

Using ecosystem modeling for fisheries management: Where are we?

Villy Christensen¹ & Carl J. Walters

University of British Columbia, Fisheries Centre, 2202 Main Mall, Vancouver BC, Canada V6T 1Z4.

Abstract

The writing is on the wall: we are moving toward ecosystem-based management of fisheries, and it is clear that ecosystem modeling is an important tool for evaluating scenarios and trade-offs as part of such a move. The current contribution evaluate if ecosystem modeling is an active research field, if the models (potentially) are useful for fisheries management, and if ecosystem models actually are being used in fisheries management. In addition we present some recommendations for how the move toward ecosystem-based management can be supported through an adaptive environmental assessment and management process.

Keywords

Ecosystem modeling, fisheries management, adaptive environmental assessment, Ecopath with Ecosim, multispecies modeling

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¹Correspondence: Villy Christensen, Fisheries Centre, University of British Columbia, 2202 Main Mall, Vancouver BC, Canada V6T 1Z4. Tel.: (1) 604 822 5751. v.christensen@fisheries.ubc.ca

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Introduction

The writing is on the wall: we are moving toward ecosystem-based management of fisheries, and it is clear that ecosystem modeling is an important tool for evaluating scenarios and trade-offs as part of such a move. The current contribution evaluate if ecosystem modeling is an active research field, if the models (potentially) are useful for fisheries management, and if ecosystem models actually are being used in fisheries management. In addition we present some recommendations for how the move toward ecosystem-based management can be supported through an adaptive environmental assessment and management process.

It is important at the outset to be careful about what it means to ‘use’ a model in fisheries management. At one extreme, we might imagine taking model predictions at face value and applying them blindly in setting reference points and regulations; some managers seem to hope for or expect such models, presumably as absolution or excuse for not making thoughtful choices in the face of uncertainty. No fisheries model, whether a highly ‘precise’ single-species assessment or a crude ecosystem biomass flow scheme, will ever be reliable enough to use in such an uncritical way, if for no other reason than unpredictability in environmental forcing changes. At the opposite extreme, we can certainly use even very simple trophic interaction calculations to screen qualitative policy options and answer very basic questions raised in management settings, (e.g., could predation rates by some particular predator that fishers do not like be high enough to ever justify a culling policy). Between these extremes, to ‘use’ a model in policy can mean a whole range of increasingly precise, and correspondingly more likely to be incorrect, qualitative and quantitative answers to queries by management stakeholders.

Development of ecosystem models would be largely a waste of time if their only intended use were to improve the predictions and prescriptions of single-species management, by accounting for some causes of variation in growth, natural mortality, and recruitment rates. Except in cases where we see persistent nonstationarity in such rates, or only have data for very limited ranges of abundance well after early stock declines due to fishing, existing single-species models based directly on measured rate data are quite adequate to define reference points and feedback policies for coping with unpredictable variation. Where ecosystem models become important, even logically necessary to compare policy options, is in situations where management objectives and policy options are broadened to include performance measures (e.g. diversity, abundance of non-target species) and actions, (e.g., habitat fertilization, protected areas) about which single-species models simply have nothing to say.

How active is the field of ecosystem modeling?

To determine how active the field of ecosystem modeling related to fisheries management is we evaluate the trend in number of primary publications dealing with ecosystem approaches to fisheries and developing or applying ecosystem modeling. Our criteria for equating trends in publications numbers with activity levels is that the number of publications appearing must be showing continuous increase in recent years.

We conducted a Web of Science search for the words (Ecosystem* + model* + fish*) as well as for (multispecies + fish*), where the wildcard (*) denotes any ending. We included results for the years 1992-2004, and combined the results obtained using the two search terms. For each of the 'hits' we examined the abstract (where available through Web of Science), or title or paper copy (where not) to determine if the publication actually developed, applied or reviewed ecosystem modeling activities as opposed to only referring to it.

The search for ecosystem and multispecies modeling publications yielded a total of 1335 results, of which 173 (or 13%) were publications developing, applying or reviewing ecosystem modeling approaches, see Figure 1. The results indicate an approximately 23% growth per year in the annual number of publications over the twelve year period studied. This indeed indicates that ecosystem modeling is an active field of research experiencing rapid growth.

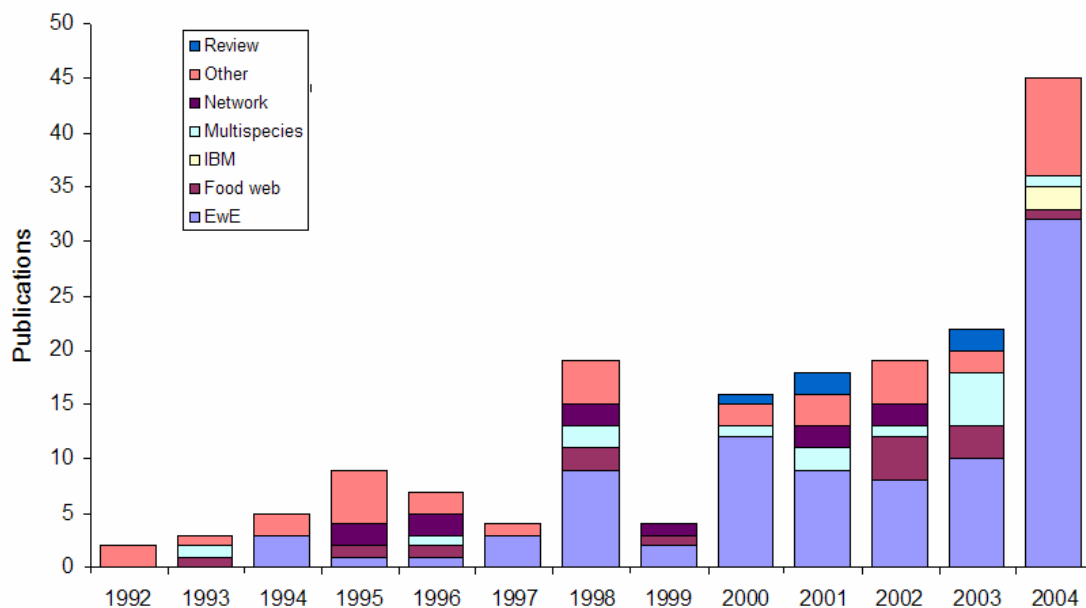


Figure 1. Trend in number of publications developing or applying ecosystem modeling related to fisheries research. Results or obtained from a Web of Science search for (Ecosystem* + model* + fish*) or (multispecies + fish*).

More than half of all the modeling publications in the Web of Science search are related to the Ecopath with Ecosim (EwE) modeling approach (www.ecopath.org; described in Polovina, 1984, Christensen and Pauly, 1992, Walters *et al.*, 1997, Walters *et al.*, 1999, Pauly *et al.*, 2000, Walters *et al.*, 2000, Christensen and Walters, 2004). We assume that this dominance of EwE modeling is caused in part by the long history of the approach and its availability combined with the extended support for it that over the time period examined (more than thirty training courses and workshops have been conducted

throughout the world, with participation of more than 600 scientists). A factor is also that the entry threshold for this kind of modeling is low; the starting point can be a descriptive model of what is known of the resources in an ecosystem, their state values and interactions. The end level is advanced, however, asking detailed questions of causes and effects over time, trade-offs, policy optimizations, and spatial design.

There are other modeling approaches and we strongly support that a diversity of models be used as we move toward ecosystem based management of fisheries. Modeling should not be a 'one fits all'; it is important that models are built to address the policy questions at hand, rather than the opposite. We will, however, focus this overview on the use and capabilities of EwE as this is the area with which we are most familiar.

Are ecosystem models potentially useful for fisheries management?

One of us did not think so just a decade ago: "We believe the food web modelling approach is hopeless as an aid to formulating management advice; the number of parameters and assumptions required are enormous." (Hilborn and Walters, 1992). While the statement serves to illustrate how careful one has to be with categorical statements (Walters and Martell, 2002), it is also clear that the author in question has changed his mind, noting that a substantial part of a recent text book is dedicated to demonstrating how food web modeling can be of use for fisheries management (Walters and Martell, 2004). We will not seek to duplicate that effort here, but rather discuss the modeling procedure as well as illustrate the usefulness of ecosystem models for management based on experience obtained from a series of case studies where ecosystem models have been used to investigate questions of interest for ecosystem based management of fisheries.

Indeed, we find that as fishery policy moves beyond the objectives in place for single-species management there is in fact no choice but to adopt more elaborate ecosystem models. Policy choices for ecosystem-based fisheries management will necessarily involve predictions about the impact of non-traditional policy choices, and our abilities to and options for making such predictions are severely limited. In the past we have based comparisons of ecosystem-related policy choices, (i.e., predictions about the efficacy of the choices) on methods ranging from very simple risk avoidance models, (e.g., don't harvest forage fishes at all because production of larger fish may be impacted), to simple food chain or trophic cascade models, (e.g., to evaluate policies related to fertilization), to very complex food web models that attempt to explore possible reverberating effects going beyond direct predator-prey interactions, (e.g., if species A is reduced by fishing, will species B increase, and will that increase in B cause a large change in abundance for species C, D, E, ... ?). Much of the recent ecosystem modeling work has been aimed mainly at assessing risks of the more complex reverberating effects such as 'cultivation-dependensation' effects (Walters and Kitchell, 2001), on the assumption that complex interactions are likely to result in counter-intuitive responses (Yodzis, 2001).

If ecosystem models are then to be of use in fisheries management it is necessary to demonstrate that the models can replicate historic trends in ecosystems, as a first step toward building some confidence that they can make plausible extrapolations to novel situations. We will start by examining this in more detail.

Evaluating model behavior

The time-dynamic module, Ecosim (Walters *et al.*, 1997, Walters *et al.*, 2000) of EwE as well as the spatial-dynamic module, Ecospace (Walters *et al.*, 1999) are built on what has been termed ‘foraging arena theory’ (Walters and Juanes, 1993, Walters and Martell, 2004). The basic assumption in this theory is that prey behavior limits predation rates, causing trophic interactions to be highly organized in time and space. When evaluating model behavior it is necessary to ‘tune’ the models so as to reflect the net effect of this organization on large-scale interaction rates. For Ecosim models this mainly involves setting one parameter for each species, a vulnerability coefficient.

Vulnerabilities are factors describing how a change in a given predator biomass will impact predation mortality for a given prey. Low vulnerability factors imply that an increase in predator biomass will not cause any noticeable increase in the predation mortality the predator will cause on the given prey. High vulnerability factors oppositely indicate that if the predator biomass is for instance doubled, it will cause close to a doubling in the predation mortality rate on a given prey. This then relates directly to assumptions about the carrying capacity for the predator in question.

If the predator is close to its carrying capacity, lowering its biomass (e.g., through a pulse fishing), will be compensated for by an increase in the consumption/biomass ratio for the remaining individuals and increased surplus production will let the predator biomass grow back towards its carrying capacity. The population is thus stable when close to its carrying capacity. If on the other the predator is far from its carrying capacity, the individual predators will be very little impacted by competitions from its peers; there is little surplus production. A change in its biomass, (be it through pulse fishing, stock enhancement, or the appearance of a strong year class), will, due to the relative absence of compensation due to surplus production factors, lead to the biomass directional change being maintained. Populations are unstable when far from their carrying capacities.

The settings for vulnerabilities, which as described above directly relates to assumptions about carrying capacities, thus have direct implications for any EwE model’s abilities to replicate historic time series data. Were historic estimates of abundance available we would in many cases have estimates for carrying capacity as well. Such cases are, however, much more the exception than the rule. On the contrary, we usually find that very little historic information is available about a given ecosystem. Typically the available assessments go back only one to two decades, (e.g., to when the VPAs are ‘converged’), and assessments are only available for the commercially most important species. For evaluating ecosystem model behavior it is, however, desirable to go further back, as it is necessary to challenge the models with time series trends with some contrasts in abundance estimates. Thus, when we talk about ‘historic’ information we typically consider a period stretching back to where industrialized fishing became a factor, typically half a century back to the mid-1900s.

For many ecosystems we have had to complement the available assessments with additional analysis for exploited species not covered by the single-species assessments conducted as part of the management process, e.g., for the Chesapeake Bay, (Christensen *et al.*, MS). For this our choice of stock assessment method has most often been stock reduction analysis (Hilborn and Walters, 1992, Walters and Martell, 2004) where

abundance estimates are obtained based on tuning catch and productivity information to time series trends or abundance data, see, e.g., Walters *et al.* (2005) and Christensen and Martell (2005, this meeting), with a deliberate aim to reach as far back in time as possible so as to estimate unfished ‘limits’ for natural abundance.

Hindcasting

The first part of the criteria we set for using ecosystem models for fisheries management is that the models can replicate historic trends in ecosystems. We evaluate this by estimating the sum squared residuals between observations and ecosystem model predictions using log likelihood criteria (Figure 2). The ‘observations’ include biomasses, mortality rates, average weights, diets and/or catches, and may be predictions from other models, e.g., VPAs or stock synthesis models, which may be a call for concern. It is indeed preferable to fit the ecosystem models to the data that is used in the stock assessments rather than the output from these assessments. This is, however, rarely possible due to data availability, yet it is a concern that should be taken seriously when the purpose of the analysis is not so much to compare single-species and ecosystem model predictions, but to use the ecosystem models as part of the management process.

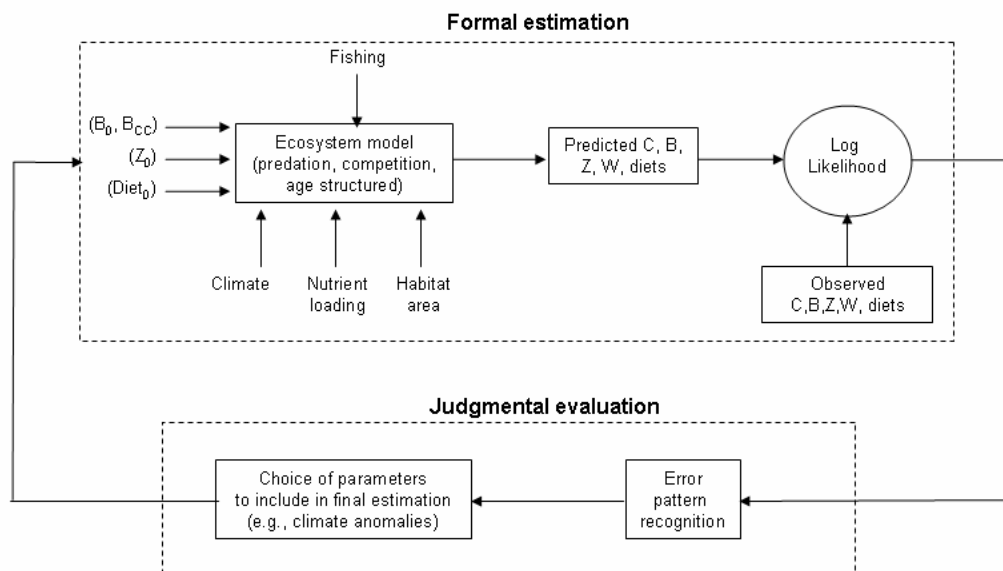


Figure 2. Overview of the EwE modeling process. Using log likelihood criteria the input parameters or anomalies, (e.g., climate) may be estimated based on a non-linear search routine. Prediction (fitting) failures after each estimation trial then inform judgmental changes in model structure and parameters. B is biomass, Z is total mortality, C is catch, W is average weight. Subscript 0 refers to the Ecopath model base year, and CC to carrying capacity.

We typically see that the summed squared residuals are reduced to around one third of initial values as part of the model fitting procedure (Figure 2). This procedure includes an iterative approach with,

- (1) Iterations between the Ecopath and Ecosim modules of EwE to weed out obvious data errors (typically conversion mistakes or erroneous estimates of species productivity rates);
- (2) Estimation of biomass accumulation rates (changes in biomass of a given species in the base Ecopath model year) to set the Ecosim simulations of on the right tangent;
- (3) Estimation of vulnerabilities, typically predator-specific rates, relating to assumptions of carrying capacities as discussed above. The estimation can be done using a non-linear search routine, minimizing the summed squared residuals;
- (4) Estimation of system productivity rates over time. This estimation can also be through a non-linear search routine, and is often combined with the search above.

Table 1. Ecosystems where EwE models have been tested using historical trend data. N, S, W, E indicate geographical subsets, C is central.

Gulf of Thailand	(FAO/FISHCODE, 2001)
North Sea	(Christensen <i>et al.</i> , 2002)
C N Pacific	(Cox <i>et al.</i> , 2002)
Strait of Georgia	(Martell <i>et al.</i> , 2002)
W coast of Vancouver Island	(Martell, 2002)
English Channel	(Stanford, 2002, Araujo <i>et al.</i> , <i>in press</i>)
E Bering Sea	(NRC, 2003), Aydin
E tropical Pacific	(Olson and Watters, 2003, Watters <i>et al.</i> , 2003)
Baltic	(Harvey <i>et al.</i> , 2003)
S Benguela Current	(Shannon <i>et al.</i> , 2004)
Lake Superior	(Cox and Kitchell, 2004)
S Catalan Sea	(Coll <i>et al.</i> , 2005)
S Brazil Bight	(Freire, 2005)
Scotian Shelf	Bundy
W and C Gulf of Alaska	Gaichas
Bay of Quinte	Millard et al.
Oneida Lake	Millard et al.
C Chile	Neira and Arancibia
Chesapeake Bay	(Christensen <i>et al.</i> , MS)
Aleutian Islands	(Guénette <i>et al.</i> , MS, Heymans <i>et al.</i> , MS)
Southeast Alaska	(Guénette <i>et al.</i> , MS, Heymans <i>et al.</i> , MS)
N British Columbia	(Ainsworth, <i>in prep.</i>)
Norwegian/Barents Sea	(Skaret and Pitcher, <i>in prep.</i>)
South China Sea	(Cheung, <i>in prep.</i>)
Upper Gulf of California	(Lozano, <i>in prep.</i>)
NE Pacific	(Preikshot, <i>in prep.</i>)
NW Hawaiian Islands	Polovina et al. (unpublished)
Tampa Bay	Walters et al (unpublished)

Fitting to time series data has been conducted for a series of ecosystems; see the overview in Table 1 for an incomplete list. If we seek to generalize the results from the model fitting to time series data, it seems that most commonly the models can be made to

fit well (though ‘fit well’ is not a strictly defined term; it is more a visual impression). We mention some experience from the model fitting:

- For species or groups where there are strong impacts from fisheries the ecosystem models generally fit very well. This is not a big wonder though as this typically involves comparing results from single-species and ecosystem models, with both predominantly driven by fishing pressure;
- Where single-species and ecosystem model predictions diverge the difference can often be explained by obvious model assumptions. An example is the Strait of Georgia where juvenile herring abundance in the 1960s was much higher in the VPA than in the Ecosim model (Walters and Martell, 2002). The difference is attributed to juvenile herring having a much higher natural mortality due to higher predator abundance in the 1960s in the Ecosim simulations. Using the Ecosim model to obtain estimates for predation mortality (similar to how the MSVPA is used in the ICES area) would minimize the differences between model hindcasts;
- Ecosystem models may get some species right, yet have problems with others; it was as an example possible for Cox and Kitchell (2004) to simulate population trends for the some important species of whitefish, (e.g., lake chub) while being unable to fit for some of the other species, e.g. siskowet lake trout. Inability to fit the data for some species may well be due to lack of information about species-specific environmental drivers and effects for such species. The implication of this ‘one rotten apple doesn’t spoil the barrel’ behavior is that ecosystem models can be used to address specific policy questions relating to part of the ecosystem only, indeed we encourage this approach to modeling compared to an approach aimed at including and explaining everything for all parts of an ecosystem in the same go;
- We observe impacts on forage species from changes in predator abundance, and occasionally also the other way around where changes in prey abundance impacts predators, e.g., for striped bass and menhaden in the Chesapeake Bay (Christensen *et al.*, MS). It is often impossible to evaluate the credibility of the predicted trends for forage species due to a lack of time series information as well as uncertainty about variation in productivity rates;
- We generally see very strong reactions throughout the ecosystem from changes in system productivity. Thus, it is as a rule necessary to incorporate or estimate overall system productivity over time in form of nutrient availability or primary productivity. Indeed, the only ecosystem where we have been unable to find indications of changes in system productivity is for the Gulf of Thailand (FAO/FISHCODE, 2001) where the fisheries impacts are very strong while system productivity may be less variable.

Overall, we find that the ecosystem model applications in Table 1 demonstrate a capability for hindcasting ecosystem trends over the time periods where data is available. Given past experience with the chaotic nature of Lotka-Volterra type models where population explosions and extinctions is extremely difficult to circumvent, we attribute

the improved behavior of the Ecosim type models (which are Lotka-Volterra derived) to the inclusion of prey behavior through the foraging arena theory as discussed above.

It is clear from the time series fitting that this form of testing for ecosystem models is data demanding. However, the data required are mainly data that should be accessible in any case; it is the ecosystem history – how we have exploited the ecosystem resources and what we know about population sizes over time. It is, however, not so that an extraordinary number of parameters must be estimated (by fitting the model to historical data) as part of the modeling process. Indeed, if we sum up the number of parameters included in the single-species assessments in an area with fisheries management, the sum may well exceed the number of parameters included for ecosystem modeling of the same area.

Predictive capabilities

Models are made for predictions, and the hindcasts discussed above, though they are model predictions they are not necessarily good examples of predictions. The next step is to evaluate if the ecosystem models can make plausible extrapolations to novel situations.

We emphasize that the predictive approaches are uncertain and that this uncertainty is largely caused by limited data on states and processes. There is a clear lack of long-term monitoring data on non-target species and life stages, and the concentration of interaction effects (trophic, habitat) that takes place in early life stages (recruitment) is difficult to monitor. Juvenile mortality rates are as a rule very high; this is thought to be mainly by predation, concentrated in short space-time windows, for instance marine mammal predation on juvenile salmon as they enter saltwater.

Contributing to the difficulty is that while we may fit the historic data well, it is far from certain that the fitting is good for the right reason (Essington, 2004). We typically evaluate fishery, environmental, and trophic effects in historical data, and seek to find simple explanations for often confounding results. An example: the initial Ecosim run for the Northwestern Hawaiian Islands indicated that a major decline in rock lobster abundance could be linked to fishing pressure and trophic interactions, and that this might had caused the endangered monk seal to decline; a simple parsimonious explanation. When, however, it was incorporated that a subsequent stop to the rock lobster fisheries did not lead to a recovery of rock lobster (and subsequently of monk seals) the story had to change. Satellite chlorophyll data indicated a persistent 40-50% decline in primary production at the time, and when this was incorporated the combined effect ‘explained’ both the continued monk seal decline and persistent low lobster abundance. Can we be sure we have the right explanation? Only time will tell.

It is clear, however, that we cannot resolve uncertainty about how ecosystems change based on models and time-series data only. As an example: salmon declines are often assumed to be linked to freshwater habitat loss, and we can make models that replicates trends in salmon runs by changing habitat parameters; but how do we know that ocean ecosystem changes aren’t more important factors (Bradford, 1999)?

Most of the ecosystem management predictive questions being asked today concern three issues: (1) ‘top-down effects’, i.e., impacts of harvesting top predators on species at lower trophic levels; (2) ‘bottom-up effects’, i.e., impacts of fisheries on species at intermediate

trophic levels on the capacities of those species to provide ‘ecosystem support services’ to species (harvested or not) at higher trophic levels; and (3) impacts of deliberate or inadvertent changes in habitat factors, e.g., through coastal nutrient loading, clearing of mangrove for aquaculture, or removal of bottom structure through trawling.

Based on experience with a series of ecosystem model case studies we have found,

1. Top-down effects. Models seem capable of predicting top-down effects, i.e., mortality of prey does seem to change with predator abundance. We have not observed in historical data sets used for model testing that the predicted changes in mortality rates have somehow been cancelled out by abundance increases or switching behaviors of alternate predators. Hence, it appears that predation mortality rates in general are additive. An excellent example has been increases in shrimp stocks in the Atlantic in apparent response to reduced cod abundances (Worm and Myers, 2003). There is no clear indication (yet) that other species besides cod will replace the ‘natural’ predation impacts of cod so as to reduce shrimp production to its former lower levels.
2. Bottom-up effects. We do not often see clear evidence of strong bottom-up effects; it appears that where forage species are reduced by fishing, the predatory fish may shift to other prey. There are exceptions though, e.g., very large effects of fishing have been observed on piscivorous birds in the Peru current (see Tovar *et al.*, 1987) and in parts of the Northeast Atlantic, but we have also found that it is very difficult to replace the dominant prey species menhaden in the diets of the common predator striped bass in the Chesapeake Bay (Christensen *et al.*, MS). Predictions of bottom-up effects are extremely difficult due to the potential for switching and appearance of alternate food organisms combined with complex effects of trophic ontogeny. As an example, herring reduction fisheries in the Strait of Georgia in the 1960s were expected have a clear negative impact on Chinook and coho salmon, which feed on herring. The opposite happened; the reduced herring abundance was associated with strong salmon runs. It may well be that herring’s influence as a competitor with juvenile salmon (both feed on copepods and euphausiids) outweighed its role as a prey for the older salmon.
3. We have found it very troublesome to define the term ‘essential fish habitat’ when working with ecosystem models. Most fish undergo complex trophic and habitat use ontogenies and some display considerable capabilities to use alternative habitats when preferred habitats are no longer available or overpopulated. Such adaptations and flexibilities are virtually impossible to predict without empirical, experimental experience.

While there are a multiplicity of reasons for why it is problematic and uncertain to make predictions about future ecosystem structure and the potential impact of management interventions we really have no choice. Fortunately, there has been considerable progress in ecosystem modeling in recent years – some of it referred to in this contribution – and we conclude that it is becoming possible to develop useful predictive models for ecosystem management. We can with some credibility describe agents of mortality and trophic interdependencies, while evaluation of the relative impact of fisheries and environmental factors is progressing; we are at the ‘looking for correlation’-stage.

Overall, we can present our capabilities to use models to address ecosystem management questions in form of a popularized report card, see Table 2.

Table 2. A report card for using models to address ecosystem management questions

Concern	Grade	Comment
Bycatch impacts	A-	Prediction capabilities for direct effect of fishing in general is quite good
Top-down effects, (e.g., of predator culling or protection)	C	Trophic effects of fishing can be classified as ‘top-down’ or ‘bottom-up’ with respect to where management controls are exerted
• on valued prey	B	Changes in M for prey species already subject to assessment can be handled quite well
• on ‘rare’ prey	F	Prediction capabilities of outbreaks of previously rare species is wanting
Bottom-up effects, (e.g., effects of prey harvesting on predator stocks)	C	Uncertainty about flexibility of predators to find alternative food sources when prey are fished
Multiple stable states	B	‘Cultivation-dependensation’ mechanisms (Walters and Kitchell, 2001) may be main cause of ‘flips’
Habitat damage	D	Lack of understanding about real habitat dependencies, bottlenecks
Selective fishing practices / policies	F	Lack of experience in this area
Production regime changes	B	Models look good when fitted to data, but have not stood the test of time
Regime shifts	C	Appropriate policy adjustments in response to ecosystem-scale productivity change is unclear

Use of ecosystem models as part of the fisheries management process

If, as stated above, ecosystem models have reached a state where they can be of use to address fisheries policy questions, it may be worth examining the current state of affair with respect to actually using such models as part of the fisheries management process.

While there are a large number of scientists working with ecosystem models throughout the world, and while many do so with the explicit intention of using the models to address policy questions, we have only been able to identify very few cases where the models have actually been incorporated as part of the management process. What we see are cautious attempts to incorporate the models:

- Multispecies models
 - Estimating predation mortality for stock assessment, (ICES);
 - Limit harvest of prey species to meet predator demands;
 - Impact of changing mesh size, North Sea roundfish (ICES);
 - Minke whale and harp seal management, Norway
 - Environmental Impact Assessment (EIA), Alaska groundfish;
 - Target species response to TACs, Bering Sea.

- EwE models
 - Evaluate impact of shrimp trawling, Gulf of California (Arreguin-Sanchez);
 - Evaluate impact of bycatch, Gulf of California (Arreguin-Sanchez);
 - Evaluate impact of predators on shrimp, Gulf of Mexico (Arreguin-Sanchez);
 - Demonstrate ecological role of species, Gulf of Mexico (Arreguin-Sanchez);
 - Impact of proposed fisheries interventions, Namibia (Le Roux)
 - EIA of proposed fisheries interventions, Bering Sea;
 - EIA of alternative TAC's, Bering Sea and Gulf of Alaska;
 - Target species response to TACs, Bering Sea
 - Closed area sizing, Great Barrier Reef, Australia (Gribble).

We see the relative few examples of use of ecosystem models in the management process as indicative of (1) lack of experience with use of ecosystem models for predictive purposes and recognition that our capabilities to provide advice about large-scale dynamics is limited, and (2) software (like EwE) to make ecosystem data management relatively inexpensive has been available for less than a decade.

As important is that ecosystem modeling is designed and of use for strategic management, i.e. to address questions of what the impact is likely to be in the medium to long-term. As such the modeling supplements the tactical single-species assessment that currently dominates the management process; it does not and will not replace it. The fisheries management process is indeed trapped in short-term tactical approaches with strategic decisions only rarely being part of the agenda.

We argue that in the current situation where ecosystem manipulation is common, but based on a less than optimal set of decisions tools (often being done accidentally) it is time to implement strategic considerations as part of the management process. Such strategic considerations cry out for incorporating ecosystem models into the management process.

We recognize that there has been much apparent distrust of multispecies and ecosystem models for fisheries policy analysis. At least some of this distrust is due to concern about the lack of adequate data to quantify often complex predictions. Unfortunately, however, some of the objections also represent not real scientific concern but rather shortsighted assumptions about the objectives of management and unwillingness to take on a broader account of the impact of fishing, and a defensive approach by scientists who are comfortable with current management procedures and may lack the time, opportunity and/or capability to master the more complex steps involved in developing ecosystem-scale assessment models. Adding to the latter problem has been severe difficulties in developing ecosystem modeling software that is stable and flexible enough to be easily

adopted and used by assessment scientists for whom time is at a premium due to the demands of maintaining existing assessment and management machinery.

Ecosystem models as tools in adaptive policy design

All fisheries management policies are adaptive in the sense that when making policy choices we can never trust any single model prediction if for no other reason than the unpredictability of environmental effects. Instead we use the models to guide initial choices, and we expect to have to monitor over time in order to determine what the actual impact of each choice finally was. This requirement applies even more importantly to ecosystem management than it always has to single-species management. We are moving into new domains of policy choice involving new policy instruments such as protected areas, predator culling, and other management choices, which go beyond simple harvest regulation. Here, it will be critical to treat every test of such choices as an adaptive management experiment, and to design ‘treatment’ plans and monitoring programs using sound principles of experimental design in so far as practical (Walters, 1986). This may call for comparison of treatment of reference policies, temporal and spatial replication, as well as monitoring for unexpected and expected treatment responses.

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