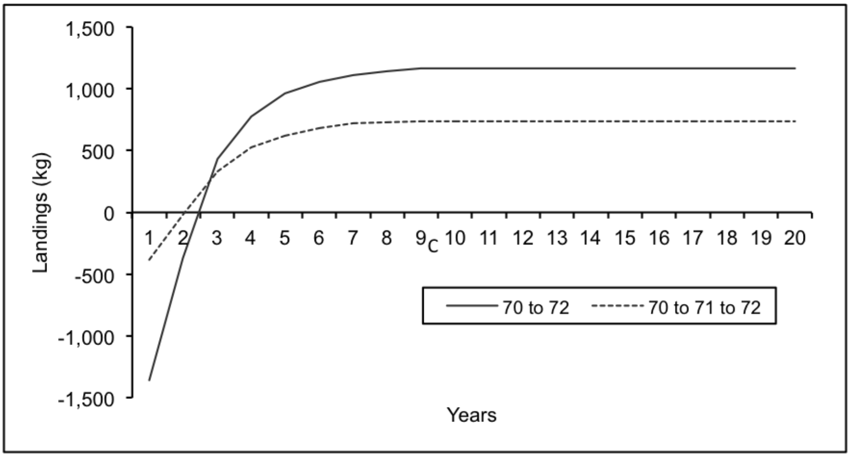

Figure 6. Comparison of cumulative differential for lobster (*Homarus americanus*) landings (kg) from a single recruitment cohort (3 age classes) over a 15 year period for an increase of minimum legal size from 70 to 72 mm carapace length in 1 year (solid line); and over two years, 1 mm per year (dotted line), for a summer fishery. The difference was 400 kg representing a difference of 57%.

for lobster fishing enterprises. Energy consumption would be lowered from 488.7 to 439.8 Gj (Table 4). The production of greenhouse gas, as expressed in kilotons (kT) of CO_2 equivalent, would decrease from 21.0 to 18.9 kT (Table 4).

Discussion

In the objectives of the Atlantic Lobster Sustainability Measures program, the long-term sustainability of the lobster fishery is linked to economic prosperity. Although DFO produces stock assessment of harvested species on a regular basis, the economic prosperity of fisheries has never been formally integrated into that process. This integration would be a major paradigm shift for fisheries managers responsible to implement conservation measures for the sustainability of fisheries. To address this need, the bio-economic model presented in this paper enables stakeholders and fisheries managers to concurrently assess the effect of management measure changes (e.g., increase MLS and effort reduction) and price fluctuations on the lobster population, lobster fishing enterprises and the economy as a whole for the first time. The model

Table 4. Net revenues of lobster (*Homarus americanus*) fishing enterprises and direct economic impacts for a reduction of lobster traps by 20% from 250 and 200.

	250 traps	200 traps	Percent change
After-tax net revenues	$4.21	$5.70	35.4%
Employment	443	421	–5.0%
Gross domestic product	$12.18	$13.60	11.6%
Tax revenues	$1.03	$1.23	19.0%
Energy consumed (Gj)	488.7	439.8	–10.0%
GHG production (kT)	21.0	18.9	–10.0%

$ = million. GHG = greenhouse gases. Gj = gigajoules. kT = kiloton CO_2.

showed that both lobster biology and human activities are closely linked with the major drivers being revenues and profits (i.e., the actual trigger for the ALSM program). The economic prosperity of stakeholders (harvesters, communities, processors, etc.) in a sustainable fishery framework depends on two main elements: an adequate biomass and markets. Prices are determined by the dynamic relationship between these two elements. Thus, a bidirectional model integrating both biological and economic parameters represents a better and integrated approach to provide information and make recommendations to fisheries managers.

The bio-economic model makes it possible to determine the biological, social, and economic impacts of a number of issues facing the lobster fishery. It could be used to assess changes in conservation measures. In terms of the lobster population, an increase in the MLS produced, as anticipated, an increase in egg production. Based on an egg-per-recruit model, similar results have already been presented (Lanteigne et al. 1998). Indeed, female maturity follows a logistic curve (Comeau and Savoie 2002) and fecundity is exponential (Campbell and Robinson 1983) resulting in a significantly higher number of eggs produced for a small increase in female sizes close to the SOM_{50} (72 mm CL). Reaching the SOM_{50} as recommended by the FRCC (2007) also fulfill one of the expected results of the sustainability plan that harvester associations have to submit before accessing self-rationalization funding under the ALSM program.

The major difference between increasing the MLS over 1 year instead of 2 years is the greater long-term gain for both the egg production and landings. In fact, annual recruitment of legal size animals to the fishery fluctuates every year (Comeau et al. 2008) and to avoid a drop in landings, which could negatively affect the revenues for harvesters, the increase of the MLS should be implemented to correspond with an increase in the biomass. Similarly, an increase of the MLS with a strong demand for lobster could create an increase in the revenue of fishing enterprises.

Reducing the number of traps has a solely beneficial effect on the net revenues of fishing enterprises because the level of reduction only accounts for empty traps (Comeau et al. 2004, 2008). A reduction of the exploitation level in the short term should also not be anticipated (Comeau et al. 2004, 2008). Although it has been shown that more that 50% of traps are empty during the first week of the summer fishery, with a much higher number for the entire fishing season (Comeau et al. 2008), harvesters have always been reluctant to reduce their individual allocation. Through extensive lobster advisory meetings, fisheries managers were unable to reduce the nominal effort based on biological reference points. Fishery managers would now be able to rationalize a reduction of effort as an economic benefit for harvesters based on the results generated by our bio-economic model.

The model could also be a useful tool for the industry and DFO to address and assess self-restructuring projects of the ALSM program. In the eligible expenditures of this program, government funds could be used by the lobster industry as a buyback to retire commercial licenses and/or reduce the number of traps. The model can provide useful guidelines about which approach can be used to reduce effort for rationalizing the lobster fishery from the lobster population and fishing enterprise perspective. The increase of net revenues of fishing enterprises by 35% could be explained by a reduction of the expenses without a reduction of the gross income from landings, and hence have no effect on the lobster population. In terms of profitability, 40 licenses would need to be removed at a total cost of $5 million based on a value of $125,000 per license. Thus, a total contribution of about $2.5 million from the ALSM program, based on 50% eligible expenditures for the lobster fishing area 25, would be needed for a buyback of 40 licenses compared to nil by reducing the number of traps by 20%.

It is imperative that the time periods used in the model match the important biological events (i.e., growth, natural mortality, spawning, larval release) of the fishery in question; therefore more detailed time periods representing the summer months were used in the current model. An earlier study (Comeau and Savoie 2002) concluded that current models to manage the fishery (FRCC 1995, Lanteigne et al. 1998) were not fully capturing the biological processes associated with the summer fishery. The egg-per-recruit model in use (FRCC 1995) is based on dividing the year into four periods, and assumes a mated female is only subjected to the fishery once before extruding eggs. In contrast, a mated female in a summer fishery is subjected to a first full-year of fishing (as in a spring fishery) and then large numbers of mated females are removed during the first weeks of the second fishing season before egg extrusion occurs (Comeau et al. 2004, 2008). The failure to account for this second year of removals results in an overestimation of egg production. The current model (FRCC 1995, Lanteigne et al. 1998) makes no accommodation for the differences in mated female mortality between the spring and summer fisheries; the same assumptions are used and this seriously overestimates the egg-per-recruit. The model used here corrects for this mismatch between the actual biological events in the summer fishery and the previously used assumptions.

Inputs into the lobster population dynamics (LPD) module from the socioeconomic module come from two sub-modules. First, the fisheries management sub-module regulates most of the interactions between the lobster biomass and humans. The second sub-module from the socioeconomic module, which provides inputs into the LPD module, is the lobster fishing enterprise sub-module. The fishing enterprises sub-module could be refined with a sub-sub-module (SSM) to take into account the competitive nature of this input fishery and the decision-making processes in terms of fishing strategies (i.e., effective effort) of harvesters to maximize or increase their net income. It is recognized that the possible lack of success of input management regimes arises from the incentive for harvesters to substitute control inputs by uncontrolled inputs to achieve, or increase, their landings, and the lack of ownership or stewardship of the fishery resource (Grafton et al. 2006). In an input fishery, harvesters could/would modify their fishing strategies and behavior by developing tactics, including technological innovations, to obtain greater catch than their fellow harvesters. This difference in catch rates has been defined as relative fishing efficiency or fishing performance (Pálsson and Durrenberger 1982, Hilborn 1985) and is directly related to the effective effort. Information on effective effort and its changes could be gathered by questionnaires or interviews directly to harvesters about their fishing habits, strategies, and fishing gear (Lanteigne 1999) to quantify the effective effort. In our model, a link from the fishing enterprises sub-module to the fishery component of the LPD module (Fig. 1) could be activated to better simulate landings incorporating social anthropology inputs.

Another refinement to the fishing enterprise sub-module would be to specifically take into account the Canadian Employment Insurance (EI) Program in relation to the fishing industry. The Fisheries Resource Conservation Council (FRCC 2007) indicated that although the number of active licenses should decrease when fishing yields low catches, this is not the case because the EI system acts as a buffer, enabling harvesters to maintain a presence particularly in a declining lobster population. Thus, other than the incentive to increase their revenues by increasing landings, harvesters in Canada could adjust their fishing activities to qualify for EI benefits (Roy 1998). An EI SSM already exists in the economic impact sub-module for seasonal workers to model the economic impact for the economy as a whole. However, EI input must be adapted to take into account employment insurance fishing benefits, because harvesters are considered a self-employed person engaged in fishing and can receive EI benefits. No other self-employed Canadian has that privilege. Hence, this economic incentive has to be addressed in the fishing enterprises sub-module (Fig. 1) because it could modify fishing strategies (e.g., effective effort) and affect estimates generated from the bio-economic model simulations.

A module dealing with ecosystem services and linked to the natural mortality input within the LPD module and to the fisheries management sub-module, in terms of governance structure for decision-making, could be developed and added to our bio-economic model for a more holistic approach to ecosystem-based fisheries management. Ecosystem services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life (Daily 1997). Within a fisheries management framework, an ecosystem service is primarily considered the fish stock and its sustainable yield over time, while the scope of a stock assessment is typically limited to the fishing fleet as the driver of changes (fishing mortality) and the stock status (biomass). However, there are other drivers of anthropogenic activities (including economic, social, and cultural) that can have direct and indirect effects on ecosystem components and processes that can undermine this ecosystem service (Cormier et al. 2010). To ensure the most effective accountability and governance structure, with inclusiveness and transparency, an ecosystem services module would need a formalized risk analysis process to enhance the effectiveness of management measures (Cormier et al. 2010). Hence, information on land-based anthropogenic activities that could alter the ecosystem service (e.g., water quality, habitat alteration, disruption, or destruction) needs to be included in an ecosystem-based fisheries management framework.

Each module of the bio-economic model provides essential and complementary information for bidirectional linkages between anthropogenic activities, including economic objectives that go beyond biological and fishery parameters normally built into fishery assessment models, and the lobster biology. The LPD module is required to generate information on the number and size of lobsters landed by year, and the socioeconomic module provides revenues for harvesters linked to modifications of management measures for the lobster industry and the economy as a whole. The bio-economic model presented in this paper shows that an integrated approach for a successful ecosystem-based fisheries management based on both biological and economic objectives is required for the lobster fishery.

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Natural Indicators of Salmon Run Timing and Abundance

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Abstract

Natural indicators are empirical observations that correlate with specific ecological phenomena. These ecosystem-based observations have been made by Alaska Native fishermen and women over numerous generations and have been critical to their success and, ultimately, their survival. While local and traditional knowledge (LTK) can aid in the understanding of environmental variability that influences fluctuations in almost all animal and plant populations, our focus is on Pacific salmon returning to the Yukon River drainage. Inputs of Pacific salmon to the ecosystem have been shown to be important for the success of many species, with recent studies demonstrating that diversity within a salmon population is important for the long-term sustainability of a salmon population. Proper fisheries management that optimizes the diversity within a salmon population will ultimately be beneficial to all plant and animal species in the area. Our overall goal focuses on how LTK can assist in a more holistic way with the understanding of salmon abundance, run timing, and population health. The objective of the project component presented here is to begin to understand the mechanisms that allow natural indicators to be of value as well as to identify specific data sources for further evaluation.

Introduction

Yukon River fishers have relied on local and traditional ecological knowledge (LTK) to predict salmon run timing and abundance for generations (Moncrieff et al. 2009). This knowledge is based on observations of the natural surroundings and has been passed from one generation to the next in an oral fashion. LTK is typically specific to an area. For example, fishermen in the middle Yukon River have long correlated the release of *Populus* spp. seeds or "cotton" with the arrival of Chinook salmon (*Oncorhynchus tshawytscha*) while fishermen at the mouth of the river use observations of migrating birds to indicate salmon arrival. Environmental observations that appear to correlate with salmon run timing and behavior, as well as aspects of the subsistence round (annual cycle of harvesting, processing, and storing of subsistence food) are termed "natural indicators" by the authors. A person who has learned the observation techniques from their Elders and has actively practiced these observation techniques throughout their life is a "natural indicator practitioner."

This project began when the nonprofit Yukon River Drainage Fisheries Association (YRDFA) and the Alaska Department of Fish and Game, Subsistence Division (ADFG), interviewed 61 knowledgeable Elders and active fishers in the five lower and middle Yukon River villages of Hooper Bay, Emmonak, St. Mary's, Grayling, and Kaltag (Fig. 1) about natural indicators they use to predict salmon run timing and abundance (Moncrieff et al. 2009). Natural indicators are defined as empirical observations of the environment that correlate with specific ecological phenomena. This initial study demonstrated that many fishermen and women still make observations of the phenology of the plants, birds, and salmon, and use these observations to improve their success with salmon fishing.

Local and traditional knowledge recognizes a variety of relationships in the natural environment and it was thought that combining LTK with run timing and run strength indicators collected by ADFG, Division of Commercial Fisheries, may improve the accuracy of inseason fisheries management decisions. Mundy (1982) found that the timing of Chinook salmon migration into the Yukon River may depend on factors related to air temperature, while more recently Mundy and Evenson (2011) found significant correlations between the entry timing of Chinook salmon into the Yukon River with spring sea surface temperatures, spring ice cover, and spring air temperatures in the vicinity of the mouth of the Yukon River. Other researchers (Ruggerone 2004, Anderson and Beer 2009, Blackbourn 1987) have also found useful correlations between salmon timing and environmental variables. While LTK typically does not directly measure the environmental variables examined in these previous studies, it does provide observations of other physical factors and biological taxa that may be influenced by the same environmental inputs. Salmon in the Yukon River are rarely observed unless caught and the ability to infer salmon behavior using readily available observations has tremendous advantages.

Natural indicators or LTK can add to the fishery managers' tool kit by informing questions about salmon behavior. The results of Moncrieff

Figure 1. Map of Alaska portion of the Yukon River showing the study communities of Hooper Bay, Emmonak, St. Mary's, Grayling, and Kaltag.

et al. 2009 was discussed at two meetings between LTK holders from the Yukon River, Western scientists, and Yukon River managers in May and October 2010 with the purpose of identifying areas where LTK and traditional fisheries science may be integrated. This paper discusses the result of those meetings and identifies the study questions that showed the greatest potential for obtaining a better understanding of the mechanisms behind the natural indicators and Yukon River salmon migratory behavior and abundance.

Local and traditional ecological knowledge

Today's local and traditional knowledge is built on hundreds or thousand of years of environmental observations, including weather, plants, birds, and other animals, and has been passed down through the generations (Berkes 2008). This knowledge was the primary source of information that guided people in preparation for the salmon arrival, and the timing and location of other subsistence resources prior to Western contact and influence. Generations ago, the knowledgeable Elders were the scientists and teachers for their communities, and were turned to for advice and guidance—a role that is still carried on today.

There is growing acknowledgement from the science community that LTK may provide informed ecological understanding and value in addressing modern-day environmental issues (Berkes 2008). Scientists are interested in LTK partly because of its rich time-depth and place-based value. Applied environmental anthropologists are promoting the value of community-based alternatives to top-down management by government agencies, and the incorporation of observations and knowledge of fishers into fisheries management (McCay 2001, Smith 1982, Wilson and Kleban 1992). There are challenges in working with LTK, including assessing the information that is often encoded in cultural beliefs and behavior (Huntington 2000), and how to make that knowledge compatible with scientific research and management (Nadasdy 1999). In today's environment, it makes sense to include local people in management discussions because they spend more time on the river or land and can see more things and for longer periods of time. LTK holders can bring a different perspective to the discussion and can compliment the tools and techniques of scientists (Usher 2000).

Materials and methods

This project is in a research scoping phase in which we facilitated discussions between LTK holders from the Yukon River and scientists whose expertise correlates with each natural indicator to be examined. The goal of this phase was to select a set of projects that showed a promise for immediate results, to prepare to implement those projects, and to secure working partners and funding. The first meeting took place in May 2010 in Fairbanks and was attended by two scientists for every LTK holder from the Yukon River. During this meeting the natural indicators were discussed at length and clarified by a question and answer session. Between May and October, environmental variables and potential data sets were identified that correlated to each natural indicator, and limited research into the robustness and practical use of each data set was conducted. A second meeting was held in Anchorage in October in which the same LTK holders attended along with scientists selected to match best with the natural indicator projects that had the most potential for immediate research. The October 2010 meeting focused on three projects that met the criteria: useful to fishers; useful to managers; with available data sets; researchable questions; and potentially fundable.

Common to all of the projects envisioned is the fisheries information collected and maintained by the Alaska Department of Fish and Game. Run timing information is available from test fisheries near Emmonak (1989-present), a village near the mouth of the Yukon River, and the Pilot Station sonar site (1986-present; ADFG unpubl.). Abundance information from Pilot Station (1995 and 1997-2009) and Eagle (2005-2009) is also available (JTC 2010).

Results

According to LTK, Chinook salmon entry into the Yukon River is correlated with the appearance of migrating birds, a certain level of plant growth, and winter winds along the Bering Sea coast of western Alaska. A greater understanding of the seasonal phenology may show a relationship that is useful information for salmon managers who base regulatory decisions on Chinook salmon run-timing. Migrating birds, plant growth, and winds were selected from the long list of natural indicators observed by fishers from the Yukon River because of their potential to quickly provide information to assist fishery managers and the availability of appropriate data sets.

Birds

Natural indicator observations from Yukon River fishers report that migrating bird arrival indicates Chinook salmon arrival, migrating bird abundance correlates with salmon abundance, and the duration of migrating bird presence correlates with salmon abundance. In addition, Yukon River fishers report that clutch size of sandpiper or snipe is an indicator of salmon abundance (Moncrieff et al. 2009).

In Emmonak and St. Mary’s, fishers report that migrating bird arrival timing correlates with Chinook salmon arrival timing. Villagers anticipate the arrival of Chinook salmon when they see the migrating white fronted geese (*Anser albifrons frontalis*) or Taverner Canada geese (*Branta hutchinsii taverneri*). In both villages, observations of the migrating bird arrival timing, speed, arrival patterns, and path are closely watched and Chinook salmon reportedly mimic the birds’ behavior. In Hooper Bay, Emmonak, and Mountain Village fishers observe the cliff swallows (Hirundinidae) as an indicator of Chinook salmon arrival. Fishers say that the activity level, behavior, and arrival timing of the cliff swallows are indicators of Chinook salmon arrival timing.

The Yukon Delta is one of the largest migratory waterfowl–producing areas in the world. The U.S. Fish and Wildlife Service (USFWS) has a long history of conducting field research on migratory birds in the Yukon-Kuskokwim Delta and have had annual field projects operating on a consistent basis there since 1985 (Fischer et al. 2009). Past studies have demonstrated the effects of weather on migratory timing (Lindberg et al. 1997; Dau and Mickelson 1979; Hupp et al. 2006, 2008) and it is quite possible that the same weather phenomena that are factors in bird migration also play a role in the migration of Chinook salmon, making migratory birds possible indicators of salmon migration.

Migratory bird nesting information is published in Fisher et al. 2009 and bird arrival and presence information by species is kept in field notebooks at several sites in the Yukon Delta. These data have the potential of providing arrival timing of migratory birds, which could be compared with Chinook salmon arrival timing to determine if relationships exist.

A preliminary examination of the usefulness of using cackling goose (*Branta hutchinsii*) hatch timing to predict Chinook salmon run timing into the Yukon River shows a significant correlation ($p < 0.001$) but is probably not useful for predicting Chinook salmon arrival (Fig. 2). While the hatch dates for cackling goose generally occur after the critical arrival points for Chinook salmon management, the strong correlation between the data sets supports the LTK and indicates there may be more useful information available.

In St. Mary's and Koyukuk fishers report that migrating bird abundance indicates salmon abundance in their communities. In St. Mary's fishers also note that the longer the migrating birds stay around their community, the more salmon there will be that season. Emmonak fishers watch nesting birds for clutch size, which they use as an indicator for salmon abundance. To investigate this series of questions, bird abundance records from the USFWS Yukon-Kuskokwim Delta could be compared to Chinook salmon arrival timing and abundance records from ADFG.

Plants

Natural indicator practitioners from most of the communities included in the study by Moncrieff et al. (2009) reported that Chinook salmon arrival correlated with plant growth and salmon abundance correlated with the abundance of plant growth. A better understanding of the mechanisms behind the phenology of plants may lead to better predictions of salmon migratory behavior.

Plant growth; including appearance of flowers in Kaltag; cotton blowing (north of Alaska Range = balsam poplar *Populus balsamifera,* south of Alaska Range = black cottonwood *Populus tricocarpa*) in St. Mary's, Grayling, and Kaltag; and grass height, willow and alder leafing out, and rhubarb (*Rheum rhabarbarum*) growth in St. Mary's, Emmonak, and Kaltag are all indicators of Chinook salmon arrival timing. Plant growth can be an indicator of salmon abundance in Grayling where natural indicator practitioners watch for the amount of cotton produced, and in St. Mary's where they watch the grass height and other plants.

Under normal growing conditions, the development rate from emergence to maturity for locally adapted plants depends primarily on

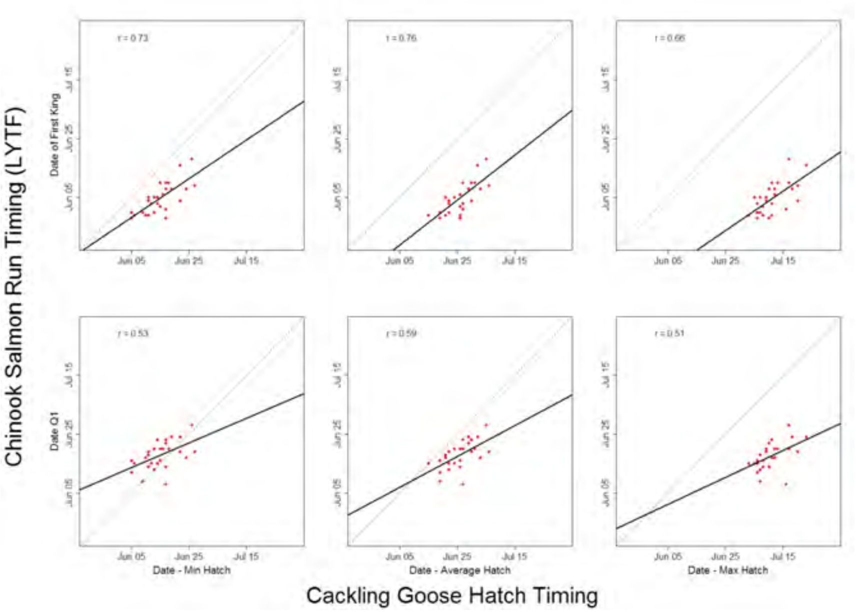

Figure 2. Relationship of Chinook salmon run timing at the Lower Yukon River test fishery to hatch timing for cackling goose in the Yukon-Kuskokwim Delta for the 1985-2009 seasons. The solid line indicates the best linear fit to the data while the fine dotted line indicates where salmon run timing and hatch timing occur on the same date.

the daily air temperature (Glenn Juday, University of Alaska Fairbanks, pers. comm.). Because many developmental events of plants depend on the accumulation of specific quantities of heat, it is possible to predict when these events should occur during a growing season regardless of differences in temperatures from year to year. The accepted method of estimating the amount of heat accumulated by plants is growing degree days or growing degree units (Womach 2005).

Work may need to be done to determine the number of growing degree days required to get the indicator plants to the appropriate growing stage. The historical database of temperature data maintained by the National Oceanic and Atmospheric Administration (NOAA) could be used to estimate the date at which the critical value for growing degree days was reached for past years at a location. That date could then be compared to arrival timing information to determine whether a significant correlation existed. If a correlation exists, it would be relatively easy for fishery managers to monitor daily temperature information to estimate salmon arrival.

Wind

Fishers from the villages around the mouth of the Yukon River rely heavily on observations of wind direction and speed to predict which mouth of the Yukon River the salmon will enter. In the village of Emmonak, fishers report that dominant winds from the north during December, January, and February indicate salmon will primarily enter the south mouth while prevailing winds from the south during the same months indicate salmon will be most abundant in the north mouth.

NOAA has a long time series of wind data, which could be compared to salmon catches at the ADFG test fisheries in the south, middle, and north mouths of the Yukon River to determine if there is a relationship between prevailing wind direction in the winter and where salmon enter the Yukon River.

To pursue this study question, years of predominant north or south winter winds will be identified using NOAA wind data and compared to catch information from the ADFG Lower Yukon River Test Fishery. Wind data are recorded hourly in the Yukon River south mouth community of Emmonak (Fig. 3).

Database

In addition to the three science projects discussed above, a database is being developed that will be a repository for natural indicator observations. This database will be a place where Elders, fishers, students, and others can develop a record of quantitative information (annual observations) that can be tied to records of salmon variability and environmental variability. The database idea was presented at both 2010 natural indicator meetings and fully supported by both the fishers and the scientists. This database will be accessible online for registered users to input data. Natural indicator observations will be recorded with each participating village documenting information specific to their location. Examples of these data include (1) bird arrival information such as the number of birds by species by day, and the number of eggs in a nest; (2) insect information such as the type of insect, date observed, and abundance; (3) plant growth observations such as the date particular flowers appear, date that grass reaches knee height, date that *Populus* sp. cotton begins blowing, the relative abundance of cotton in air, date that willow and alder leaves fully expand, and rhubarb growth; (4) fish information such as sheefish (*Stenodus leucichthys*) presence and abundance, the date of arrival and abundance of salmon (*Oncorhynchus* spp.), smelt (*Thaleichthys pacificus*), and eel (*Lampetra tridentata*), and date of presence of phenotypic varieties of Chinook salmon (blueback, blacknose, whitenose); (5) river information including water level, change in

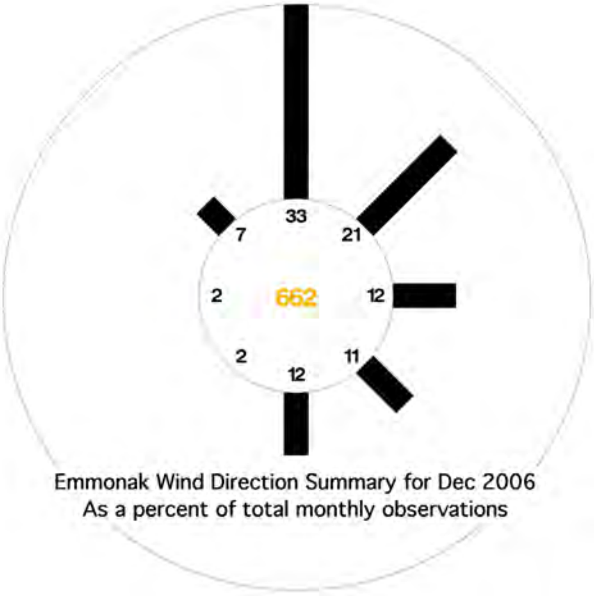

Figure 3. Summary of wind observations at Emmonak during December 2006. Of 662 hourly observations, 33% were from the north, 21% were from the northeast, and 12% were from the south.

water level, temperature, and date of snowmelt; and (6) observations of the moon to include the shape of the moon and position in the sky. The database may also include Elders' predictions or observations related to wind and other weather events.

Discussion

Salmon management on the Yukon River is a complicated and imperfect system. The length of the river, diversity of the user groups, and international treaty obligations all contribute to the complexity of the fishery and put a tremendous demand on state and federal managers. In addition, the unpredictable nature of Pacific salmon returns is not well understood, with recent declines in abundance putting even more pressure on the ecosystem and resource managers. Fishery scientists are beginning to look to LTK for insight in designing research studies and it is hoped that this project will be of assistance in those endeavors.

It has long been acknowledged that salmon provide nutrients and food for freshwater and terrestrial ecosystems (e.g., Cederholm 1999, Moore et al. 2008, Quinn et al. 2009), and in return, these ecosystems provide the habitat essential for the success of the freshwater life stages of salmon (e.g., Helfield and Naiman 2001, Mossop and Bradford 2004). It is logical that salmon, plants, and other animal species would be linked and that population characteristics such as timing and abundance would be shared. It is the strength of these relationships that will ultimately determine the usefulness of a natural indicator or group of indicators in predicting salmon arrival and abundance sufficiently well to improve fisheries management.

Alaska Native fishers have long relied on observations of their surroundings to predict when and where salmon can be found. Salmon was and still is an important food source for Alaska Native fishers, and the ability to accurately predict when and where to fish makes food gathering more efficient. The problem is that salmon are found in larger, mostly turbid waters in the lower Yukon River, which makes direct observation nearly impossible. The association of migratory bird arrival and state of plant growth to indicate when to fish, coupled with winter wind information indicating where to fish could be extremely important for the fisher's continued survival.

The authors recognize that migratory bird behavior, plant growth, and the direction of winter winds may not directly influence salmon. Rather, it is more likely that the same factors influencing salmon are also influencing other aspects of the environment. For example, spring temperatures dictate the timing of snow and ice melting on the tundra, which influences the timing of bird arrival and nesting. The same temperatures influence the timing of sea and river ice melting, which also determines migration timing of salmon. Plant growth generally reflects the amount of heat that has accumulated in an area, which relates directly to river and nearshore ocean conditions and ultimately salmon migration timing. How the direction of winter winds influences which river mouth salmon will arrive at is unknown to these authors but may be more apparent to others.

The successful incorporation of LTK into salmon management will require both the possessors of LTK and scientists to work together and to continue the dialog that has begun between the groups. Information exchange between cultures as well as scientific disciplines is essential if future work is going to be successful. While the authors of this paper acknowledge that some Western scientists are not ready for investigations based on or inspired by LTK, we also are aware that there is a growing group of scientists who are ready for this type of partnership and are looking for new approaches to help explain the phenomena taking place today. We are not looking for scientific confirmation of LTK; instead we are trying to understand the mechanisms that have made the observations of natural indicator practitioners reliable predictors of salmon run timing and abundance.

As the three proposed studies and database proceed, the authors of this project suggest that weather, and temperature in particular, may prove to be a common factor influencing run timing and other variables within the ecosystem. The birds, plant growth, and salmon may all be keyed to temperature influences. This leads to the concerns raised by LTK holders over their observed changes in the environment and concern by others for climate change. While beyond the scope of this paper, climate change and its resulting effects such as permafrost melt, increasing temperature, and wholesale changes to the environment raise more questions about the future of salmon management than answers.

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Progress on Implementing Ecosystem-Based Management in the Gulf of Maine

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Abstract

The 2009 Gulf of Maine Symposium—Advancing Ecosystem Research for the Future of the Gulf examined progress in implementation of the ecosystem approach. The Gulf of Maine is well studied and highly managed, so offers a useful case study. The context for implementation of an ecosystem approach has evolved recently through development of the international context, legislative changes in both U.S. and Canada, major realizations regarding climate change and the need for management in the face of change, and increased public interest.

There have been numerous advances in understanding and progress in networking of people and data/information. There has been an increase in the recognition of different spatial scales and greater appreciation of aspects related to conservation objectives of productivity, biodiversity, and habitat. There is explicit recognition that ecosystem approach to management (EAM) includes ecological and social/economic considerations and that it demands an interdisciplinary approach.

There is unquestionable evolution in management, and this shapes the priorities of research for the next few years. Research is required in support of development of an integrated approach to the management of multiple human activities, in relation to a more diverse set of objectives that include a higher standard of ecological integrity and diverse aspects of sustainability in the face of environmental change. Impediments to implementation of an EAM in the Gulf of Maine include diversity in definition/approach, complexity in jurisdiction, ecosystem complexity, the need for enhanced monitoring and information to

support evolving management landscape, and a need for institutional (governance) to support cross disciplinary and inter-jurisdictional considerations.

Introduction

The 2009 Gulf of Maine Symposium—Advancing Ecosystem Research for the Future of the Gulf was held in St. Andrews, New Brunswick, October 5-9, 2009. This symposium, initiated by the Regional Association for Research on the Gulf of Maine (RARGOM), was designed to share new developments in scientific knowledge and policy development regarding the ecosystem approach, and to identify research priorities to meet future needs of the evolving management in the Gulf of Maine. Of particular interest were developments since the previous RARGOM Symposium held in 1996 (Wallace and Braasch 1997).

The Gulf of Maine is arguably one of the best-studied and most highly managed marine ecosystems in the world, and as such is an interesting case study in the implementation of an ecosystem approach. The symposium objectives were (1) to examine what progress had been made over the past decade in implementing an ecosystem approach, (2) to evaluate how well positioned we are to implement an ecosystem approach in the Gulf of Maine, and (3) to propose relevant research priorities. The symposium involved over 200 participants, from approximately 75 institutions (mainly in Atlantic Canada and New England), and approximately 100 presentations (Cooper et al. 2010).

The geographical focus of the symposium was the Gulf of Maine watershed, delineated by the eastern tip of Massachusetts in the southwest and Cape Sable at the southern tip of Nova Scotia in the northeast, and including both Massachusetts Bay and the Bay of Fundy (Fig. 1).

Based on the recommendations of a large and diverse scientific steering committee, the symposium included invited and contributed papers and posters, and was structured around four themes: tools for integrated policy and management; structure and function of the Gulf of Maine system; anthropogenic and external influences in the Gulf of Maine ecosystem; and monitoring/observation, data collection, analyses, and tools required for an ecosystem approach in the Gulf of Maine. There were also concurrent technical sessions featuring research contributions from both natural and social scientists, who sought to connect the relevance of their research to marine resource managers and policy and decision-makers in the following areas:

- Ecosystem Services in the Gulf of Maine
- Biodiversity in the Gulf of Maine
- Seafloor Mapping for Ecosystem Management in the Gulf of Maine

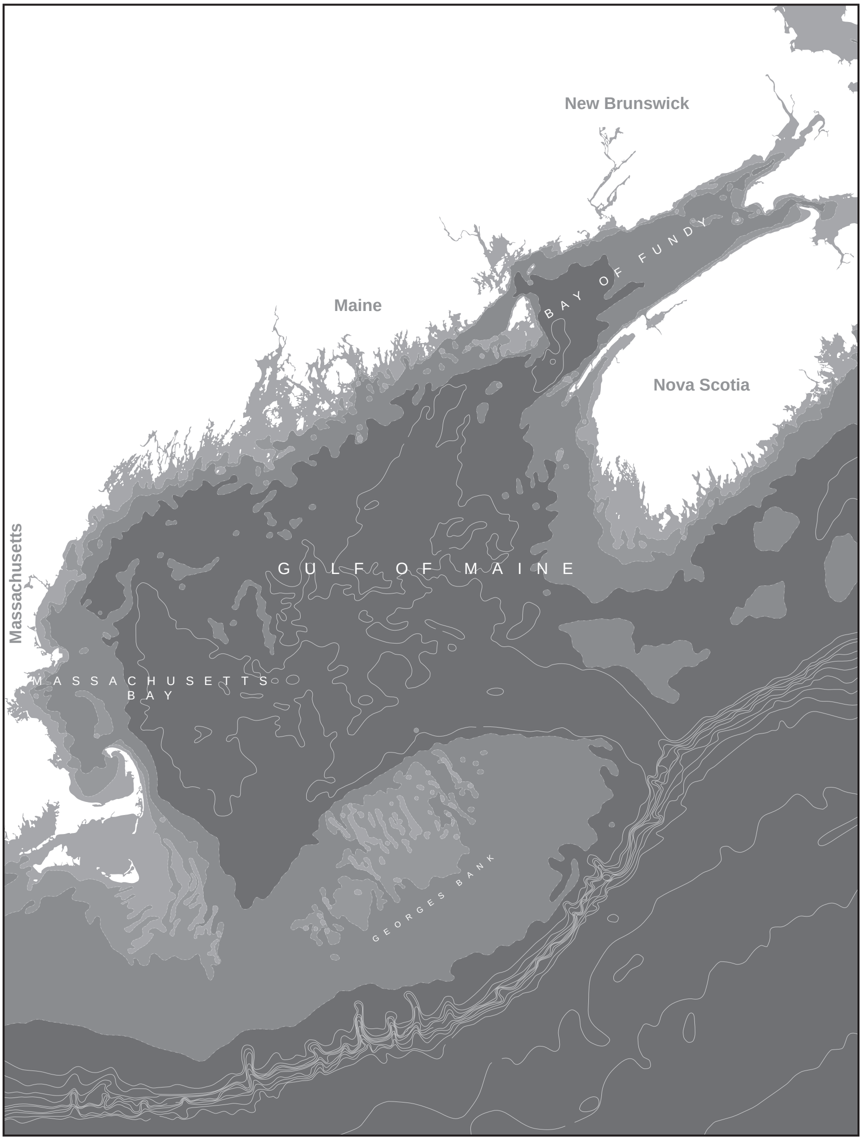

Figure 1. Gulf of Maine.

- Life Histories of Gulf of Maine Fishes and Invertebrates

This paper highlights the scope and major conclusions of the symposium. A more thorough description of the program (including abstracts and rapporteur reports has been presented by Cooper et al. (2010), and 28 contributions from the symposium are being published by the American Fisheries Society in a dedicated, peer reviewed volume (Stephenson et al. 2012).

Changing policy context for the Gulf of Maine

Among the major reasons for convening the symposium were the significant advancements in integrated policy and management in the decade since the previous RARGOM Symposium, and an improved understanding of the evolving landscape of management (both in the Gulf of Maine and internationally) that was bringing scientists and policy makers together around ecosystem-based management. It is now widely appreciated that a more holistic "ecosystem-based" or "integrated" management approach is needed to account for the full spectrum of human impacts in the marine environment and the implications for the ecosystem services these systems provide. The ecosystem approach to management (EAM), or ecosystem-based management (EBM), embodies several key attributes: (1) it is place-based and entails the development of integrated management plans for defined ecological regions; (2) it considers humans as integral components of the ecosystem; and (3) it requires an understanding of the interrelationships among the components of the system and the environment. Adoption of ecosystem-based management strategies for the Gulf of Maine will require the implementation of regulatory and legislative frameworks to allow the full spectrum of management considerations that will emerge, including confronting trade-offs among and within different ocean use sectors. It will further require the development of appropriate governance structures and close cooperation between the United States and Canada.

While not yet fully articulated or operational, these approaches will inevitably include more diverse objectives with respect to the impacts of fishing and other activities on the ecosystem as well increased awareness of the impact of the ecosystem (including ecosystem change) on management, and will demand integration of conservation (including maintenance of productivity at the organism and trophic levels, conservation of biodiversity of populations and communities, and protection of habitat) as well as social, economic, and institutional aspects of management.

There have been changes in legislation and in policy frameworks in both the U.S. and Canada in support of an ecosystem approach to management. In Canada, the Oceans Act (proclaimed in 1997) and efforts to renew the Fisheries Act have led to the creation of new policies and strategies to manage our oceans, including Canada's Oceans Strategy (2002) and the Sustainable Fisheries Framework (2009). In the U.S., the 1996 Sustainable Fisheries Act amended the Magnuson-Stevens Act with a requirement to identify essential fish habitat and to make progress toward reducing, avoiding, or mitigating both fishing and non-fishing impacts to these habitats. More recently, the U.S. National Oceans Policy (July 2010) established nine objectives including marine spatial planning. Parallel efforts in the private sector include development of eco-certification initiatives in fisheries and development of ISO standards in business.

While there is a better understanding of the overall context of oceans management and legislation and policy to help implement an ecosystem approach to management, the current operating environment still faces a number of challenges. The Gulf of Maine is administratively complex, as evident in the diverse composition of the Gulf of Maine Council on the Marine Environment (http://www.gulfofmaine.org/new-site/). Oceans management on both sides of the Canada-U.S. border is complex with multiple jurisdictions (national and states/provinces) responsible for managing fisheries and other aspects of the marine environment in the Gulf of Maine. This complexity has been compounded by the use of the court system to address fisheries management issues, particularly in the U.S. There has been a proliferation of organizations and networks focused on the Gulf of Maine marine environment. The number of organizations with an interest in the Gulf of Maine demonstrates the importance of the region to a wide range of stakeholders, including politicians and decision-makers, students and researchers, resource users, and those living in coastal communities, while at the same time posing communication and governance challenges. While the diverse interests bring together the necessary diversity of perspective to discussions, it also makes timely integration of policy, management, and science difficult. There is no legislative/administrative framework in which diverse participants can work together effectively in implementing a common vision of the ecosystem approach.

Advances in knowledge for an ecosystem approach in the Gulf of Maine

The ecosystem approach is generally considered to demand more and different information. The Gulf of Maine Symposium provided new insights with respect to advances in understanding structure and function of the Gulf of Maine ecosystem, including ecosystem change, monitoring and data observation, ecology and biodiversity, seafloor mapping, ecosystem services, anthropogenic influences, and tools for integrated policy and management.

Ecosystem change

The Gulf of Maine has undergone dramatic changes in fundamental aspects of its structure that can be tracked on multi-decadal to centennial time scales. The prospect of future climate change in the Gulf of Maine highlights the need to understand its current status and past changes that can be related to anthropogenic and natural forcing factors. In complex systems, such as the Gulf of Maine, the possibility of rapid change to alternate stable states must be anticipated. It is essential to continue emphasis on synthesis and integration of the rich body of research in the Gulf, and the development of models that can be used to predict the effects of changing environmental conditions, and the implications of alternative management actions. Important strides have been made, particularly in development of coupled biophysical models. Linkage of numerical hydrodynamic models to a broader array of ecological models will be necessary to place the modeling efforts in service to management. There is need for approaches that link our hydrodynamic models to general circulation models at the basin scale to evaluate the potential impacts of climate change on the Gulf.

Monitoring and observation systems

Over the past 15 years, considerable steps have been taken to organize observation and to document change in the region. The Canadian Atlantic Zonal Monitoring Program began in 1998, supporting the Halifax line and a fixed station on the Scotian Shelf off Halifax and in the Bay of Fundy, providing sustained funding for observation of physical and biological variables at the upstream boundary of the Gulf of Maine. The Gulf of Maine Ocean Observing System (GoMOOS) was established in the late 1990s, resulting in the installation of a series of observational moorings along the Maine and Massachusetts coasts for weather and hydrographic measurements and limited phytoplankton sampling. In 2009, the transition from GoMOOS to NERACOOS (Northeastern Regional Association of Coastal Observing Systems) was made after several years of planning by a multidisciplinary advisory committee of regional experts. The geographic range of NERACOOS includes not only the Gulf of Maine but also the southern New England Bight and Long Island Sound. This regional infrastructure has advanced the capacity for coordinated observation of change, although the extent of sustained funding and range of variables to be observed is not yet established.

In addition to long time series of annual and biannual fish trawl surveys conducted by both NMFS and DFO, NMFS has sustained the Continuous Plankton Recorder line between Cape Sable and Boston, as well as the EcoMon surveys that are conducted six times each year. Satellite-derived sea surface chlorophyll time series began in 1997, and presently two satellites (SeaWifs and MODIS) cover the Gulf of Maine each day. The duration of these time series is beginning to yield information on interannual and interdecadal variability and trends.

New tools for integrating and analyzing observational data have been developed. Notable are great advances in coupled physical-biological modeling, supported in large part by the U.S. GLOBEC Northwest Atlantic/Georges Bank program. Trophic models of energy and material flux across the ecosystem food web and time series and multivariate statistical approaches have also advanced significantly. These approaches have been facilitated by the tremendous increase in computer capacity over the past decade.

In terms of ecosystem approach to management, both Canadian and U.S. federal agencies responsible for management of harvested resources have been working toward a strategy for integrating observations into information for management decisions. The U.S. strategy involves development of integrated ecosystem assessments within a framework of evaluating drivers (e.g., large scale climate forcing), pressures (e.g., temperature increase), ecosystem states (from observations and indicators), assessment of impacts (using ecosystem models), and determination of responses. The Canadian strategy is more incremental, building on existing fisheries management practices within a framework of defining objectives (e.g., maintain productivity, preserve biodiversity, protect habitat), determining strategies and specifying tactics to achieve objectives.

The establishment of AZMP in Canada and NERACOOS in the U.S., combined with ongoing resource surveys by federal and state agencies and research-industry partnerships, provide a foundation for providing information to management about change in the Gulf of Maine. Knowledge of past and present change is substantially improved. New developments in ecosystem modeling and multivariate statistical and time series analysis offer tools to integrate and interpret multidisciplinary data. Different scientific approaches using different types of ecosystem modeling to assess change and inform management are being developed, involving considerable thought and effort.

There is a need for collection of time series observations across ecosystem levels, particularly in the nearshore and coastal regions, and for the benthos. The science needs to continue along many fronts to analyze and interpret data and to develop predictive, integrative models of climate change scenarios on the Gulf of Maine ecosystem and its resource populations. Bridges to transfer new research knowledge, understanding, and information support tools from science to management implementation need to be developed and maintained. The NOAA cooperative institute (CINAR) in the Gulf of Maine is an example. There is a need for better integration of the full range of available information into management of fisheries and other activities, and connections to coastal managers need to be fostered.

Ecology and life history

There has been considerable recent progress in the development of new analytical tools and the implementation of new research and monitoring programs with respect to ecology and biodiversity in the Gulf of Maine. Advances in genomics have made possible the identification of previously underrepresented microbial components. The now–routine use of satellite observations for estimation of chlorophyll concentration on fine spatial and temporal scales has revolutionized our ability to document critical ecosystem processes related to bloom dynamics and overall levels of productivity. Advanced in situ sampling tools ranging from gliders to coastal observatories have provided important adjuncts to traditional sampling devices in the Gulf. Isotopic signatures have been examined in investigations ranging from identifying water mass characteristics to diet composition and trophodynamics. Advances in high-end computing resources have opened important avenues for the development of coupled physical-biological models with data assimilation capabilities.

Numerical models of coupled physical-biological systems have now been developed and implemented to explore seasonal phytoplankton bloom dynamics in relation to changing salinity characteristics, to set the stage for development of operational forecasts of red tide blooms, and to understand the role of high frequency internal waves on patchiness of phytoplankton in the Gulf of Maine. Biophysical models also hold considerable promise for understanding dispersal pathways and recruitment processes for meroplanktonic organisms. Collectively, these initiatives open the way for predictive modeling capabilities for application in fisheries and environmental management.

There continues to be significant research on fisheries. Most commercial species, notably the two iconic species Atlantic cod and the American lobster that have shaped the history of the region and determined the character of local fishing communities, have changed dramatically in abundance. While cod populations in the Gulf have undergone long term declines, lobster populations and catches have increased markedly over the last several decades. The symposium emphasized the importance of basic life history traits, including growth, mortality, migration, maturation, fecundity, spawning, and recruitment, which are the drivers of population dynamics. These traits are key inputs into stock assessment models, and therefore play a pivotal role in our development of an ecosystem approach and management strategies. Fisheries stock assessments have become increasingly sophisticated and complex as analytical tools have developed and data sources have grown, and will become even more so as we continue to move toward EAM. Understanding the underlying demographic processes that shape fish stocks and their role in the ecosystem will be as, if not more, important in this new paradigm. It is critical that attention to basic biology not be lost. Knowledge of basic biology, particularly related to the key species in the ecosystem, are the basic building blocks for single species and for multispecies/ecosystem approaches. For some species/fisheries we do not have the information, and for others we have only basic parameters (such as growth).

Ecosystem attributes and process have clear but complex effects on life histories. It is clear that there are environmental impacts on growth and recruitment, yet the relationships have not, in general, been defined. Both single species assessments and the ecosystem approach are expected to evolve to consider temporal and spatial complexity. It will be important, in the context of EAM to consider how attributes at the organism level scale up through individuals to drive evolutionary, population, and ecosystem dynamics. There is need for greater attention to the needs, motivations, behaviors, and other complexities of the human component of the ecosystem. At present, human needs and behaviors are often incorporated simply as fishing mortality rates applied and revenue earned. However, we are a species at least as complex as those for which we are uncovering variation and change in life histories, and our complexities need to be considered as well. Enhanced basic sampling and biological study should be a high priority.

Biodiversity in the Gulf of Maine

The Gulf of Maine Area (GoMA) program was the regional ecosystem project of the international Census of Marine Life, a ten-year initiative (2000-2010) that set out to describe the biodiversity of life in the oceans: past, present, and future. GoMA's overarching goals were to increase understanding of biodiversity in the Gulf of Maine area, describe how this biodiversity supports regional ecosystem functioning, and suggest ways in which biodiversity information can be used to support decision-making regarding the marine environment. As part of its synthesis phase, GoMA convened six expert groups organized around a combination of trophic/community types and habitats: coastal margins; benthos and demersal nekton; slope and seamount environments; microbial communities; zooplankton and pelagic nekton; and upper trophic level predators. In addition to evaluating current understanding of diversity, structure, and function within specific ecosystem "compartments," each expert group was asked to identify promising new lines of scientific enquiry and new technologies (emerging or needed), and to identify how biodiversity knowledge in each compartment can contribute to EAM.

The Gulf of Maine is an extensively studied system with a long history of ecological research conducted by an impressive concentration of research facilities distributed along the coast. The biodiversity of mid and upper trophic levels in particular is well known and the ecological roles of these species have been intensively investigated. As a result of the Census of Marine Life initiative, more than 50,000 new viral and bacterial operational taxonomic units have now been recognized. A species register for the Gulf of Maine is under development and will provide an invaluable reference source for overall levels of biodiversity in the Gulf. Important new insights also have been gleaned from studies conducted on a broad spectrum of spatial and temporal scales. These include a new Census of Marine Life Discovery Corridor initiative encompassing a broad swath from the intertidal, through deep ocean basins, to the edge of the continental shelf and beyond. In-depth studies documenting local biodiversity hotspots in the Gulf have also been undertaken. This work has been nicely complemented by studies of nearshore biogeographic patterns and biodiversity of benthos in relation to physical characteristics. It also has been possible to examine evidence of temporal changes in fish and macroinvertebrate biodiversity through a careful comparison of samples collected in two research programs separated by a century. Further, sustained monitoring of plankton communities in the Gulf, using both continuous plankton recorder and standardized Bongo sampling (MARMAP-EcoMon) Programs, show changes in plankton species composition and biodiversity on decadal time scales.

Several recurring themes and recommendations emerged during the symposium:

- The closer we look, the more we see. Even in areas that have received a lot of attention, new sampling is revealing new species or range extensions (e.g., in Jordan Basin). This means that EAM, which includes the conservation of biodiversity at functional levels, must employ approaches that are robust to the lack of specific knowledge about all species, their distributions and dynamics, or their ecosystem roles.

- Data mining is still providing new insights, including new species and range extensions. This highlights the importance of historical data that have been underutilized—often because they have not been very available. Digital databases are changing this, and making an important contribution to future management.

- The Gulf of Maine Register of Marine Species remains a work in progress. Adding species to the register involves a specific set of comparisons against original taxonomic authorities, and other regional and global registers. Careful comparisons of the existing register against the databases being compiled by several expert groups will provide for some prioritization for ongoing work to make the register as comprehensive and useful as possible.

- Although the Gulf of Maine is an exceptionally well-studied system (compared to many other regional ecosystems) many of the research and monitoring programs have been sporadic. There are actually few standardized and regionally comprehensive (in time and space) monitoring programs, and only some databases are available to the wider research community. This demonstrates the need for strategic Gulf-wide and nested, smaller-scale biological observing programs that include plans for standardization and sharing of data.

- Although historical sampling may not be thorough, i.e., we can't determine the full range of species that were present, we can get a sense of what species were historically abundant, and gain insights into shifting baselines over time.
- While each of these expert groups is examining biodiversity within a specific "compartment," there is a need for end-to-end coupling and integration of the knowledge across trophic levels as well as spatial domains. This gets to be a very large problem and will require new modeling strategies that reduce the problem to tractable levels without losing credibility.

Conserving biodiversity, one of the central goals of EAM, is challenging because the majority of marine biodiversity is still unknown and will remain so for the foreseeable future. Most species are comparatively rare, and the "importance" (function) of rare species is difficult to quantify in the present and impossible to predict for the future. We are still in the discovery phase for biodiversity characterization and mapping. We need to promote and recognize contributions to ocean biogeographic databases and build a comprehensive regional ecoinformatics framework. The latter requires exchange of technical approaches, and building and publishing scientific workflows (from standardized data acquisition techniques, through analytical routines for generation of biodiversity metrics).

Seafloor mapping

The Gulf of Maine ecosystem contains a diverse array of geological and biological substrates that serve as habitat for commercially and ecologically important mammals, seabirds, fish, and invertebrates. Seafloor mapping is a relatively new technology that already has been used as an effective tool in assessing ecosystem impacts of human activities and defining essential fish habitat of economically important species, and it is expected to feature more prominently in marine spatial planning as part of an ecosystem approach.

Over the past decade, the Gulf of Maine Mapping Initiative (GOMMI) has organized forums and coordinated efforts to advance mapping of critical seabed habitat characteristics in highly utilized regions of the Gulf of Maine. In April 2009, GOMMI and the Gulf of Maine Research

Institute convened a workshop on how to integrate seabed information into fisheries management decisions more effectively.

The mapping workshop at the Gulf of Maine Symposium brought together geologists, benthic ecologists, fisheries scientists, and coastal managers from Canada and the United States to explore how seabed information is currently being used in ecosystem management, and how this information can be made more useful to those who manage fisheries, energy development, and other human activities in coastal and offshore waters of the Gulf of Maine. There was agreement that ecosystem-based management will require improved information about the habitats of economically and ecologically important species and the impacts of different human activities.

There is, at present, limited seabed substrate information in the Gulf of Maine, especially in the U.S. (see the Gulf of Maine Mapping Initiative's website for a map showing regional coverage: http://www.gulfofmaine.org/gommi/coverage-map.php). This limits the ability of managers to use the parameter in ecosystem management activities that require more holistic coverage of this bioregion. The field of acoustic seafloor mapping is relatively young, and data acquisition technologies and analytical approaches are evolving rapidly. The development of methods that enhance our ability to discriminate between benthic habitats will consequently improve our ability to manage the valuable resources associated with these habitats.

Seafloor information is often the foundation upon which marine spatial planning is based, so that attempts to manage and sustain marine resources and ecosystems will require accurate seafloor information.

Better geophysical seabed information certainly will enhance efforts to manage the Gulf of Maine ecosystem and associated resources; yet in many cases this information only provides the basis for further investigations. For example, attempts to link seafloor substrate to criteria of essential fish habitat requires coupling seabed information with additional investigations of how these substrates influence fish life history parameters such as survival, growth, and ultimately productivity.

Participants of the workshop debated the utility of striving for one overarching map or relying on a mapping strategy that uses multiple complementary maps. Several of the presentations demonstrated the utility of this latter approach to achieve ecosystem management goals.

Ecosystem services

"Sustaining the long-term capacity of systems to deliver ecosystem services is the core goal of ecosystem-based management (EBM) for the oceans" (McLeod and Leslie 2009).

Ecosystems and biodiversity are essential for human well-being. "Ecosystem services" are the benefits humans derive from ecosystems—the things we need and care about that we get from Nature. This approach makes the importance of a healthy environment more obvious and relevant to politicians, economists, business people, and the public. It is hoped this will motivate conservation and sustainability. Often, ecosystem services—other than goods provided that we can easily put a dollar value on—are not factored into important decisions that affect ecosystems and are not included in cost-benefit analyses. This damages the ability of nature to provide services, making human society and the environment poorer.

The Gulf of Maine provides humans many ecosystem services. Seafood, recreation, and aesthetics are highly visible to the public but there are numerous other services that need to be recognized. The Millennium Ecosystem Assessment in 2005 identified four categories based on function: (1) Provisioning services such as food, fresh water, fuel, and timber; (2) Regulating services such as the regulation of climate, floods, disease, water, and air quality; (3) Cultural services such as recreational or aesthetic enhancement; and (4) Supporting services such as upwelling, nutrient cycling, and primary production. The symposium gathered researchers focused on evaluating ecosystem services from Canada and the United States to share methods, techniques, and experiences.

Participants identified further research needed in the Gulf of Maine to transform the way we account for the type, quality, and magnitude of nature's goods and services so they can be considered in management decisions. They discussed the needs for data, methods, and models to better understand and communicate the benefits of considering ecosystem services. They also gauged interest in forming a partnership to identify, map, quantify, and evaluate ecosystem services in the Gulf of Maine including estuaries and coastal wetlands.

There are currently many gaps in identifying, quantifying, and valuing ecosystem services that would benefit from more research and more policy/governance work. Ecosystem services may be evaluated in a Driver-Pressure-State-Impact-Response (DPSIR) model. There is a need to specify a manageable number of indicators that could be used to quantify potential management scenario outcomes, and then work together to develop the metrics that would inform decisions. This approach is necessarily interdisciplinary, and may include economic and other social objectives.

Outstanding research questions include:

- More complete understanding of **all** ecosystem services and how they can be valued.
- How do the links within and among socio-ecological systems influence the delivery of marine ecosystem services?

- How can emerging science be more effectively connected with management and policy processes, particularly in terms of trade-off analyses?
- How do we measure success of this new approach?

Anthropogenic and external influences on the Gulf of Maine ecosystem

As noted in previous Gulf of Maine conferences, (dating back at least 20 years) there have been major anthropogenic influences both from activities in the marine environment (including fisheries and aquaculture) and from drainage basin colonization (including agriculture and urban waste disposal).

Climate change, rarely mentioned or investigated around the Gulf in earlier years, has emerged in the past decade as a high priority problem. The debate is no longer about whether climate change is occurring, but whether we can slow the change and/or adapt. Communities and governments are undertaking pilot projects to evaluate adaptation options. Increasing storm surges, sea level rise, more violent storms, and changes in precipitation are likely to have significant impacts on coastal habitats and societal infrastructure, especially in low-lying areas, e.g., floodplains, estuaries. In addition to encouraging community adaptive responses, governments are interested in the potential to slow change by reducing carbon and greenhouse gas emissions.

Perhaps the most significant change in our perspective on the Gulf of Maine is the recognition that ecosystem management, to be operationally feasible and effective, must consider intertwined ecological and social factors. During the past decade, researchers have focused on biophysical and socioeconomic impacts of activities in Gulf of Maine, the utility and value of ecosystem services, and insights gained from hazard and risk assessment and monitoring. The symposium pointed to the continuing challenges to ecosystem health of climate change, pollutants, and overuse of natural resources, but also featured signs of resilience, recovery, and adaptation pertinent to management strategies.

Looking ahead, there is need for conceptual models and an operational framework for managing the Gulf of Maine and Bay of Fundy in the context of unpredictable ecosystem change. Scientific research, long-term monitoring programs, more frequent state of environment assessments, interdisciplinary and interagency collaboration, and communication are crucial.

Communication and joint action among stakeholders is essential for progress in achieving the sustainability of the ecosystems of the Gulf of Maine. Sharing knowledge of the history of the region, of the booms and busts in fisheries, and of changes in habitat, between fishermen and scientists may lead to innovative approaches to fisheries management. In the past decade, steps toward successful collaboration and communication between scientists and fishermen have been taken through the cooperative research projects sponsored by the Northeast Consortium, Gulf of Maine Research Institute, NOAA Fisheries, and the Canadian Fisherman and Scientists Society. In increasing numbers of cases, conversations among fishermen and scientists involved in collaborative work have been mutually beneficial, but the challenge of quantifying qualitative data for use in models remains elusive. Public education, outreach, and communication among the other stakeholders for the Gulf of Maine, and Bay of Fundy, are also vital.

Emerging tools for integrated policy and management

The symposium explored the management and policy tools and approaches that are required to implement integrated management in the Gulf of Maine. Tools for integrated policy and management meet a wide range of functions and originate from a variety of fields, including the physical and biological sciences, the social sciences, and management/organizational science. Tools presented at the symposium included those used for organizing and evaluating information, tools for assessing the implementation of management objectives, and tools for communication, such as adult learning models and new media technology.

New scientific tools to support ecosystem approaches to management include the methods of identifying essential fish habitat to improve management of these designated areas in the U.S., and the classification of inlets along the Atlantic coast of Nova Scotia using a geographic information system (GIS). These approaches could form the basis for management activities of conservation planning including marine spatial planning. There are a number of tools to better organize, evaluate, and share existing information with the goal of implementing ecosystem-based management. The Ecosystem Based Management Tools Network launched an EBM Roadmap in the spring of 2009 for EBM practitioners. The Maritimes Region of Fisheries and Oceans Canada is developing an Ecosystem Approach to Management (EAM) framework that will assist in developing specific management strategies to respond to EAM objectives. State of the environment reporting for the Gulf of Maine is being developed through the Gulf of Maine Council on the Marine Environment. The goal is to help decision-makers better understand the main issues affecting the Gulf, and thus assist in designing appropriate management responses. The COINAtlantic initiative aims to provide one-stop access to online data for those responsible for implementing integrated management.

In designing ecosystem-based management programs in the Gulf of Maine there will be the challenge of accounting for the diversity of views and potential contributions of a wider constituency involved with integrated management. The application of developmental psychology and adult learning models to oceans management and the emphasis on exploring novel means of communicating are new considerations when employing an integrated management approach. Time spent in building capacity and understanding among those involved with ecosystem-based management may be, at the very least, as important as developing the technical tools to implement it. The potential of new technologies working together; to enable greater collaboration, communication, and knowledge sharing; and to permit management to be more open and transparent is acknowledged.

Better tools for evaluating and managing cumulative effects are still needed. The Gulf of Maine is already heavily subscribed, and new activities, such as tidal and wind power, have been proposed. Collaborative efforts to develop tools and approaches for addressing cumulative effects will be needed in order to fully implement the ecosystem-based management approach.

While it is recognized that much progress has been made toward integrated policy and management, an overall evaluation of that progress is difficult. The application of state of the environment reporting, frameworks such as the ecosystem approach to management, or approaches developed by the EBM Tools Network may assist in measuring and evaluating progress toward ecosystem-based management.

Improved communication among those involved in research, education, and policy in the Gulf of Maine is still required. Raising the general levels of ocean literacy is also desirable. Communicating research results should be a significant component of the research planning cycle and financial resources for this aspect should be built into all research programs. Furthermore, it was recommended that researchers engage those they think will use or benefit from their research at the earliest possible stage. The integration of decision-makers, scientists, and others involved in “integrated policy and management” in developing tools and approaches will aid in making them practical, pragmatic, and, above all, applicable to ecosystem-based management in the Gulf of Maine.

The symposium provided further insights into questions related to the appropriate spatial scales for management. It is clear that distinctive differences in physical and ecological characteristics exist in the eastern and western Gulf of Maine. It is further evident that for a number of reasons, the nearshore Gulf of Maine may require special consideration. The diversity of human activities on the coast and in the immediate coastal zone requires consideration of cumulative impacts of fishing, pollution, and habitat alteration/destruction. The nearshore region is also substantially affected by watershed influences that affect productivity patterns and other characteristics. Collectively, these suggest that nested structures for management may need to be recognized within the Gulf to account for spatial differences.

With respect to understanding interrelationships among parts of the system and with the environment, again it is clear that we have much to build on. There is important evidence of bottom-up control in the system with effects throughout the food web. Further, results presented at the session show that the system has undergone regime shifts related to climate and physical forcing affecting nutrient dynamics with attendant consequences for primary production, zooplankton community composition, and fish community structure. These considerations will necessarily play an important role in devising management strategies in an ecosystem context. In particular for ecosystem-based fishery management, understanding shifts in productivity states will be essential in devising sustainable exploitation strategies at the ecosystem level that account for shifting environmental states. We also have a rich database on diet composition of higher trophic levels that are essential for building and refining multispecies and ecosystem models for management. In all of this, it must be remembered that the Gulf of Maine was subject to important alteration long before detailed scientific studies were undertaken. These include the dramatic reduction of whale populations, the decimation of anadromous fish stocks due to obstruction of rivers, habitat loss and overfishing, and overfishing of once-dominant species such as halibut. We must be aware of the implications of these changes for overall system productivity.

Finally, it must be recognized that we are still in the very early stages of understanding how best to integrate the human dimension into EBM in the Gulf. We need to understand how humans have and will impact the system and how changes in the Gulf of Maine affect human communities.

Discussion

The Gulf of Maine is a semi-enclosed continental shelf sea with distinctive characteristics relative to adjacent regions such as Georges Bank and the Scotian Shelf. Reasonable arguments can accordingly be made for the Gulf of Maine proper as a spatial unit for EBM. It is clear that the rich history of research in the Gulf of Maine provides a strong foundation for moving towards ecosystem-based management in this region.

The previous major symposium for science in the Gulf of Maine in 1996 focused on ecosystem modeling (noting that there was “no overarching conceptual model”) with the hope of “enhancing scientific endeavors” and “improving management decisions” (Wallace and Braasch 1997). Interestingly, there was little if any mention in that meeting of climate change, or of the need to implement an ecosystem approach. In the following decade there was considerable development in the management landscape of the Gulf of Maine. The ecosystem approach to management emerged as a major topic of research interest internationally, and there were changes in legislation (in both Canada and the U.S.), so that federal, provincial, and state jurisdictions in Canada and the U.S. moved forward with "ecosystem approaches" to managing marine resources. It was considered timely to review and update the policy approaches and science, and make recommendations on the knowledge required to move forward with an integrated management approach/ecosystem approach to management (IM/EAM) in the Gulf of Maine.

The 2009 Gulf of Maine Symposium documented several developments toward EAM in the Gulf of Maine over the past decade. Notably there has been explicit recognition of the importance of climate change and the need for adaptation. Increased observation, resulting in part from new technologies, has resulted in new time series that will show environmental shifts. Improved legislation (in both Canada and U.S.) demands more holistic approaches and multiple uses, and there has been development of the concept and framework for development of the ecosystem-based approach, including consideration of essential habitat and the creation of tools for marine spatial planning and state of the environment reporting. There has been great progress in high-resolution seafloor mapping and the development of approaches to characterize habitat in mapping. The Census of Marine Life program has increased knowledge of patterns of biodiversity in the Gulf of Maine. Progress continues on coupled biophysical models and advanced models of the physical system (e.g., FVCOM) leading to better understanding and modeling of circulation and to recognition that flows are changing. Knowledge of the life histories and biology of relevant species continues to increase (although the knowledge base differs widely among species), and there is a greater appreciation of aspects related to conservation objectives such as the importance of population subunits (ecotypes), variability of growth and productivity, and species richness/biodiversity. There has been progress in networking related to people and data/informatics (including ocean data partnership, data mining, rapid assessment). There is now explicit recognition that EAM includes ecological and social/economic considerations and that it demands an interdisciplinary approach. There is also recognition of the importance of different spatial scales. There has been research related to novel approaches to energy generation (e.g., in-stream tidal power), and better understanding of the lessons of past management and long-term effects of exploitation. We have seen progress on understanding of pollutants (from listing to active monitoring) and increased knowledge of nutrient dynamics.

One of the major objectives of the symposium was to consider how ready we are to implement EAM. There is a range of approaches to the ecosystem approach at present. In some cases current management plans are being modified to be more consistent with EAM (evolutionary) whereas others propose a more revolutionary approach.

It is clear from the symposium that we now have enhanced understanding of the context for EAM. There is awareness of the multiple biophysical and socioeconomic impacts, of ecosystem services, and of climate change. There has been considerable development in ecosystem models, There is a better foundation for conservation planning arising from Census of Marine Life and other initiatives. There have been insights from hazard and risk assessment and from management of pollutants, and of invasive species. We have learned much about resilience and adaptation from fisheries and environmental histories and there has been recent improvement in appreciation of growth and population complexity. There is improved ocean planning and evolving participatory processes, including better information that can form the basis for siting of projects, resolving conflicts, and for future "marine spatial planning."

However, there is a challenge in implementing the ecosystem approach in the Gulf of Maine. This is in part because of the complexity in jurisdiction and increasingly litigious environment. There remains the problem of ecosystem complexity including "the scope of the unknown and ubiquity of the rare." There is an issue of need for enhanced monitoring and information to support evolving the management landscape. There is a need for institutional (governance) change to support cross-disciplinary and inter-jurisdictional considerations. In general there is a feeling among participants that we know the general direction in which we need to go, and although there is some urgency to take action, we lack the governance structures and leadership to allow progress to full implementation of an ecosystem approach.

The discussion of future priorities with respect to implementation of EAM fell into two categories—needs of science and needs of institutions.

There is urgent need for appropriate governance structures and improved institutional capacity for implementation of an ecosystem approach, including:

- A common vision of goals and objectives
- Interdisciplinary participation
- Legislative basis and development of appropriate governance structures
- Consistency among jurisdictions
- Participatory structures (engagement of users)

- More comprehensive approach to ocean use planning
- Enhanced collaboration in evaluations/assessments

In addition, there is need for increased basic understanding related to environmental and governance change, including:

- Evaluation of models of climate (is the Gulf of Maine warming?) and flow (are flows changing?)
- Enhanced understanding of coastal processes and life histories in relation to environmental change
- Knowledge of benthic, and especially microbial processes
- Strengthened science and monitoring of aspects of relevance to EAM decisions in management
- Strong link among those with science, management, social, and legal expertise
- Institutional "bridges" to link information and research to management
- Requirement for greater general ocean literacy
- Clarification of terminology, and consistency of use
- Enhanced understanding of cumulative impacts
- Metrics of progress and of success (how will we know when we are there?)
- Continued mapping for marine spatial planning

It is clear that success in implementing an ecosystem approach for the Gulf of Maine requires agreement on what the ecosystem approach really is, and the integration of currently disparate elements within a framework that links management, science, and users in a more holistic approach. The need for the holistic approach was described in three summaries as follows:

- Development of an operating framework for managing the Gulf of Maine using an ecosystem-based approach and in the face of ecosystem change.
- Development of a comprehensive, interdisciplinary approach to management and a framework for the evaluation of management (management strategy evaluation).

- Development of an organized approach to adaptation throughout the region (perhaps through the Gulf of Maine Council on the Marine Environment).

The greatest need appears to be the development of consensus around the Gulf of Maine, on a practical framework for implementing the ecosystem approach, for an integrated approach to the management of multiple human activities in relation to a more diverse set of objectives and a changing environment, that includes a higher standard of ecological integrity and diverse aspects of sustainability.

Progress continues in the development of New England Fishery Management Council/NOAA and DFO Maritimes approaches to EAM and in development of coordinated attempts at "state of the environment" reporting. There are also pilot initiatives such as the southwest New Brunswick Marine Resource Planning Initiative. There continues to be collaboration among scientists in RARGOM, and among the relevant governance parties in the Gulf of Maine Council on the Marine Environment. We propose that there is need for some leadership by the Gulf of Maine Council on the Marine Environment (or another body working on behalf of various administrations), in definition of the practical implementation of EAM.

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Ecosystem-Based Management of Fish Species in the Barents Sea

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Abstract

This review paper gives an overview of modeling for management of species in the Barents Sea ecosystem with special emphasis on capelin, and points to ways forward for increased focus on ecosystem processes in fisheries models for this area. The capelin fishery in the Barents Sea, managed by Russia and Norway through the Joint Norwegian-Russian Fisheries Commission, is one of only a few fisheries in which multispecies interactions are taken explicitly into consideration when quotas are set. Given its important role as a forage species, the capelin stock is managed using a target escapement strategy. Capelin spawning stock size is forecasted 6 months ahead of time based on survey estimates of capelin and Atlantic cod (northeast arctic stock) and historical estimates of cod consumption of capelin to estimate capelin natural mortality. This model was implemented in stock assessment and fisheries management advice in 1991, and since then the catches have varied between 0 and 1.1 million t. Proposed further developments of the capelin assessment model include additional predation on capelin by harp seals, main zooplankton groups as prey of capelin, and modification of capelin recruitment as affected by the biomass of young stages of the Norwegian spring spawning herring, a major predator on capelin larvae. An integrated assessment of capelin and cod is also being considered, in which capelin biomass is used to model cod growth. Although a full ecosystem-based fishery management approach has not been implemented for Barents Sea fisheries, inclusion of key trophic interactions is a significant step in this direction.

Introduction

Background on geographical setting and fishery catches

The Barents Sea (Fig. 1) is a high-latitude, shallow, and productive shelf sea of 1.6 million km^2, which provides a major source of harvestable

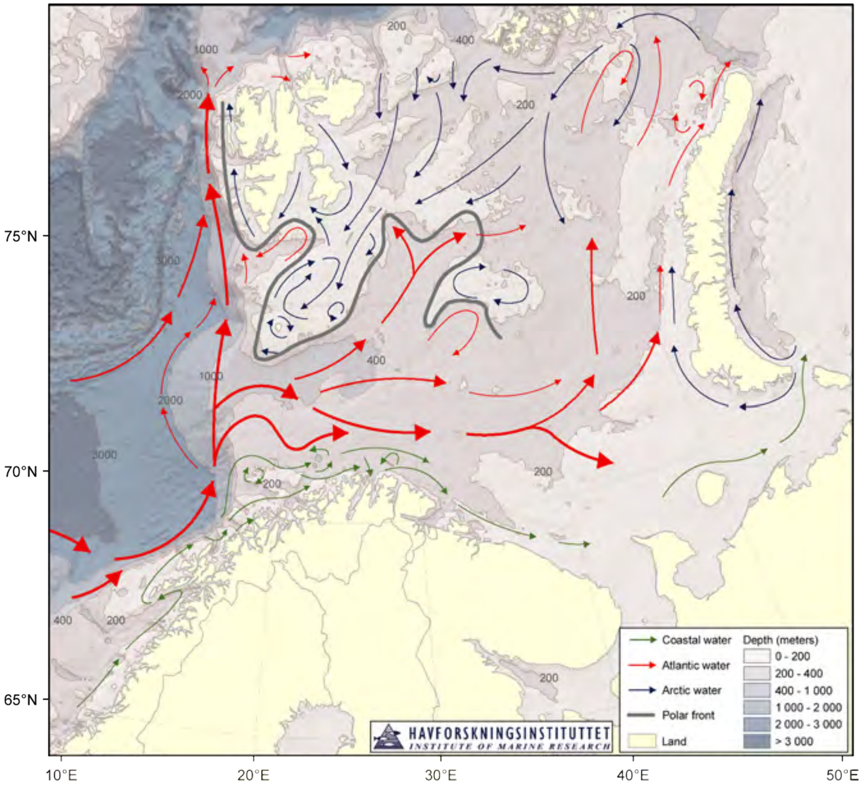

Figure 1. Barents Sea and adjacent waters.

fish resources. It is bordered by the northern Norwegian and Russian coasts to the south and Novaya Zemlya Island to the east. The 500 m depth contour is often used to delimit the Barents Sea toward the deeper Norwegian Sea in the west and the polar basin to the north. Relatively warm, saline Atlantic water flows into the Barents Sea from the southwest, whereas the northeastern part is cold with low salinity and covered with ice during winter.

Major commercial fish species inhabiting the area are Atlantic cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), Greenland halibut (*Reinhardtius hippoglossoides*), two species of redfish (*Sebastes* spp.), and capelin (*Mallotus villosus*). Deepwater prawn (*Pandalus borealis*), red king crab (*Paralithodes camtschaticus*), and some minor fish species such as wolffishes (*Anarhichas* spp.) and polar cod (*Boreogadus saida*) are also harvested in the Barents Sea. In Norwegian coastal areas, saithe

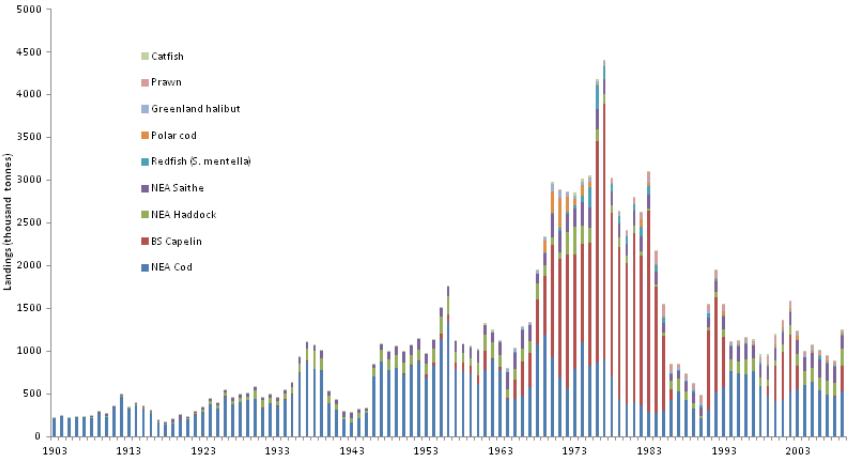

Figure 2. Landings of various fish species (thousand metric tons) from the Barents Sea during 1903-2009.

(*Pollachius virens*) is also important. Norwegian spring spawning herring (*Clupea harengus*) stay in the area as young, but normally leave the area before attaining fishable size. In some periods, blue whiting (*Micromestitius poutassou*) are found in the western parts of the area. Small numbers of harp seals (*Pagophilus groenlandicus*) and minke whales (*Balaenoptera acutorostrata*) are also hunted in this area. The average annual landings from this area have been about 1.1 million t of fish over the last 15 years (Fig. 2). The landings of gadoids were about 950,000 t in 2010 and are increasing. The second largest landings are associated with capelin, which have fluctuated from 0 to 3 million t during the last 40 years (Fig. 3).

Prior to the mid 1970s when few regulations were in force, the fishery outside the territorial waters was open to fishers from any country. Regulations were the results of multilateral negotiations under the auspices of the Northeast Atlantic Fisheries Commission (NEAFC) (Hønneland 2004). Exclusive economic zones (EEZs) of 200 nautical miles from the coast were established, following the international process leading to the United Nations Convention on the Law of the Seas (UNCLOS). Thus, the 200-mile zone around Svalbard, called a “fisheries protection zone,” is a non-discriminating zone under Norway’s jurisdiction. The establishment of the EEZs resulted in a small area of international waters in the middle of the Barents Sea, referred to as the “loophole.” The process leading up to formation of the EEZs

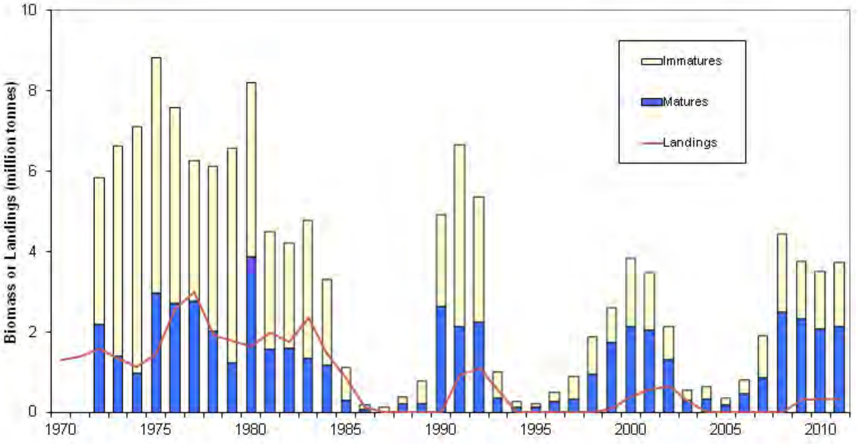

Figure 3. History of Barents Sea capelin stock biomass and fishery landings.

also triggered the formation of the Joint Norwegian-Russian Fisheries Commission in 1976 (Hønneland 2006), which thenceforth has carried out management of the fish stocks in the Barents Sea. Management includes various measures such as total allowable catches for cod, haddock, capelin, Greenland halibut, and harp seal (allotted to Norway, Russia (USSR), and some additional countries) and technical regulations as minimum mesh size in trawls, minimum landing size, closed seasons and areas, prohibition of discarding, etc.

While the initial regulatory measures had single-species perspectives, the focus of the management has gradually moved from single-species toward multispecies, and finally toward the ecosystem. This has been a difficult process and still, 25 years after the first multispecies model intended for fisheries management in the Barents Sea, the main perspective in the fisheries management in the Barents Sea is a conventional single-species perspective. However, some progress has been made and some of the ambitions toward a more holistic management approach have been fulfilled. In the present essay, we describe the management of the living resources in the Barents Sea and the development that has taken place historically and in recent years. The purpose of this paper is to provide a detailed review of Barents Sea capelin as a case study incorporating some ecosystem approaches in stock assessment and fishery management.

Historical development of management and modeling in the Barents Sea

The development of fisheries management in the Barents Sea is closely linked to the development of the cooperative research between Russia and Norway as basis for management (Haug et al. 2009). The background to the collaboration was the negative development of the stocks and fisheries of northeast arctic cod and Norwegian spring-spawning herring in the 1950s, a period marked by increasing exploitation. Both countries are longstanding members of the International Council for the Exploration of the Seas (ICES), which provides advice on the management of fish stocks in the North Atlantic, but felt the need for a closer cooperation. At that time ICES did not consider advice on the total quantities that could be harvested from the stocks as necessary, and the advice only dealt with various types of technical regulations. The need to limit the total landings gradually evolved, and from the late 1970s the main commercial stocks in the area were regulated by total allowable catches (TACs).

Fisheries biology in Russia traditionally focused on ecological issues pertaining to stock interactions and biological processes, while in Norway the focus was more on issues of population dynamics and quantitative aspects such as effects of harvest on single stocks. For example, since the 1930s the diet of cod has been studied at the Polar Institute for Oceanography and Fisheries (PINRO), Murmansk (Zatsepin and Petrova 1939, Dolgov et al. 2011 and references therein). Owing to collaboration between Norwegian and Russian research institutions these interests gradually merged. In 1984, a joint program of stomach content analysis was initiated between the Institute of Marine Research (IMR) in Bergen and PINRO in Murmansk. This program is still ongoing and today more than 315,000 fish stomachs have been analyzed quantitatively and entered into the joint database. Mainly cod stomachs, but also stomachs from haddock, Greenland halibut, saithe, and other piscivorous fish, as well as from capelin, polar cod, and herring, have been analyzed (Dolgov et al. 2007). Alongside the sampling and analysis of stomach content, experimental work to establish gastric evacuation rates and other parameters was undertaken to be able to calculate consumption, and models for such calculations were developed (Temming and Andersen 1994, dos Santos and Jobling 1995, Bogstad and Mehl 1997).

Some drastic events in the Barents Sea during 1984-1988 changed the view among managers, fishers, and marine scientists on fisheries management. The inadequacies of the traditional single-species models became obvious. The capelin stock suddenly collapsed, and during the years that followed, the cod stock was also drastically reduced in size, caused by reduced growth and increased mortality (including cannibalism). More than 100,000 harp seals invaded the coastal areas and drowned in fisher's nets, and breeding colonies of seabirds were abandoned or drastically reduced in size (Markussen and Øritsland 1991, Hamre 1994, Gjøsæter 1995, Barrett et al. 1997). Gradually, the fishing industry, managers, and scientists showed more interest in multispecies models. It became clear that not only did multispecies interactions play a major role in the drastic upheavals experienced in the Barents Sea, but also that, if population dynamics models were to be realistic, such interactions had to be taken into consideration. Several projects were launched to build such multispecies models, e.g., MULTSPEC (Bogstad et al. 1997). An overview of multispecies models for the Barents Sea is given in Bogstad and Filin (2011).

These first-generation multispecies models were quite complicated and at least two obstacles inhibited their application. First, more data were needed than were available for model parameterization. Second, it was not clear how multispecies models could be applied in practical management. The solution was not obvious because, although multispecies models could provide a set of catch options (quotas for each species), these models did not take socioeconomics into account. For instance, is it preferable to harvest 5 kg of capelin and 1 kg of cod or to abandon the capelin fishery altogether and increase the yield of cod from 1 to 2 kg? Such decisions require economic analysis of the fishing industry and hard decisions about who is allowed to fish in the future. These decisions are made more complicated by the fact that several nations fish in the same area.

Since the mid-1990s, priority has been given to development of less complex models with fewer biological processes. IMR developed the Bifrost model, which is now used in the management of capelin (Gjøsæter et al. 2002) and for exploring harvest strategies for cod, capelin, and herring (Tjelmeland 2005). SeaStar, the assessment model used until 2008 for assessment of Norwegian spring-spawning herring (Tjelmeland and Lindstrøm 2005), allows for including predation by minke whales on herring. A Gadget model (Begley and Howell 2004) has also been set up for the Barents Sea (Lindstrøm et al. 2009, Howell and Bogstad 2010). Gadget is a successor of MULTSPEC and another early multispecies model, Bormicon (Stefánsson and Pálsson 1998). At PINRO, Stocobar (Filin 2005) has been developed. These models fall into the category of "minimum realistic models" (Plagányi 2007), where only those aspects of the ecosystem considered essential for the management task are addressed quantitatively.

This movement toward simpler models might be viewed as a step backward. Although complicated multispecies models had their use, they were not tools for calculating quotas. They were useful in exploring various scenarios for future stock development. Complicated models could, however, be a source of realistic parameters, which are input to single-species models. This realization has opened up development of a new generation of assessment models that incorporate species interactions and environmental effects. This is where the management of the Barents Sea fisheries stands today—the TACs are set for single stocks, but for a few of them, information on other stocks and the environment are taken into account. The most important commercial stocks are now managed with harvest control rules (HCRs).

In addition to the attempts to take into account ecosystem or environmental factors in TAC calculations of single stocks, the movement from ad hoc quota settings to the use of HCRs is the most important development in fisheries management in the Barents Sea. The HCRs are based on management objectives in accordance with the precautionary approach to fisheries management. For some stocks the rules also include a constraint on annual variation in TAC. The next step is to amend the HCRs and formally base them on the maximum sustainable yield (MSY) concept, while keeping them precautionary. In principle, and for the capelin stock in practice, the multispecies models can be used to explore the performance of these rules in long-term simulations.

Capelin stock assessment and management—a case study

The movement from pure single-species assessment came first, and has been carried furthest, for the Barents Sea capelin stock. This is not by chance; as mentioned above the collapse of the Barents Sea capelin stock in 1984-1986 was the event that first demonstrated the need for broadening the assessment from single-species to multispecies. This fact has been underlined by two additional capelin stock collapses since then, collapses that are considered to be natural fluctuations caused by recruitment failures in successive years triggered by increased larval predation from big year classes of herring entering the Barents Sea (Hamre 1988, Gjøsæter 1995, Gjøsæter and Bogstad 1998, Hallfredsson and Pedersen 2009, Hjermann et al. 2010). These strong fluctuations (Fig. 3) in one of the key elements in the ecosystem have had profound effects to the whole ecosystem, but the different collapses affected the ecosystem quite differently (Gjøsæter et al. 2009).

Historical overview of capelin stock assessment

This description of the capelin stock assessment is based on Gjøsæter et al. (2002), where more detail may be found. Fishing for capelin has been going on in the Barents Sea since the beginning of the twentieth century, but catches were low until the early 1960s. When the stock of Norwegian spring spawning herring collapsed, much fishing effort shifted to the Barents Sea capelin and a major purse seine fishery on this “new” resource was developed (Gjøsæter 1995). The stock has been monitored by annual acoustic surveys since 1972. The management goals in this early period were rather ad hoc. Only in two years, 1974 and 1978, was a national catch quota introduced for the Norwegian fishery.

In 1978, the USSR-Norwegian Fishery Commission asked scientists from the two countries to evaluate the state of the stock and to submit proposals for necessary joint management actions. Two meetings of scientists were held in 1978 (Anon. 1978a,b), resulting in the following agreement: a total allowable catch (TAC) advice for capelin should be based on acoustic stock measurements carried out jointly in autumn, the quota period should cover the winter and subsequent autumn fishery, and the aim should be to preserve a minimum stock of spawners of 500,000 t. The strategy for a minimum spawning stock of 500,000 t was based on a rough evaluation of the relationship between spawning stock biomass (SSB) and recruitment; for spawning stocks below that level, the risk of poor recruitment seemingly increased. During this period, the model Capelin was developed by Tjelmeland (1985) as an aid to carrying out the assessment outlined above.

Because capelin die after spawning, and because there are no measurements between the September estimate and the time of spawning, the management of capelin depends on modeling the development of the maturing stock from October 1 to the time of spawning, a projection period of 6 months. One of the most demanding tasks was to split the estimated stock into a maturing and an immature component in autumn, which was done by assuming maturation to be length-dependent. Also, the natural mortality of the maturing component during winter was difficult to estimate, since it cannot be done from stock size estimates alone. Since consumption by cod is the most important source of natural mortality, the management of capelin thus becomes a multispecies problem. Capelin management using an escapement strategy is impossible without modeling cod predation on capelin. The Capelin model comprised a recruitment sub-model, which allowed for long-term simulations where MSY and other relevant quantities were calculated. Using this model, Hamre and Tjelmeland (1982) analyzed the yield functions for various fishing patterns and allocations of total catch for the autumn and winter fishery. They introduced the new concept "*M*-output biomass," denoting the production of capelin available to predators as natural mortality, *M*. One of their conclusions was that MSY of capelin would be reached with a SSB of about 400,000 t. Another observation was that fishing during autumn would maximize the yield but lower the *M*-output biomass considerably. In a multispecies context, winter fishing would, therefore, be preferable to autumn fishing since more capelin would be left in the sea as food for other species (e.g., cod). Based on those analyses, they recommended that a minimum spawning stock level was kept at 500,000 t, and that the catches should be taken during winter. For the period 1986-1990, a fishing ban was recommended because, even in the absence of fishing, the spawning stock size was estimated to be below the 500,000 t limit. The fishery was allowed in 1986 although no catch was advised. Management advice has been followed closely in all years thereafter, including zero-catch advice (except for small quantities of research catch).

In an ecosystem-based approach to fisheries management one should take account of the impact the fisheries has on the ecosystem. Since management strategies for cod influence long-term average biomass levels of cod, models of cod predation on capelin biomass are a first step toward including the effects of the cod fishery on the capelin stock.

After the stock collapse of 1985-1989, it was realized that the assessment model used previously was inadequate. First, the manner in which *M* of mature capelin during winter was estimated was simplistic (Tjelmeland and Bogstad 1993). Previous knowledge (e.g., Zatsepin and Petrova 1939, Ponomarenko and Ponomarenko 1975) indicated that cod exerted tremendous predation pressure on capelin. Already in 1975, the ICES working group dealing with cod noted that "if the capelin stock were to decline, it is not known if the surplus food thus made available would be used by organisms which could serve as food items for the cod stock" (ICES 1975). In 1978 it was suggested that varying predation pressure from cod would influence the fishable stock of capelin (Anon. 1978b). In 1984, a joint Norwegian-Russian cod stomach sampling program was initiated (Mehl and Yaragina 1992), which soon confirmed that capelin constituted a considerable proportion of the food consumed by cod during winter.

During this period work was initiated to build a multispecies model (MULTSPEC), including the main stocks of fish and marine mammals in the Barents Sea (see described earlier). This model was, however, rather complicated and data-demanding and it was never used in its full version as a tool in stock assessment for quota-regulation purposes. An attempt was made to utilize the data in the stomach-content database to estimate the quantity of capelin consumed by cod (Bogstad and Gjøsæter 1994), and to use this as an estimate of the *M* of maturing capelin during winter. This was a first attempt to include quantitatively the influence of other species in the assessment of capelin. Based on a combination of the capelin model and some ad hoc methods connected with the calculation of assumed capelin consumption by cod, recommendations of non-zero TACs were advised during the period 1991-1993. It was, however, realized that the large stock size at that time was mainly based on a single year-class (1989). From 1992 on, a new period of recruitment failures was apparent, the stock dwindled, and again a fishing ban was recommended and introduced (Gjøsæter 1998).

The development of assessment methods taking into consideration the influence of other fish stocks continued during and after the second capelin stock collapse. This included continued work on the inclusion of the influence of cod on capelin mortality during winter (Bogstad and Gjøsæter 2001). However, because the recruitment failure of capelin, resulting in the two stock collapses, was partly attributable to the stock of young herring occurring periodically in the Barents Sea, Gjøsæter and Bogstad (1998) pointed out that the effect of herring should also be taken into account. During this period, the single-species model, Capelin, was abandoned in favor of a model in the MULTSPEC family of models: the Bifrost model (see above). This model is also a multispecies model but has no geographical resolution. For TAC calculations, it is combined with the spreadsheet model CapTool implemented in the @RISK add-on to MS Excel. The parameters of the model are estimated in Bifrost, and this model is also used to construct replicate parameter files for stochastic future development of the stocks. These replicate files are fed into CapTool, which, for given catch quotas, gives probability functions for capelin stock development and a risk of the spawning stock falling below a certain size.

This pair of models has been used to calculate capelin TACs since 1998. With them, a probabilistic assessment is introduced. The precautionary approach is implemented based on stochastic simulations where the proportion of 6-month simulations with SSBs below B_{lim} is calculated. Then, a catch level that gives the desired probability of SSBs below B_{lim} (e.g., 5%) is determined. Since the capelin SSB only has a one-time value, no explicit B_{pa} (or trigger point for reducing F in a harvest control rule) is relevant. Also, a rule for reducing fishing when B is less than B_{pa} (implicitly defined as the median of the distribution of SSBs when the 5% quantile is B_{lim} = 200,000 t) is not relevant because after the present rule the advised catch is zero in such a case. Albeit based on multispecies considerations, the assessment is still of a single-species nature, because neither the effect of fishing herring (which affects capelin recruitment) nor the effect that fishing capelin exerts on cod growth are quantified and taken into consideration. According to the classification by Plagányi (2007), this is an “extended single-species assessment model.”

Currently within ICES, the stocks are assessed ideally every year according to a written protocol (“stock annex”), containing input data and methods to apply. Periodically, the stocks are put to a “benchmark assessment,” where the experts normally involved in the assessment, as well as other assessment experts within the ICES community, scrutinize the assessment to consider whether new (and better) methods are available or new data series should be brought into the assessment. This process and the resulting written stock annex are reviewed and evaluated by assessment experts outside the ICES community. In August 2009 the Barents Sea capelin assessment, together with the assessments of some other short-lived species, were handled by the benchmark assessment group WKSHORT (ICES 2009). The outcome of the external reviewers' evaluation demonstrated some of the problems involved when stepwise enhancement of an assessment has been going on for some years: the documentation of changes made may lag behind in the process. The experts behind the capelin stock assessment were criticized by the external reviewers for not having an updated stock annex. However, they were commended for the way in which the Barents Sea capelin assessment has incorporated predator-prey interactions; the reviewers even suggested that "this is world-leading in development of an ecosystem approach."

The inclusion of predator-prey interactions in the assessment partly accounts for *M* and has the added advantage of providing knowledge about the year-to-year variations in *M*. It is not obvious, however, how the remaining potential *M* should be treated. What fraction of the total *M* is addressed by predators? The capelin assessment we have so far includes only predation by immature cod during January-March. In the present situation, with a large mature cod stock and a high overlap between capelin and cod (immature and mature) in autumn, it seems appropriate to take into account cod predation on capelin in autumn and by mature cod during spring (ICES 2011). We have so far chosen to ignore the residual *M* during the months January-March, when the cod predation is at its maximum, because we think that the capelin mortality other than cod predation is negligible. Whether this is true remains to be demonstrated, and inclusion of other predators, such as harp seals, is on the list of tasks in the capelin assessment.

Reestablishing MSY-based management for Barents Sea capelin

The present HCR for capelin should be tested against the MSY approach. This has been done for cod (Kovalev and Bogstad 2005). This test showed that the adopted HCR for cod, which is in accordance with the precautionary approach, also is within the range of HCRs that are associated with high long-term yield. Such testing for capelin will be left for the near future. Here, it is shown how an MSY-based target escapement SSB for capelin could be established, where both the cod-capelin interaction already implemented in management and the significance of herring for capelin recruitment are accounted for.

The replicate parameters in CapTool that underlie present (as of 2010) management of capelin were estimated in 2003. In connection with the establishment of a new stock annex for capelin (given in ICES 2011) the 2003 version of Bifrost has been modified slightly and updated with new data (see www.assessment.imr.no). A large number of possible recruitment models have been estimated using herring, cannibalism, predation from 0-group cod, and various models for temperature (all based on the Kola section observations) as possible covariates, multiplicative to the Beverton-Holt recruitment model. Thus, the recruits tend to zero as SSB tends to zero. The recruits were taken as one-year-old capelin, back-calculated from the measured number of two-year-old capelin, the catch statistics, and an annual estimate of natural mortality of immature capelin. The observed number of one-year-old capelin was not used, as the measurements in earlier periods have proven to be unreliable. Herring and temperature were invariably picked out as significant covariates. During prognostic simulations used to evaluate possible HCRs, the recruitment relation to use for each trajectory is drawn at random using Akaike weights (Burnham and Anderson 2002), similar to what was done when investigating HCRs for Norwegian spring spawning herring (Røttingen and Tjelmeland 2009). The amount of herring in the Barents Sea during prognostic simulations is based on the assessment of herring using data after 1991, when the stock had been fully rebuilt, and where reasonable autocorrelations have been preserved by repeatedly cycling through the historic time series. The cod stock is kept constant at the present level. For both cod and herring, a scaling factor may be applied. Fig. 4 shows the mean long-term yield as a function of SSB_{target} for different assumptions about future cod and herring stocks.

The SSB corresponding to the MSY (SSB_{MSY}) is estimated at somewhat lower levels than by Hamre and Tjelmeland (1982), who used a strictly single-species model, but also considerably lower than that of Røttingen and Tjelmeland (2009), who used a more elaborate multispecies model incorporating population dynamics of herring and cod, and also including consumption by harp seals. The main point, however, is that in the present model SSB_{MSY} is estimated at about the same value for a wide range of different assumptions on future stock sizes of cod and herring giving widely different expected yields. So, our recipe for management action remains the same even if we may be uncertain about future biological driving forces for the capelin.

The present analyses are illustrative of how one could arrive at an escapement SSB based on MSY-based reference for capelin, where assessment uncertainty, model uncertainty, and uncertainty concerning future distribution of driving factors can be accounted for. In accordance with present ICES recommendations an MSY-based management should not replace a management based on the precautionary approach (PA), but should be complementary. Thus, incorporated into the present methodology used for capelin, one could select a catch quota that ensured that the realized SSB is above B_{lim} with 95% probability and above the MSY-based reference point with 50% probability. Recent history indicates that the PA could be limiting.

Using PA-based HCRs has been criticized because there is normally not enough information to model the tail of the distributions with suf-

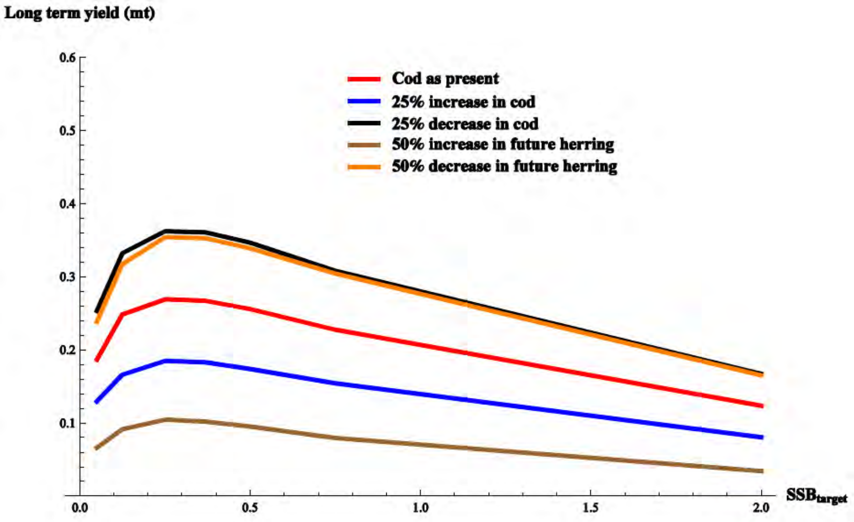

Figure 4. Mean long-term yield of Barents Sea capelin as a function of SSB_{target} for different assumptions regarding cod and herring stock size. Yield and SSB in million metric tons.

ficient accuracy (Rochet and Rice 2009, Kraak et al. 2010) during simulation testing. Complementing the PA with an MSY-based approach based on 50% probability could induce more stability in the process.

Even though using HCR mechanically in the management process is being criticized (e.g., Kraak et al. 2010) this has created stability and a meaningful reduction of fishing pressure in the management of Barents Sea demersal species. Without agreed upon HCRs the quota setting process is open for annual negotiations between Russia and Norway, most often to the detriment of long-term sound management. To both scientists and managers, the HCR has become the common and well-known interface. New knowledge and improved assessment technology can be incorporated into the HCRs without altering the form of the scientific presentation of results to managers, as long as the researchers from both countries agree. When the use of HCRs in management is discussed in recent literature the power of the HCR to forge cooperation between different stakeholders is often overlooked. Butterworth et al. (2010) give examples of the strength of testing and implementing formal HCRs in promoting co-management.

The management of capelin is often heavily debated in Norway. While parts of the industry would want to have a fishery more often

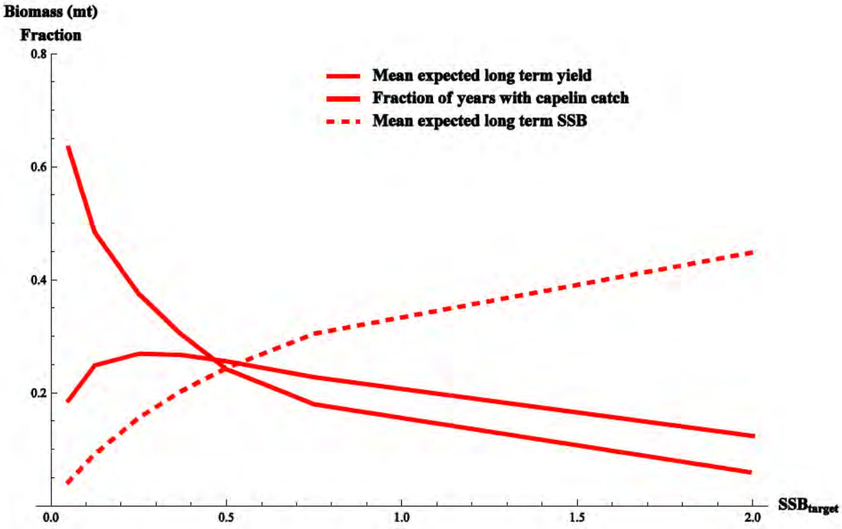

Figure 5. Mean long-term yield, mean long-term SSB, and fraction of years with capelin fishery as function of SSB_{target}. Yield and SSB in million metric tons.

than is allowed by the present HCR, other parts would want the capelin fishery to be reduced to let as much capelin as possible spawn and die, thereby serving as additional food for other fish species and as fertilizer for the coastal ecosystem. More scientific information could be brought to this discussion than is available today, for instance by routinely presenting the mean expected SSB (potential fertilizer) and the mean expected fraction of years with fishery; an example is shown in Fig. 5. Even in the absence of using endogenously dynamic cod and herring in the simulations as did Røttingen and Tjelmeland (2009), we have shown here how HCRs involving predation by cod and herring on capelin can be tested by simulation, thus setting the agenda for a multispecies-based HCR for Barents Sea capelin.

Incorporation of species interactions in assessment of species other than capelin

The incorporation of predator-prey interactions is not restricted to the capelin assessment. Since 1995, cannibalism by large cod on small cod and predation by large cod on small haddock have been included in the assessments of cod and haddock. In these cases the same Russian-

Norwegian stomach content database is used as in the case of capelin. In the cod assessment, adult cod is entered as an additional fleet exploiting young cod, and the traditional sequential population analysis is iterated until the predation on young cod equals the outcome of the consumption estimate based on the stomach content database and the consumption model (ICES 2011). In the haddock assessment the predation by cod is entered as an additional source of natural mortality. Cod cannibalism has also been taken into account in studies of MSY for cod (Kovalev and Bogstad 2005).

It has been found that the consumption of capelin by cod has varied from 229,000 to 3,209,000 t, consumption of haddock by cod from 3,000 to 359,000 t, and cod cannibalism from 8,000 to 536,000 t. The variation is a function of both predator and prey abundance.

Other species interactions that could be implemented in assessment models and that have been implemented in Bifrost for experimental use (Røttingen and Tjelmeland 2009) include the following:

- *Effect of capelin abundance on cod growth.* Capelin abundance may influence cod growth considerably in some periods (Mehl and Sunnanå 1991), even though during the two latest capelin collapses cod has been able to switch to other preys, thus limiting the effect of capelin collapse on cod growth (Gjøsæter et al. 2009). When modeling cod growth as a function of food supply, it is thus important to include both capelin and other important prey items.

- *Effect of capelin on cod recruitment/cannibalism.* Cod cannibalism seems to be inversely related to capelin abundance (Yaragina et al. 2009, ICES 2011). Thus capelin abundance could be included in short-term predictions of cod recruitment.

- *Effect of size of capelin spawning stock on the ecosystem.* Spawning and post-spawning (dying or dead) capelin, as well as capelin eggs, may have a considerable value for benthic predators (haddock and many others). Thus, a large spawning stock of capelin may be beneficial to the ecosystem. In a fisheries management context, however, the first step could be to relate the growth of haddock to capelin spawning stock size.

It is important, though, to always keep the model(s) inside the minimum realistic realm, which puts a limit as to how far one can go in including ecosystem elements.

Conclusion

The Barents Sea capelin gives an example of how we can implement the ecosystem approach to management by incorporating ecosystem components into the assessment itself, thus elucidating ecosystem considerations in a setting familiar to managers. It goes without saying that this approach is quite restricted because only the most data-rich, commercially important species can be embraced. However, considerations around these species are of most interest to various parts of the industry and the general public. For other ecosystem considerations the parallel process of using ecosystem indicators must be used.

Moving from single-species to multispecies management is a daunting task, not only biologically and mathematically, but also because it involves socioeconomic and political components, which are more demanding than single-species management. We think that the work carried out during recent decades show that, in areas with strong and clear-cut interactions and good knowledge of the ecosystem (e.g., Barents and Baltic seas), it is possible to move stepwise toward multispecies management. This should be done using a combination of extended single-species models (where main interactions are included in a single-species assessment model) and larger multispecies models, which aim at quantifying the interactions in more detail.

The modeling described here focuses on the development of multispecies models for use in tactical decisions, of which there are few examples worldwide (FAO 2008). This is done by assuring that, at each development stage, the models are kept minimally realistic. This implies that the models are simple enough for the parameters to be estimated in a meaningful way on the basis of the data while being comprehensive enough to encapsulate the description of essential population dynamics. According to FAO's best practice for ecosystem models (FAO 2008), such models should be tested using more widely scoped ecosystem models in a process termed "management strategy evaluation." Although a management strategy evaluation is yet to be achieved for the models described in this manuscript, it remains an important future objective.

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Co-Management of Reef Fisheries of the Snapper-Grouper Complex in a Human Ecological Context in Brazil

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Abstract

Most commercially important reef fish are late maturing species and have slow growth, being thus vulnerable to overfishing. In Brazil, coastal artisanal fishers usually depend upon reef fishes, such as groupers (Serranidae) and snappers (Lutjanidae), because these fish attain higher market values compared to other fishing resources. Most of these local and small-scale coastal fishers are located in high biodiversity areas, such as the Atlantic forest coastal remnants. Therefore, the management of artisanal reef fisheries faces a dilemma regarding the economic and ecological contexts: to conserve fish stocks (avoid their depletion) or to catch fish to fulfill immediate economic needs. Drawing from examples of different fisheries from northeastern to southern Brazilian coastal sites, we described this dilemma and provided management suggestions, aiming to allow the development of sustainable artisanal fisheries. We recorded data from 1,761 fish landings, sampled and collected 1,453 fish from these landings, and interviewed 585 fishers from 14 fishing communities on the northeastern, southeastern, and southern Brazilian coast. The main reef fish caught by the studied fishers were snappers (12 species) and groupers (16 species), the former more common on the northeastern coast and the second more common on the southern coast. However, some of these reef fishes showed many individuals caught below the size at first maturity, which may adversely affect the exploited fish stocks. In order to change this trend, we suggested that these reef fishes would be best managed through co-management processes, which involve the participation of fishers on where and when to fish. Managing reef fish stocks could thus gain the support of local fishers, if they contribute to the monitoring of reef fishes located close to or in their fishing spots, preferably through economic incentives.

Introduction

Reef fisheries are challenging to manage due to their complexity (varied fishing gear among other aspects); their local scale, which makes it difficult to devise general management measures; and problems related to conciliating biodiversity conservation with social needs (McClanahan et al. 1997, 2009; Pauly 2006). A compounding problem is that these fisheries usually target large, highly valued reef fish, such as groupers (Serranidae) and snappers (Lutjanidae), which are vulnerable to recruitment overfishing due to their large size, slow growth, and late reproductive maturity (Sadovy 2001, Saenz-Arroyo et al. 2005).

Reef fishes are important targets of artisanal fisheries in Latin America, especially along the tropical northeastern Brazilian coast, because they are caught by simple fishing gear (hook and line) and can attain a high market price (Costa et al. 2003; Teixeira and Ferreira 2004; Fredou et al. 2006, 2009; Francini-Filho and Moura 2008). A high diversity of fish species is caught in Brazilian artisanal fisheries (Begossi and Figueiredo 1995), but reef fishes are usually considered “noble” fish, which may induce fishers to deploy more effort to catch them (Nehrer and Begossi 2000).

Marine catches in Brazil accounted for 485,000 tons in 2003 (Haimovici et al. 2006). Artisanal fisheries account for as much as 50% of Brazilian total fish production, but this amount varies according to the region. For example, on the northeastern Brazilian coast, artisanal fisheries contribute 88% of the total fish production, while on the southeastern coast artisanal fisheries account for 34% (Vasconcellos et al. 2007). Northeast Brazil contributes about 12% of the national fish production, led by Bahia State (40% of northeastern production). Artisanal fisheries in this region use sailboats (74%) and operate in depths to 250 meters, especially due to the narrow continental shelf, which allows artisanal fishers to fish in deeper areas. Hooks and lines are often used, but the use of longlines and traps has also been reported (Lessa 2006). Fredou et al. (2006) report that between 1996 and 2000, reef fisheries in northeast Brazil caught 187 fish species; lutjanids (snappers) represent 40% of the catch, including five species (*Ocyurus chrysurus*, *Lutjanus synagris, L. analis, L. jocu,* and *L. vivanus*). Vasconcellos et al. (2007) note the importance of snappers and their current overexploited status in northeastern Brazilian fisheries. Two species (*L. vivanus* and *L. jocu*) are described as overexploited to their maximum limits and two other species (*O. chrysurus* and *Romboplites aurorubens*) are described as overexploited (Vasconcellos et al. 2007).

In southeast Brazil, artisanal fisheries encompass areas located along the shore, often shallower than in northeast. Southeastern fishers use motorized and paddle canoes, and often fish with hook and lines or set gillnets (Begossi and Figueiredo 1995). Groupers (Serranidae), especially dusky grouper (*Epinephelus marginatus*) and comb grouper (*Mycteroperca acutirostris*), are also important reef fish targets that may suffer overexploitation (Begossi and Silvano 2008, Begossi 2010).

Among the several problems associated with managing artisanal fisheries in Brazil, two deserve special attention: the lack of data about fish catches and the social complexity of fishers, who are usually rural indigenous inhabitants (Begossi 2010). The lack of fish production data can be attributed to the absence of systematic sampling. For example, we lack data on what, where, or how many fish are caught, and also temporal data on fish landings (Freire et al. 2007).

The local ecological knowledge of fishers (LEK) has been a useful source of information on the biology and ecology of marine resources, as shown by many studies done in tropical, temperate, and arctic oceans (Johannes 1981, Huntington 2000, Ruddle 2000, Berkes 2008). In Brazil, research on artisanal fisheries has shown the importance of fishers' LEK for management, for training fishers to collaborate in research, and for validating ecological information (Silvano et al. 2006, Begossi 2008, Silvano and Valbo-Jorgensen 2008, Gerhardinger et al. 2009).

The objectives of this study were to use data collected between 2002 and 2009 at 14 artisanal fishing communities in northeast, southeast, and south Brazil to (a) evaluate the main fish species caught, including data on their diet, reproduction, and size; (b) analyze the results in a human ecological context, by showing the relevance of fishers' LEK (Lopes and Begossi 2009); (c) assess the usefulness of fishers' LEK to obtain short-term data and to foster future collaborative efforts between researchers and locals; and finally (d) offer some suggestions on how to improve current artisanal reef fisheries management in Brazil.

Materials and methods

The study sites included seven fishing communities in northeast Brazil (Mucuripe, Ponta Negra, Riacho Doce, Porto do Sauípe, Itacimirim, Arembepe, and the Tento neighborhood in the city of Valença); six fishing communities in southeast Brazil (Gamboa, Jaguanum, Copacabana, Itaipu, Búzios Island, and Bertioga), and one fishing community in south Brazil (Pântano do Sul). The locations of these communities along the Brazilian coast are in Table 1 and Fig. 1. Further information on these sites can be found in previous surveys (Begossi 2006, Silvano et al. 2006, Begossi and Silvano 2008, Silvano and Begossi 2010).

The 14 study sites were visited on different fieldwork trips made during distinct research projects and two major periods: 1986-2002 (Gamboa, Jaguanum, and Búzios Island); and 2002-2009 (the other 11 communities, Table 1). Data collection included interviews with fishers, monitoring of fish landings, fish collection for identification, and biological analyses of fish stomach contents and presence of mature gonads (visible eggs) (biological sampling was performed in 2007-2009 only). The biological analyses were performed on site, in fish stores. For the interviews, we obtained verbal consent from the fishers and then followed a protocol based on an open-ended questionnaire. Interviews were performed with artisanal fishers in each community, usually at landing points or at the fishers' houses. Most interviews focused on ethnoecology and ethnobiology and were performed with fishers older than 40 years who had lived and fished at the site for more than 25 years (groupers) or more than 10 years (snappers) (Silvano et al. 2006, Begossi and Silvano 2008). Fish landings were sampled monthly to obtain the number and weight of fish species caught, fishing site, and gear used. Fish identifications were made using taxonomic keys (Menezes and Figueiredo 1980, Figueiredo and Menezes 1980) and revised by J.L. Figueiredo and R. Caires from the Zoological Museum of the University of São Paulo (MZUSP). In Copacabana in 2008-2009, methods included a visit, twice a week, to the "Colônia de Pescadores do Posto 6" (where there is a local fish market for artisanal fishers), in order to collect snappers. Stomach contents were analyzed qualitatively, crabs were identified using a crustacean key (Melo 1996), and identifications were reviewed by G.S. Melo (MZUSP). Gonads were measured by weight and volume (in ml) and a macroscopic analysis of eggs (visible or not visible) was performed. The choice for macroscopic analysis of gonads was made because histological analyses are costly (Sadovy 1996), and we decided to apply a method that would take into account the scarce laboratory options in Brazil, as well as the possibility to develop a method that fishers could use themselves. For details of these methods, see Table 1 and Begossi (2008).

Results

In the 14 fishing communities, a total of 585 fishers were interviewed, 1,453 fish were collected, and 1,761 fish landings were recorded, from 2002-2009 (except for Búzios Island, which was sampled earlier) (Table 1).

Fish species caught

The most common reef fish caught usually by hook and line in the studied artisanal fisheries on the southern and northern Brazilian coasts were from two families: snappers (12 species of the family Lutjanidae) and groupers (16 species of the family Serranidae) (Table 1, Fig. 1). In northeastern Brazil, snappers and groupers accounted for 13 species each; in southeastern Brazil, six species of groupers and four of snappers were caught, whereas in the south only groupers were observed (three species) (Table 1, Fig. 1). Snappers were more common in northeast, whereas groupers were more common in southeast and south Brazil (Table 1, Fig. 1). The most common snappers were *L. analis* (mutton snapper), *L. synagris* (lane snapper), *L. vivanus* (silk snapper), *O. chrysurus* (yellowtail snapper), and *R. saliens* (vermilion snapper), while the most common groupers were *E. marginatus* (dusky grouper), *M. acutirostris* (comb grouper), and *Cephalopholis fulva* (coney).

Length of caught fish

Our results showed that most *M. acutirostris* caught in Bertioga and Copacabana (southeastern Brazil, Fig. 1) were between 350 and 450 mm in length, and most *C. fulva* were 200 to 300 mm; these sizes were bigger than the length at first maturity of both species. Our sample sizes were relatively small (Table 1) and we may have missed some large specimens that were occasionally caught. We also observed that big valuable fish are often sold very soon after being caught by fishers; therefore it was not easy to record these big fishes at markets or landing points. However, the studied artisanal fishers often caught small and juvenile fish (Fig. 2), which deserves further attention. The snappers *L. analis* and *L. vivanus*, and the grouper *E. marginatus*, were frequently caught at lengths below the first maturity (Fig. 2), which indicated that large individuals of these species might be rarely caught.

Gonad development

We analyzed the gonads of several species of snappers and groupers. Eggs of the snappers *L. synagris*, *L. vivanus*, and *O. chrysurus* were especially visible in the spring (September-November). We did not observe many mature gonads in grouper specimens. The gonads of *E. marginatus* were not macroscopically visible, so we concluded that the gonad stages were either immature or resting (Begossi and Silvano 2008).

Table 1. Reef fishes of interest to local artisanal fisheries studied on the north and south Brazilian coasts. Species caught most often by fishers are in bold. Not all species have site folk names.

Region	Fishing village/town, state	Fieldwork	No. fishers interviewed[a]	No. landings, fish, or fishing trips	Reef fish species/folk name at site	Reference
NE	Mucuripe /Fortaleza, Ceará	Jan-Feb 2007 (Serranidae)	42[a]	173 fish collected from landing point (*Cephalopholis fulva*)	Lutjanidae: *Lutjanus alexandrei*[c] *L. analis*/cioba *L. buccanella*/pargo *L. vivanus*/pargo *L. synagris*/ariocó *L. jocu*/carapitanga *L. purpureus*/pachuco *Ocyurus chrysurus*/guaiúba *Rhomboplites aurorubens*/pargo-ferreira Serranidae: ***Cephalopholis fulva*/piraúna (173)** *Mycteroperca bonaci*/sirigado *M. acutirostris*/sirigado *M. interstitialis*/sirigado *Epinephelus adcensionis*/garoupa *E. morio*/garoupa	Begossi et al. (2007)
NE	Ponta Negra/ Natal, Rio Grande do Norte	Jan 2007 (Serranidae)	28[a]	No fish data	Lutjanidae: *Lutjanus analis*/cioba *L. buccanella*/pargo *L. apodus*/baúna-decisco *L. jocu*/dentão *L. vivanus*/pargo-olho-de-vidro *L. cyanopterus*/caranha *L. purpureus*/pargo-caxuxo *L. synagris*/ariocó *Ocyurus chrysurus*/guaiuba Serranidae: *Alphestes afer*/peixe-pedra ***Cephalopholis fulva*/piraúna** *Mycteroperca bonaci*/garoupa *Epinephelus niveatus*/garoupão *E. adscensionis*/peixe-gato *E. itajara*/gassarapé *E. striatus*/sirigado *E. marginatus*/garoupa *E. morio*/garoupa *E. mystacinus*/garoupão *E. nigritus*/garoupão	Begossi et al. (2007)
NE	Riacho Doce, Maceió, Alagoas	Jan 2009 (Lutjanidae)	13[b]	28 Lutjanidae collected from landing point	Lutjanidae: *Lutjanus buccanella*/boca-negra ***Lutjanus synagris* (15)** *L. vivanus*/pargo	Begossi et al. (2010)
NE	Porto Sauípe, Bahia	2005-2008, July 2008, Oct 2008 (Lutjanidae)	22 and 14[b]	89 fish collected from landing point or local fish market in 2005 137 Lutjanidae collected from landing point or local fish market in 2008-2009	Lutjanidae: *Etelis oculatus*/salamonete *Lutjanus analis*/cióba *L. alexandrei*/caranha *L. buccanella*/boca-negra *L. jocu*/dentão *L. synagris*/ariocó ***Lutjanus vivanus* (29)/vermelho verdadeiro** ***Ocyurus chrysurus* (66)/guaiúba, rabo-aberto** ***Romboplites aurorubens* (16)/ paramirim, promirim**	Silvano et al. (2006), Begossi et al. (2010)
NE	Itacimirim, Bahia	2005	15[a]	5 fish collected	Lutjanidae: *Lutjanus jocu*/dentão	Begossi and Silvano (2008), Begossi et al. (2008)
NE	Arembepe, Bahia	2003	35	124 fish in net sampling and fish store	Lutjanidae: *Lutjanus alexandrei* *L. synagris*/vermelho-ariocó *Ocyurus chrysurus*/rabo-aberto *Romboplites aurorubens*/paramirim	Begossi et al. (2008) (information on fishery)

Table 1. (continued)

NE Brazil	Valença, Bahia		18	278 fish from sampling	Lutjanidae: *Lutjanus jocu* *L. synagris* *Ocyurus chrysurus* *Romboplites aurorubens* Serranidae: *Cephalopholis fulva*/jabu *Diplectrum radiale*/margarida *Epinephelus adcensionis*/mero, mero-gato *Paranthias furcifer*/batata	Begossi (2006)
SE	Paraty, Rio de Janeiro	Aug, Nov 2008 (Lutjanidae)	15	44 Lutjanidae collected at local fish market	Lutjanidae: *Lutjanus analis*/cióba ***Lutjanus synagris*** **(34)/verdadeiro, legítimo** *L. jocu*/caranha Serranidae: *Epinephelus marginatus*/garoupa-legítima *Micteroperca microlepis*/badejo da areia	Begossi et al. (2010)
SE	Sepetiba Bay, Itacuruçá I. (Gamboa) and Jaguanum I., Rio de Janeiro	1989-1990	170	272 fishing trips at landing points 73 fish species identified	Lutjanidae: *Lutjanus synagris*/vermelho *Diplectrum radiale*/michóli Serranidae: *Epinephelus niveatus*/cherne *Mycteroperca acutirostris*/mira	Begossi (1995) Begossi and Figueiredo (1995)
SE	Itaipu/Niterói, Rio de Janeiro	2002-2003	18	142 fishing trips at landing points	Serranidae: *Diplectrum radiale*/micholi *Mycteroperca bonac*i/badejo	Begossi (2006)
SE	Copacabana/Rio de Janeiro, Rio de Janeiro	2005-2008, Feb-Nov 2006-2007 (Serranidae), April-March 2008-2009 (Lutjanidae)	23, 10[a], and 18[b]	441 fish landings 2005 140 fish collected 2005 108 fish collected 2006 35 Lutjanidae collected 2008	Lutjanidae: ***Lutjanus analis*** **(32)/cióba, vermelho-cióba** *L. cyanopterus*/caranha-preta *L. synagris* Serranidae: ***Epinephelus marginatus*** **(40)/garoupa** *Mycteroperca acutirostris*/badejo	Begossi et al. (2010)
SE	Búzios Island/São Sebastião, São Paulo	1986-1987, 14 months	75	906 fish landings (fishing trips sampled) 115 fish species identified	Lutjanidae: *Lutjanus analis*/caranha *L. synagris*/vermelho-xióva *L. cyanopterus*/caranha *Romboplites aurorubens*/xióva Serranidae: *Diplectum formosum*/michóli *Epinephelus marginatus*/garoupa *E. niveatus*/cherne *E. morio*/garoupa s. tomé *Mycteroperca acutirostris*/miracelo *M. bonaci*/badejo-badejo-preto *M. interstitialis*/agua fria *Paranthias furcifer*/namorado	Begossi and Figueiredo (1995), Begossi (1996)
SE	Fish markets/Bertioga, São Paulo	Oct-Sept 2006-2007 (Serranidae), June-Sept 2008 (Lutjanidae)	24 and 11[a]	44 Lutjanidae collected from local landing point and fish market	Lutjanidae: *Lutjanus jocu*/vermelho ***Lutjanus synagris*** **(29)/vermelho, vermelho-cióba** Serranidae: ***Epinephelus marginatus*** **(22)/garoupa** *Mycteroperca acutirostris*/badejo	Silvano et al. (2006) Begossi et al. (2010)
S	Pântano do Sul/Florianópolis, Santa Catarina	Feb 2005	23 and 11[a]	60 fish collected	Serranidae: *Dules auriga* *Epinephelus marginatus*/garoupa *Mycteroperca acutirostris*/badejo	

[a] Fishers ≥40 years old, with more than 25 years of fishing experience (groupers). [b] 10 years living and fishing. [c] Moura and Lindeman (2007). NE = northeast. SE = southeast. S = South

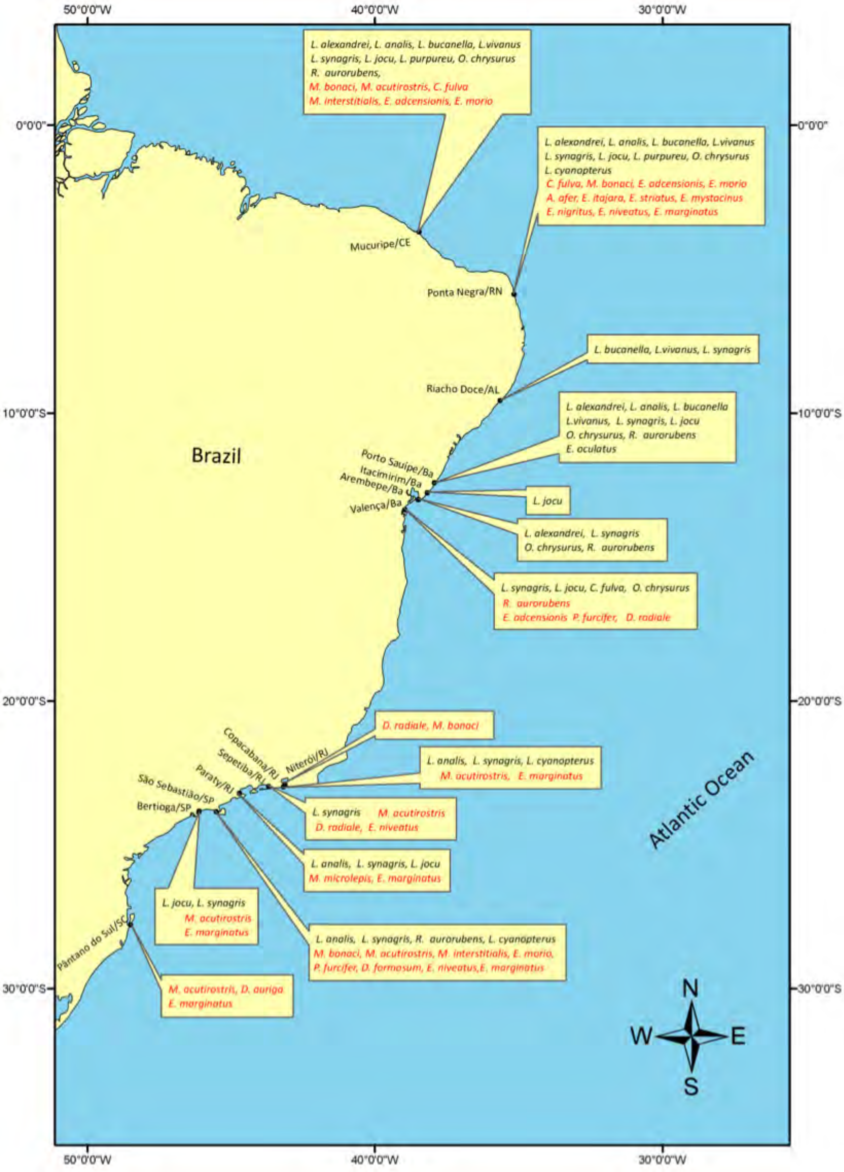

Figure 1. Map of the study sites in Brazil and distribution of species of Lutjanidae (black text) and Serranidae (red text) collected.

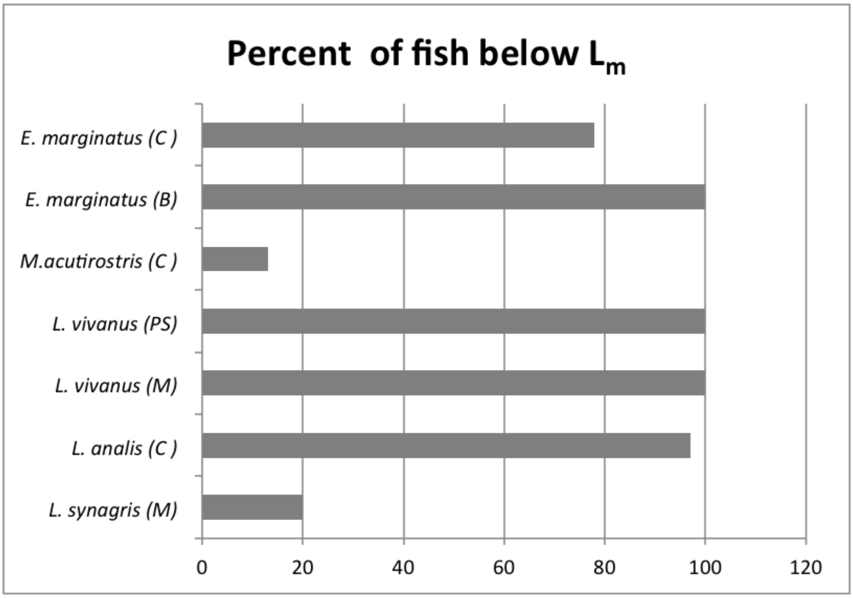

Figure 2. **Percent of fish below the length at first maturity (L_m) of snappers and groupers caught by artisanal fisheries of northeast and southeast Brazil. In parentheses are the sites or fishing communities: C = Copacabana (southeast region), B = Bertioga (southeast region), PS = Porto Sauípe (northeast region), and M = Mucuripe (northeast region).**

During interviews, fishers from Porto do Sauípe suggested that we would more likely find mature fish in October (spring). This allowed us to observe mature snapper gonads by sampling during this time of the year. When we asked 41 fishers in Mucuripe (Ceará) about coney (*C. fulva*), 30 did not know when the gonads ripened, while spring months, summer, and winter were mentioned respectively by seven, two, and one fisher. For comb grouper (*M. acutirostris*), among 21 fishers from Bertioga and Copacabana, 14 did not know when the gonad ripened, whereas three fishers thought it was in the summer, two in the winter, one in the autumn, and one in the spring.

Diet: local knowledge and observations

The stomach contents of snappers typically had fish and crustaceans, in agreement with the general consensus of interviewed fishers and with the literature (Begossi et al. 2011). Regarding groupers, fishers stressed

Table 2. Data on the diet of Serranidae species according to local fishers, based on interviews with fishers 40 years or older, 25 years living and fishing in the site (2005-2006).

	Dusky grouper Red grouper Garoupa *Epinephelus marginatus* (SE/S), *E. morio* (NE)	Snowy grouper Cherne *Epinephelus niveatus*	Comb grouper Black grouper Badejo (Sirigado in Mucuripe) *Mycteroperca acutirostris* (SE) *M. bonaci* (NE)	Coney Piraúna *Cephalopholis fulva*
Site (no. fishers)	Diet (no. citations)	Diet (no. citations)	Diet (no. citations)	Diet (no. citations)
NE: Mucuripe, Fortaleza, Ceará (14)	Fish (13)	Fish (6, most do not know)	Fish (14)	Fish (13) Anything (2)
NE: Natal, Rio Grande do Norte (28)^MA	Fish (26)	—	Fish (23)	Fish (21) Shrimp (14)
NE: Itacimirim, Bahia (15)	Fish (13)	Fish (5, most do not know)	Fish (14)	—
SE: Copacabana, Rio de Janeiro (10)	Fish (6)	Fish (7) Shrimp (5)	Fish (8) Shrimp (7)	—
SE: Bertioga (11)	Fish (8) Anything/dead (7)	Fish (6)	Fish (4) Anything/dead (4)	—
S: Pântano do Sul, Florianópolis (11)	Fish (5) Anything/dead (11)	Fish (8)	Fish (11)	—
Total (89)	Fish (71)	Fish (32 out of 61)	Fish (74)	(32 out of 42)
Literature, stomach contents	*Epinephelus marginatus*: crabs, octopi, and fish[FP]; crabs and fish[BS] *E. morio*: fish and invertebrates[FP]	Fishes, gastropods, cephalopods, and brachyuran crustaceans[FP]	*Mycteroperca acutirostris*: probably feeds on plankton[FP] *M. bonaci*: fish (adults), crustaceans (juveniles[FP,G]	Small fish and crustaceans[FP]
This study, stomach contents	—	—	*Mycteroperca acutirostris* (n = 39): Empty (16) Fish (12) Shrimp (3) Undefined (8)	*C. fulva* (n = 170): Empty (150) Fish (5) Shrimp (4) Undefined (11)

Sources: G = Gibran (2007). MA = Martinelli (2010). FP = Froese and Pauly (2011). BS = Begossi and Silvano (2008). NE = northeast. SE = southeast. S = south.

that the diet of *E. marginatus* includes more fish than crabs or other crustaceans, though the latter are especially important food item for this fish, according to our samples (Begossi and Silvano 2008). The fishers also said that fish made up most of the diet of *M. acutirostris*, which concurred with our stomach content analysis (mostly fish, with shrimp as a less important component). Fishers said that the diet of *C. fulva* consisted of fish and shrimp; these items were found in a few samples, though most had empty stomachs (Table 2).

Discussion

Snappers and groupers: fishery relevance and available data

The fish species we observed and collected in the artisanal reef fisheries on the coast of Brazil (primarily snappers and groupers) probably represent the species most commonly targeted by local artisanal fisheries, similar to the findings of other studies (Fredou et al. 2006, 2009). On the northeastern Brazilian coast, a fisheries survey ranked the catch of snappers and groupers in biomass as follows: *L. analis, O. chrysurus, L. jocu* (dog snapper), *L. synagris, L. vivanus, L. purpureus* (southern red snapper), and the black grouper *M. bonaci* (Lessa 2006). We found *L. purpureus* in Mucuripe and Ponta Negra, whereas *L. buccanella* (blackfin snapper) was found at these sites as well as in Bahia (Fig. 1, Table 1). Other authors also reported six species of Lutjanidae (*L. alexandrei, L. analis, L. jocu, L. synagris, O. chrysurus*, and *R. aurorubens*) and seven species of Serranidae (*Alphestes ruber, C. fulva, Diplectum formosum, Paralabrax dewegeri, Serranus annularis, S. balwini*, and *S. flaviventris*) in Rio Grande do Norte (Garcia et al. 2010). Compared with these other surveys, we expected to observe more *L. jocu* in the artisanal fisheries in northeast Brazil, but our fieldwork in this region was mostly confined to a period in the summer, which may not have coincided with the occurrence of this species. The fish landings we observed were mostly from hook and line fisheries, while other studies reported samples coming from hook and lines as well as longlines and traps ("covo") (Lessa 2006). In the state of Bahia, snappers have been caught with hook and line in the following order: *O. chrysurus, R. aurorubens, L. jocu, L. vivanus, L. analis*, and *L. synagris* (Klippel et al. 2005). Five species were also reported as being caught in fisheries along the central coast of Brazil (including Bahia state): *C. fulva, O. chrysurus, R. aurorubens*, and *L. synagris* (Leite et al. 2005). One study done in the northeast recommends a reduction of 80 to 90% of the current fishing effort for snappers and also highlight that artisanal fisheries are catching juveniles (Fredou et al. 2009). In Porto Seguro (Bahia state), 53 fish species were caught in 352 fishing landing and snappers represented 38% of the catch in the following order: *O. chrysurus, L. analis*, and *L. jocu*, while the black grouper accounted for about 10% of the catch (Costa et al. 2003). Compared to these surveys, our samples were representative of typical catches in northeast Brazil (Table 1), except for the relatively lower catches of *L. jocu*.

Two reproductive peaks are found for snappers in Abrolhos (Bahia, northeastern Brazilian coast): September-October and February-March (this second peak was only for one species: *R. aurorubens*) (Freitas et al. 2011). These findings are in concordance with our data for a relatively nearby region (Porto do Sauípe, Bahia); the first reproductive peak was noticed for snappers, since our data were collected in July and October.

In Brazil there are few biological and fishery data available about snappers, and even less information on groupers. Few studies have focused on groupers, and those available usually address the *E. itajara* (goliath grouper), a relatively rare species that occurs mainly in the southern Brazilian coast (Gerhardinger et al. 2006, 2009), which was rarely found in our samples (Table 1). Studies have also been done about *M. bonaci* (Teixeira and Ferreira 2004) and on the behavior, ecology, and fisheries of *E. marginatus* (Gibran 2007, Begossi and Silvano 2008). One study shows that individuals of *M. bonaci* aggregate in a "correção," which refers to a probable feeding association (Teixeira and Ferreira 2004). The aggregation of reef fishes may facilitate fishing. Thus, there is an urgent need to rapidly collect information on the ecology, local knowledge, and catch of reef fishes in Brazil.

No research has focused on monitoring grouper fisheries on the Brazilian coast, despite the fact that these reef fishes include species that are critically endangered (*E. itajara*), vulnerable (*E. marginatus*), near threatened (*M. bonaci*), or just relatively unknown (*M. acutirostris*) (Froese and Pauly 2011). Sadovy (2007) stresses the deficiency of knowledge about groupers, as well as their urgent management needs. In the Caribbean area (Brule et al. 2003, Gimenez-Hurtado et al. 2005) and in Florida (Sluka et al. 2001), some data are available for the groupers *M. bonaci* and *E. morio*, whereas *E. marginatus* has its southernmost occurrence in Rio de la Plata, between Uruguay and Argentina (Rico and Acha 2003). Snappers and groupers were reported in the fisheries of the Bay Islands (Honduras) (Gobert et al. 2005) and in the southern Caribbean (Mendoza and Larez 2004). In Brazil, *M. bonaci*, together with 11 species of Epinephelinae, formed a small but valuable fisheries yield (US$4.00 per kg) in Porto Seguro, northeastern coast (Costa et al. 2003). Other frequent catches reported there were *E. morio, M. interstitialis*, and *C. fulva* (Costa et al. 2003). In southeast Brazil, such as at Búzios Island, where eight species of serranids were collected (Table 1 and Fig. 1), the bulk of the catch was composed of bluefish and squid rather than by reef fishes, such as groupers (Begossi 1996). However, groupers were valuable in the market, and thus targeted by fishers. The vulnerability of groupers is enhanced because they are late mature species and many are monandric protogynous hermaphrodites (Marino et al. 2001). This reproductive feature increases the risk of individuals being caught before spawning at least once in their lifetime (recruitment overfishing) (Sadovy 2001). Indeed, a survey based on historical data and interviews with fishers of distinct ages showed that the abundance of a grouper species became severely reduced in the Gulf of Mexico, much before the onset of regular fish landing samples (Saenz-Arroyo et al. 2005).

Evidence in the literature suggests that E. niveatus is caught in deep waters (200-250 m) (Lessa 2006). Fishers from the studied communities of Copacabana and Pântano do Sul (Fig. 1) also mentioned that this species was currently rarely found in their fisheries. This warrants further investigation because the species may indeed be rare on the southeastern coast of Brazil, or maybe fishers are not traveling far enough from the coast to catch this fish, which is vulnerable to overfishing (Cheung et al. 2005, Froese and Pauly 2011).

Groupers are among the most common reef fish in fisheries of southeast Brazil and, with the exception of *M. bonaci*, are also common in the northeast (Fig. 1). According to our results from the fish landings in southeast Brazil, both *E. marginatus* and *M. acutirostris* are still frequently caught, but the former has been caught mostly at small sizes (Fig. 2). Fishers target large and high-priced fish and often experience a dilemma between consuming and selling their catch. Typically, they sell the large, valuable fish and consume the low-cost, small fish. Thus, we expect that fishers are generally looking for large fish, but when they cannot find them, they catch and keep the available small fish instead.

Conservation of reef fishes in Brazil: current situation and suggestions

In Brazil, snappers and groupers are valuable targets for artisanal fisheries because they have high market prices. However, there is no sufficient information available on the catches, nor on the biology or the ecology of these species. Although there is some information available for snappers (Fredou et al. 2006, 2009) and for the grouper *E. itajara* (Gerhardinger et al. 2006, 2009), there are no direct conservation policies for snappers and groupers in Brazil. However, fishers' local ecological knowledge (LEK) about snappers and groupers could be used, at least in the short-term, for management (Gerhardinger et al. 2006, 2009; Silvano et al. 2006; Begossi 2008; Begossi and Silvano 2008). The only direct relevant policy of the Brazilian Federal Environmental Agency (IBAMA), titled “Portaria IBAMA #53/05,” sets the minimum capture size of *E. marginatus* at 470 mm and that of *M. acutirostris* at 230 mm (www.ibama.gov.br/pesca-amadora). Although there are governmental initiatives for extractive reserves on the coast of Brazil (Seixas et al. 2009), such initiatives have been debatable in terms of legitimate processes and contributing to threatened species (Begossi 2010), such as reef fishes, since there is no specific law controlling the fishing of those species.

Our results (Fig. 2) showed that some snappers and groupers are usually caught at juvenile stages, which is particularly dangerous for these slow-growing, late-maturing species. This trend was especially striking for *L. analis* in Copacabana, where we performed a more intense sampling (Table 1, fishers preserved snappers in ice for further sampling): we did not collect any snappers above the size at first maturity. Previous information shows snappers being caught approximately in ranges 240-560 mm for *L. synagris* (Bertioga and Riacho Doce), 330-540 mm for *O. chrysurus* (Porto do Sauípe), 250-410 mm for *L. vivanus* (Porto do Sauípe), and 270-500 mm for *R. aurorubens* (Porto do Sauípe) (Begossi et al. 2011). Regarding the few species for which data are available (Lessa 2006), we observed that *L. analis, L. synagris*, and *L. chrysurus* are being overexploited, and *L. vivanus* is approaching this point. *L. analis* is listed as vulnerable, and *E. marginatus* is listed as endangered on the IUCN Red List (Froese and Pauly 2011). Resilience is low for these reef species, except for *L. synagris*, which shows medium resilience (Froese and Pauly 2011).

These reef fishes are facing serious problems because (a) they are high-priced, valuable targets caught by poor fishers; (b) no policies aimed at direct conservation of these fishes have been implemented; (c) some policies, especially in a recent past, have supported the establishment of no-take and conservation areas, mostly in a top-down manner, excluding native populations from their original areas. However, there has been scarce monitoring, control, and attempt to determine the efficacy of such no-take areas, except for the Corumbau Extractive Reserve (Francini-Filho and Moura 2008); and (d) little research has been done to understand fishers' motivations or to explain their decision-making processes (Begossi 2010, Lopes and Begossi 2011). Moreover, many more data are needed in order to know about the reproduction of snappers and groupers on the coast of Brazil, since data for Latin American fisheries are available mostly for the Caribbean (Sadovy 1996).

We propose to overcome these problems by making conservation interesting to poor people in tropical countries (Begossi 2010) and thus co-opting fishers to work for conservation, using their local knowledge and employing them in the monitoring process, as follows:

a. Co-opt or stimulate, through economic incentives, fishers and their families to manage resources through co-management processes that include fishers in decision-making processes (Castilla and Defeo 2005, Castello et al. 2009, Seixas et al. 2009, Begossi 2010, Begossi et al. 2011).

b. Use all available knowledge, including fishers' LEK (Begossi and Silvano 2008, Silvano and Valbo-Jorgensen 2008).

c. Build up, through co-management processes, marine reserves, closed areas, rotation of islands for fishing, and other conservation measures as suggested in previous surveys (Castilla and Defeo 2005, Seixas 2006, Gelcich et al. 2009, Seixas et al. 2009). Studies have reported that fish stocks benefit from co-management in marine protected areas, extractive reserves, development reserves, or fishing agreements (Man et al. 1995, Guidetti 2006, Begossi 2010). Protogynous fish have particularly benefited from marine protected areas (Sadovy 1996, Molloy et al. 2008), and groupers thus might also benefit from such protection. On the northeastern Brazilian coast, no-take areas established in a co-management system have enhanced the abundance of several exploited fish, including groupers (Francini-Filho and Moura 2008).

d. Perform systematic samplings and biological analysis of fish from artisanal fishing trips, through collaborative efforts with fishers (Begossi 2008).

e. Finally, use "payments for environmental services (PES)," a measure commonly used for forests (Wunder et al. 2008, Muradian et al. 2010) that can be applied to fisheries, including those of snappers (Begossi et al. 2011).

Summarizing, snappers (Lutjanidae) and groupers (Serranidae) are caught by artisanal fisheries on the Brazilian coast. However, several species of these fishes are being caught before their first maturity, which will have a severe impact on their populations. To avoid reef fish depletion, co-management measures could be built, especially because artisanal tropical fisheries have characteristics that allow co-management on a local level, such as a small-scale context, the existence of local rules and local knowledge that could be applied for management purposes, and a scarcity of scientific data. The critical management question is how to promote conservation within a population of extractors with social needs, especially if the target species are high-valued. The solution is to make conservation interesting for these artisanal fishers, since taking into consideration economic attributes could motivate fishers to engage in management (Sethi et al. 2010). We therefore suggest making conservation attractive to Brazilian fishers by adopting economic incentives (e.g., PES) and by involving fishers in the construction and monitoring of management initiatives.

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Steps for Future Progress in Ecosystem-Based Fisheries Management: What's Next?

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Background

Since the 1990s, fisheries managers have been advised to broaden their scope of awareness beyond single-species considerations. Motivations for this broader approach stem from typical poor performance of single-species fishery management worldwide, heightened awareness of interactions among fisheries and ecosystems, a growing understanding of the functional value of ecosystems to humans, and recognition of a wider range of societal objectives for marine ecosystems beyond fishery catches. This new approach is often called ecosystem-based fisheries management (EBFM) or an ecosystem approach to fisheries (EAF). An EAF strives to balance diverse societal objectives by taking into account the knowledge and uncertainties of biotic, abiotic, and human components of ecosystems and their interactions, and applying an integrated approach to fisheries within ecologically meaningful boundaries (Garcia et al. 2003).

Considerable progress has been made by organizations such as the Food and Agriculture Organization of the United Nations (FAO), International Council for the Exploration of the Sea (ICES), North Pacific Marine Science Organization (PICES), and many others to develop the conceptual frameworks, ecosystem indicators, modeling approaches, risk assessments, and other facets of EBFM/EAF. Additional development and implementation of EBFM remains a high priority internationally. For instance, it forms the foundation of the current ICES Strategic Plan, “A Vision Worth Sharing,” and it is an important component of the new PICES Science Program, “Forecasting and Understanding Trends, Uncertainty and Responses of North Pacific Marine Ecosystems” (FUTURE).

The goals this international symposium, “Ecosystems 2010: Global Progress on Ecosystem-Based Fisheries Management,” were to: (1) evaluate global progress toward EBFM by reviewing regional case studies, development of new analytical tools and practical approaches toward future progress; and (2) offer explicit, practical advice for future progress in implementation of EBFM. The 18 papers published in this volume have each contributed toward the first goal. Toward the second goal, a panel session was held on the last day of the symposium. The panel was chaired by Glen Jamieson and also included Howard Browman, Rick Fletcher, Kwame Korenteng, Mitsutaku Makino, and Patricia Livingston. Panelists offered their perspectives on the take-home messages from the symposium, and meeting participants posed questions and proffered their insights. Our attempt to provide a succinct summary of the panel session follows.

Panel discussion

The symposium achieved a general consensus on several aspects of EBFM. There was a convergence on broad ecosystem management objectives, principles, approaches, tools, and the need for involvement of stakeholders. The foundations for this approach have been articulated by the FAO (Garcia et al. 2003) and others. Important elements for the practical implementation of EBFM emerged from progress in the Gulf of Maine (Stephenson and Annala 2012) which, thanks to shared governance by two wealthy neighboring nations, is one of the world's best studied ecosystems. This partnership has been able to provide: (1) *leadership* to move forward with respect to existing activities in a changing environment with a vision to the future; (2) *effective governance* of people that takes advantage of advances in the field of management science; (3) *manage or change expectations* to conform with those of society for the world's last wild capture activities; (4) *interdisciplinary teams* of experts to address the inherent multidisciplinary problems associated with EBFM; and (5) a *participatory process*—active involvement of a diverse set of stakeholders.

A consensus emerged that fisheries management is a people-based process that is informed by science and other information. Consequently, stakeholder involvement is essential with a dedicated commitment from those in leadership roles being necessary to achieve this. From a practical standpoint, effective implementation and enforcement of regulations requires "buy in" from stakeholders, just as in the case of single-species management. However, it was also pointed out that stakeholders can help with two frequently cited obstacles to EBFM implementation: failure to define operational objectives and lack of data and information. It may take numerous meetings to build trust among various sectors, as was well documented in the case of EBFM in the Shiretoko World Natural Heritage in Japan (Makino 2012).

During the panel discussion there was general agreement that EBFM should move forward even with imperfect/incomplete information. The contrasts in data, knowledge, models, and governance between the world's developed versus developing countries were quite stark. For this reason, it was noted that different kinds of situations call for different kinds of EBFM. For instance, the most successful attempts to implement EBFM in developing countries presented at this symposium all appeared to involve minimal data and no quantitative modeling, but a very high level of stakeholder involvement and commitment. It was also pointed out that EBFM is an adaptive process, changing as new data and information become available. In other words, the common statement, EBFM is an evolutionary process not a revolutionary process, remains apropos.

Through the talks presented at the conference it was made clear that EBFM will ultimately evolve toward the establishment of regional level plans within which the actions and arrangements for individual fisheries will be nested. Therefore, while EBFM will require consideration of multiple interactions at the regional level of the ecosystem, most of the management actions will still have to be made at the level of single species and individual fisheries. In other words, it is important not to become lost in the EBFM process, but instead to focus on generating good outcomes; and conventional fisheries management tools will still be the primary mechanisms used to deliver good outcomes.

Symposium participants agreed that adopting a risk-based approach provides the most appropriate and practical framework for implementing EBFM across the spectrum of data-limited to data-rich situations. With fisheries management essentially a specific form of risk management, a clear consensus emerged during the panel session for the need to conduct risk assessments to help set priorities. A large number of risk assessment tools are available for application to fisheries management situations (e.g., Fletcher 2005; Fletcher et al. 2010; Zhang et al. 2009, 2011; Hobday et al. 2011) with summaries of these now available from FAO's EAF Toolbox website (www.fao.org/fishery/eaf-net). The regional level EBFM system applied in Western Australia (Fletcher et al. 2012) highlighted the need not only to consider ecological risks but also to consider social, economic, and governance risks to determine which of these most requires direct action to deliver the best community outcomes. This approach also showed how the complexity generated by regional level assessments need to be reduced to enable practical use within the management decision making process.

There are already a number of quantitative techniques, such as stock assessment, management strategy evaluations, and ecosystem modeling (e.g., Atlantis, Ecosim), that can where appropriate be applied to assist in the application of EBFM. With respect to modeling, some presenters during the panel session stressed that the end goal of the effort should not be the model itself, but rather linkage to management. Atlantis is an example of an ecosystem model that was designed for use in management strategy evaluations (Fulton et al. 2011). Nevertheless, some symposium participants voiced concern about how the multitude of uncertainties can be addressed in a satisfactory manner using multispecies and ecosystem models where local data were lacking. Management strategy evaluations for single species have been developed whereby the perceived primary sources of uncertainty can be addressed singly or in combination. However, new methods will need to be developed to address the many sources of uncertainty in multispecies and ecosystem models. There was also a recognition that the collection of data necessary to "feed" these models comes at a cost and must be justified in terms of whether it will improve management or whether some other actions may be more beneficial. Additionally, advancements are needed in tools for spatial management and for improving predictions of the impacts of climate change.

Nonetheless, a suite of tools, ranging from expert judgment to complex ecosystem models, are currently being employed globally to address management questions. The consensus from the symposium was that we have clearly moved from trying to define or understand the concept of EBFM/EAF to actual implementation. Development of the social dimensions of EAF are at an earlier stage of development compared with the ecological aspects. The next global challenge appears to be moving toward the broader concept of an ecosystem approach to management (EAM), which involves cross-sectoral ecosystem-based management. Challenges to progress in that area are primarily institutional, legal, and social.

Finally, a clear conclusion of this symposium was that the greatest risk to the world's fisheries is not the lack of scientific information, but rather the lack of effective governance. While there was much discussion about the enormous costs required to learn all the things about marine ecosystems that we would like to know, the fact is that we can already implement EBFM even in data-limited situations by adopting a risk-based approach. Good outcomes can be achieved by applying a practical and precautionary approach that adopts good governance principles with effective implementation by having suitable political and institutional commitment. The corollary is that for systems where single-species fisheries management has failed owing to ineffective governance, adopting EBFM is likewise doomed to failure. Rectifying this common central problem of poor governance is a prerequisite for any form of successful fishery management from single species to EBFM.

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