Pitfalls and guidelines for "recycling" models for ecosystem-based fisheries management: evaluating model suitability for forage fish fisheries

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Ecosystem models have been developed for many marine systems to provide guidance on fisheries management strategies that protect key ecological functions. These models are commonly "recycled", i.e. applied to new questions or policy concerns after the initial phase of model development, testing, and application. Because decisions about the model structure are typically based on the intended model use, it is important to recognize limits in the capacity of models to address questions for which they were not specifically designed. Here, we evaluate existing foodweb models in the context of their ability to identify key forage species in foodwebs and to test management strategies for fisheries that target them. We find that the depth and breadth with which predator species are represented are commonly insufficient for evaluating sensitivities of predator populations to forage fish depletion. We demonstrate that aggregating predator species into functional groups creates bias in foodweb metrics such as connectance. Models also varied considerably with respect to the extent that they have been tuned or fitted to retrospective patterns and the degree to which key sensitivities are identified. We use this case study to provide several general recommendations when "recycling" ecosystem and foodweb models. Briefly, we suggest as routine procedure careful scrutiny of structural model attributes, of scales at which ecological processes are included, and quality of fits for key functional groups.

Keywords: ecological modelling, ecosystem-based management, forage fish.

Introduction

Fisheries management has evolved to consider multiple effects of fishing on the environment and the myriad environmental effects on stock dynamics. This movement towards ecosystem-based management has been fuelled, in part, by the awareness that multiple management objectives might be in conflict with each other due to limitations or constraints in the natural system that generates stock production (May et al., 1979; Walters et al., 2005). Thus, a central goal of ecosystem-based management is often to identify constraints, the trade-offs that they produce, then set harvest strategies that minimize the risk of adverse ecological outcomes (Pikitch et al., 2004; Link, 2010).

The movement towards ecosystem-based considerations in management is progressing despite a low level of understanding of underlying ecosystem processes, structures, and dynamics.

Some well-studied ecosystems have revealed strong top-down (Frank et al., 2005; Myers et al., 2007; Casini et al., 2008; Baum and Worm, 2009) effects of fisheries removing apex predators, whereas others have revealed strong bottom-up effects of changing abundance of small pelagic fish (Cury et al., 2000). However, these case studies are more the exception than the rule. Because of the absence of direct empirical observations that provide direct measures of interaction strengths, science to support ecosystem-based management often relies on ecological models. Models are used as strategic tools (e.g. to evaluate alternative management approaches, harvest control rules) or to identify the potential for trade-offs among fisheries management objectives (Walters and Martell, 2004). In some cases, end-to-end ecosystem models that consider foodweb dynamics and environmental forcing are used as operational models to perform management strategy evaluations (Fulton et al., 2011).

Ideally, each management question would motivate its own process of model development, testing, and evaluation to provide management advice and assess risks of alternative decisions. However, the needs for scientific advice have often outpaced the capacity for fisheries scientists to develop models for each new management question that has arisen. Consequently, complex foodweb and ecosystem models have been developed for marine ecosystems (Travers *et al.*, 2007) to address multiple scientific and management questions. Although these "multi-use" models are initially conceived to address specific sets of question, they are often "recycled" to provide guidance on other management and scientific questions. Though it is tempting to view models as multipurpose, no model is ideally suited for all questions (Starfield, 1997). That is, the decisions and simplifying assumptions that are made in the development of any model has direct bearing on the type of outputs and ecological processes that the model can represent. At the other end of the continuum are Models of Intermediate Complexity (MICE) developed around specific questions by purposefully omitting ecosystem components not relevant to the question at hand (Plagányi *et al.*, 2012).

Here, we evaluate limitations in recycling these multipurpose models to provide ecosystem-based advice for fisheries. We use as a highly relevant case study the problem of identifying the ecological effects of fishing small pelagic fish and krill, so called "forage fish". These species play critical roles in foodwebs providing a conduit of energy from small, low trophic level planktonic species to large, valuable species such as large fish, seabirds, cetaceans, and pinnipeds (Cury, 2000; Pikitch *et al.*, 2012). Consequently, fisheries for these species need to meet single-species management objectives while also protecting dependent predators, some of which may have a commercial value in other fisheries (Pikitch *et al.*, 2012). Consequently, seafood certification bodies such as the Marine Stewardship Council (MSC) have

specified additional certification requirements for these species so that fisheries do not have adverse ecological effects. The MSC requires an evaluation of whether a species is a "key" forage species, and if so, whether biomass targets and limits (i.e. the management strategy) are appropriate to protect dependent predators. We therefore used these two-stage criteria in asking whether existing models are well suited to identify species that have important ecological roles and for use in testing specific management strategies. We then use results from these specific questions to pose recommendations when "recycling" models to provide management advice in fisheries.

Methods

We selected a set of fisheries targeting forage stocks using three main selection criteria. The first was whether a stock is acknowledged as being an important forage species (e.g. Peruvian anchoveta, Barents Sea capelin, Chesapeake Bay Atlantic menhaden). The second was whether there is a large, commercially important fishery operating in the ecosystem (e.g. Gulf of Mexico menhaden, North Sea herring, California sardine). The third was whether stocks are currently certified by the MSC but entered the programme before recent modification of certification requirements for key low trophic level stocks. These are relevant because reassessment will likely involve the analysis of foodweb and ecosystem models. Together, this selection process provided a diverse collection of stocks to evaluate, spanning relatively data-rich and data-poor fisheries. In total, our review spanned 18 ecosystem models and included 27 forage stocks (Table 1).

We approached this problem by identifying desirable attributes of models with respect to evaluating ecosystem effects of forage fish fisheries. That is, we constructed a score sheet based on ideal attributes for robustly identifying key forage species and safe levels of depletion. We emphasize that many of these

Table 1. List of forage fish stocks and models used for evaluation.

Stock(s)	Model ecosystem (model type)	References
Atlanto-Scandian herring	Norwegian Sea and Barents Sea (E)	Dommasnes *et al.* (2001)
Argentinian anchovy	N. and C. Patagonia (MICE)	Koen-Alonso and Yodzis (2005)
SE Australia sardine	SE Australia Shelf (EwE); Great Australian Bight (EwE); Atlantis-SE (A)	Bulman *et al.* (2011), Goldsworthy *et al.* (2013), and Fulton *et al.* (2011, 2005)
Baltic Sea herring and sprat	Baltic Sea (EwE)	Harvey *et al.* (2003)
Barents Sea capelin	Barents Sea (EwE)	Blanchard *et al.* (2002)
California Current sardine	Northern California Current (EwE, A)	Field *et al.* (2006) and Kaplan *et al.* (2012)
Bay of Biscay sardine	Bay of Biscay (EwE)	Lassalle *et al.* (2011)
Canary Current sardine	Arguin Bank (E)	Sidi and Diop (2004)
Celtic Sea sprat and herring	Celtic Sea (EwE)	Guénette and Gascuel (2009)
Chesapeake Bay menhaden	Chesapeake Bay (EwE)	Christensen *et al.* (2009)
Gulf of California sardine and thread herring	N. Gulf of California (E,A)	Morales-Zarate *et al.* (2004) and Ainsworth *et al.* (2011)
Gulf of Mexico menhaden and bay anchovy	Gulf of Mexico (EwE)	Walters *et al.* (2008)
Humboldt Current anchovy	N. Humboldt Current (EwE,O)	Guénette *et al.* (2008), Marzloff *et al.* (2009), Tam *et al.* (2008), and Taylor *et al.* (2008)
Northeast Atlantic mackerel	North Sea (EwE)	Mackinson and Daskalov (2007)
North Sea sprat, herring, sandeel, mackerel	North Sea (EwE)	Mackinson and Daskalov (2007)
S. African sardine and anchovy	S. Benguela (EwE,O)	Shannon *et al.* (2008) and Shin *et al.* (2004)
Southern Ocean krill	Various (MICE)	Plagányi *et al.* (2012) and Watters *et al.* (2008, 2005)
W. English Channel sprat and herring	W. English Channel (EwE)	Araújo *et al.* (2005)

E, Ecopath; EwE, Ecopath with Ecosim; A, Atlantis; O, OSMOSE; MICE, Model of Intermediate Complexity.

models were not constructed specifically to address the issue of forage fish fisheries and their ecosystem effects. Thus, our scores against guidelines are not intended be critiques of the models *per se*, only an evaluation of their potential usefulness for addressing ecological effects of forage fisheries.

Below, we describe the scoring system and rationale for each scoring element, but the model scoring overview is listed in Table 2. The model scoring overview provides specific guideposts for evaluating the model adequacy for each scoring element. When there were multiple models for a stock and ecosystem, we report the highest scores looking over all available models. We also note that we attempted to be exhaustive, but we may not have identified all possible models for any given ecosystem and stock. We did not consider multispecies virtual population analyses, because these models generally do not include the bottom-up feedback of prey abundance on the growth or recruitment of predators.

Spatial and temporal scaling and data quality

Models are often built around periods when data are available or when there is funding to support modelling activities and based on spatial boundaries that match the extent of data collection or governance boundaries. If models are based on earlier periods and there have been documented ecological shifts since that time, they will be less useful for assessing forage stocks in present day conditions. Models based on spatial boundaries that do not match the range of the main forage stocks and their fisheries also may not be sufficient to identify all major predators and their resilience to forage fish fisheries. Foodweb models require extensive data inputs, and some data may be unavailable for any given ecosystem. In these cases, modellers often use data from nearby or similar ecosystems or use standardized diet information for particular groups.

Detail and breadth

All models contain simplifying assumptions that reduce the detail (the resolution at which ecosystem components are represented) and breadth (the range of ecological components that are included) that is present in the real world (Walters, 1986). Specific predator groups (seabirds, pinnipeds, odontocetes, piscivorous teleosts, and elasmobranchs) might be particularly sensitive to forage fish depletion (Pikitch *et al.*, 2012), so it is important that all main predators are represented in the models. Further, when models include multiple forage species, they would ideally be modelled as distinct state variables and not aggregated into a single-state variable. Similarly, ideal models would represent all predators as distinct state variables. Because the size structure of feeding is often important in dictating predator–prey dynamics, the ideal models would represent forage species and their predators as distinct age or size classes.

Dynamic simulations

Models are often used to simulate the effects of forage fish depletion on dependent predators because direct empirical evidence linking forage fish stocks to predator productivity is limited. Not all models were developed for dynamic simulation, however, and others may have been subjected to limited analysis. Clearly, a model should demonstrate the ability to hindcast past dynamics before being used to predict future dynamics. A model that uses many sources of data in fitting is desirable over a model that

Table 2. Overview of the scoring method used to evaluate models.

Spatial and temporal scale and trophic data	
Spatial coverage	Match to fishery?: 1, perfect match; 2, some overlap; 3, nearby or adjacent system; 4, no model match
Period	1, recent decade; 2, more than 10-year old but no known shift since then; 3, over 10-year old, known ecosystem shifts
Quality of trophic data (diet data)	1, diet data mostly from diet studies conducted in that region; 2, diet data are mostly from diet studies in nearby and similar regions; 3, diet data are largely from summaries or standardized diets
Depth and breadth	
Forage species detail (species, aggregated, species-by-size, or age)	1, by species with age structure; 2, by species no age structure; 3, some species and others aggregated; 4, potential key forage species aggregated; 5, no pelagic forage species
Predator detail (species, aggregated, species-by-size, or age)	1, by species with age structure; 2, by species no age structure; 3, some species and others predators aggregated; 4, most key LTL predators aggregated
Predator breadth (includes large pelagic fish, marine mammals, seabirds)	1, most predator guilds represented; 2, most represented but one major group omitted; 3, two or more major groups omitted
Dynamic simulation attributes	
Is model time dynamic?	Yes/no
Is model fitted to data?	Yes/no
Type of stock data	1, survey or stock assessment data for most commercially important stocks; 2, time-series of a few stocks, but they include forage fish and most main predators; 3, time-series of fish stocks, but not forage fish and/or main predators
Other data used in fitting?	Yes/no (list)
Fitting include dynamic environmental variables as inputs?	Yes/no
Quality of fit	1, good statistical fit; 2, reasonable fit but without statistical treatment; 3, poor fit
Account for uncertainty?	1, detailed treatment of parameter uncertainty, including data pedigree, alternative simulations, and sensitivity analysis; 2, parameter uncertainty is reported but not explicitly included in model runs; 3, no information on parameter uncertainty given
Represent local depletion?	Yes/no

uses only one data type. Further, environmental conditions are often important in driving population dynamics, so fitting exercises and simulations that include these drivers are ideal. We also assessed whether dynamic model fitting included environmental drivers as part of the analysis and which models could represent the localized depletion of forage species and subsequent ecological effects. This last point is important because the effect of forage fish fisheries may be localized due to the limited foraging range of central place foragers. Thus, an ideal model would incorporate spatial processes (either explicitly or implicitly) in evaluating impacts of forage fish fisheries. Because ecosystem models are usually constructed with incomplete information on structure and dynamics, it is essential that key uncertainties are identified, and the sensitivity of the model dynamics to these uncertainties is specified.

Because we are interested in the qualitative assessment of models, no formal statistical analysis of scores was conducted. In addition, the formal statistical theory for assessing ecosystem models is not well developed. Rather, we use the rank scores to identify model components that are most consistently flagged as being well-suited or ill-suited for forage fish ecosystem-based management evaluation. Further, the collection of stocks and models was not random, and the types of models that have been used are also not random across stocks and regions. For that reason, it was not possible to compare the relative performance of different types of models in a formal statistical manner.

Results
For all the 27 stocks, we identified at least one model that potentially could be used to assess forage fish fisheries. The most common model type was Ecopath with Ecosim ($n = 13$ models spanning 19 stocks). Atlantis ($n = 3$), OSMOSE ($n = 2$), and MICE ($n = 2$) were less common. For one stock, only an equilibrium Ecopath model was available.

Time/spatial matching and trophic data quality
In general, the time and spatial matching of models to stocks was good, and trophic data were available and representative of stocks in that ecosystem (Figure 1). For most stocks, there was a good match between the spatial scale of the stock and the ecosystem models that were available. Ten of the ecosystem models were deemed to be a very good match, six were deemed to have some degree of overlap, and none required the use of a model from an adjacent ecosystem. The temporal match of models was also generally good. Eleven models provided information about current state (structure, biomass, trophic connectivity) of ecosystems. However, in some cases, the information was derived from the forward projection of a dynamic foodweb model that was parameterized from data collected in earlier periods. For instance, the North Sea model (used for herring, sprat, sandeels, and two mackerel stocks) was parameterized based on biomass, consumption rates, and diet composition from the early 1990s, but was then simulated forward to present day conditions. Two models (spanning three stocks) were constructed on diet data from other systems or from "standardized" feeding habits of functional groups, but far more commonly trophic data inputs were derived from the ecosystem being modelled (Figure 1).

Model breadth and depth
Most models that we evaluated represented forage stocks at a species level, and several included more detailed representation

of age structure (Figure 2). Always the age structure consisted of a division between "juvenile" and "adult" stages, but none distinguished between larval and juvenile stages. Three models, however, had highly aggregated the representation of forage species that combined several stocks (Gulf of California, Canary Current, Argentina shelf).

Scores for predator breadth and detail tended to indicate limitations in models for assessing forage fisheries effects (Figure 2). Seven of the models did not include at least one main predator group (typically either seabirds or marine mammals). Moreover, predator species were often highly aggregated—only three models had predator groups modelled at the species level. Most commonly, seabird groups (combining piscivorous and planktivorous species) and marine mammals (combining pinnipeds and odontocetes) were aggregated into functional groups. Only a single model (California Current model) scored high on all these metrics.

Dynamic simulations, fitting, and uncertainty
Most available models were time dynamic and could therefore potentially be used to simulate consequences of alternative harvest policies for forage species (Table 3). Moreover, the vast majority of those models were tuned or fitted to some data time-series. Most ecosystem models employ *ad hoc* tuning and calibration techniques rather than more rigorous statistical approaches together with diagnostic tools for estimating parameters and evaluating model performance (Plagányi *et al.*, 2012). This impedes the ability to quantify the uncertainty of model results. Whole-of-ecosystem models that are rigorously fitted in a manner similar to that used in stock assessments are extremely rare (but see, e.g. Gaichas *et al.* 2011).

In the cases reviewed here, model fitting involved the time-series of abundance and catches for the majority of forage species and main predators included in the model. Few models used other types of data in fitting (e.g. time-series of mortality rates, diet composition, etc.). All but two models used some sort of dynamic environmental variables in the fitting process. Model fitting scores were highly variable, with six using statistical model fitting, five using qualitative fits, and two stocks had generally poor fits to data time-series. These two stocks were based on the same model (North Sea) which attempted to fit the time dynamics of 26 functional groups. In this case, a handful of functional groups could be fit, while several others produced simulated time dynamics that did not match data time-series. Eight of the 18 models had no assessment or explicit quantification of key uncertainties or sensitivities (e.g. through alternative model runs), and only three had a detailed treatment of uncertainty in model simulations.

Localized depletion
In general, existing models of ecosystems containing these forage species did not have the capacity to test for effects of localized depletion of forage species on predators. Models developed for the Southern Ocean krill fishery were the clear exception, where models explicitly considered spatial dynamics of fisheries, advection, and predation.

Implications of model structure for foodweb indicators
Models varied considerably with respect to the detail with which predators and forage species were represented; hence, we conducted a simple model experiment to determine the implications
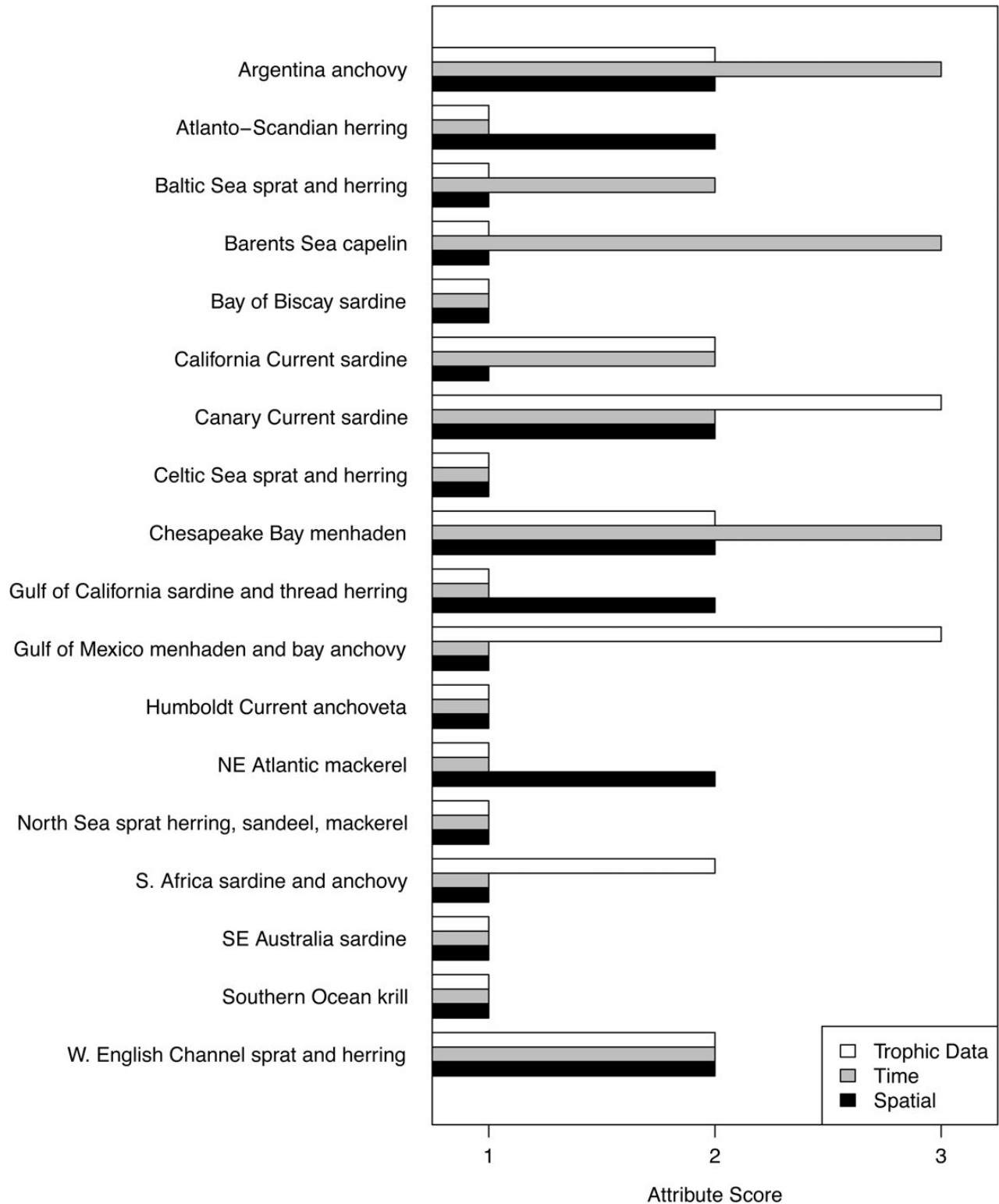
Figure 1. Summary of model evaluation scores (1, 2, or 3) on model temporal and spatial scales and the quality of trophic data. Each row depicts one or more stocks and the ecosystem model. Low scores indicate the best fit of the model to stock and best data quality (Table 2).

of varying this detail on foodweb metrics that have been proposed to identify low trophic level species that play especially critical roles in foodwebs. Specifically, Smith *et al*. (2011) identified two attributes of forage stocks that indicate their importance to the

ecosystem—the connectance (the proportion of foodweb links that include the forage species) and the consumer biomass ratio (the fraction of total consumer biomass that consists of a species). However, these measures might be sensitive to the
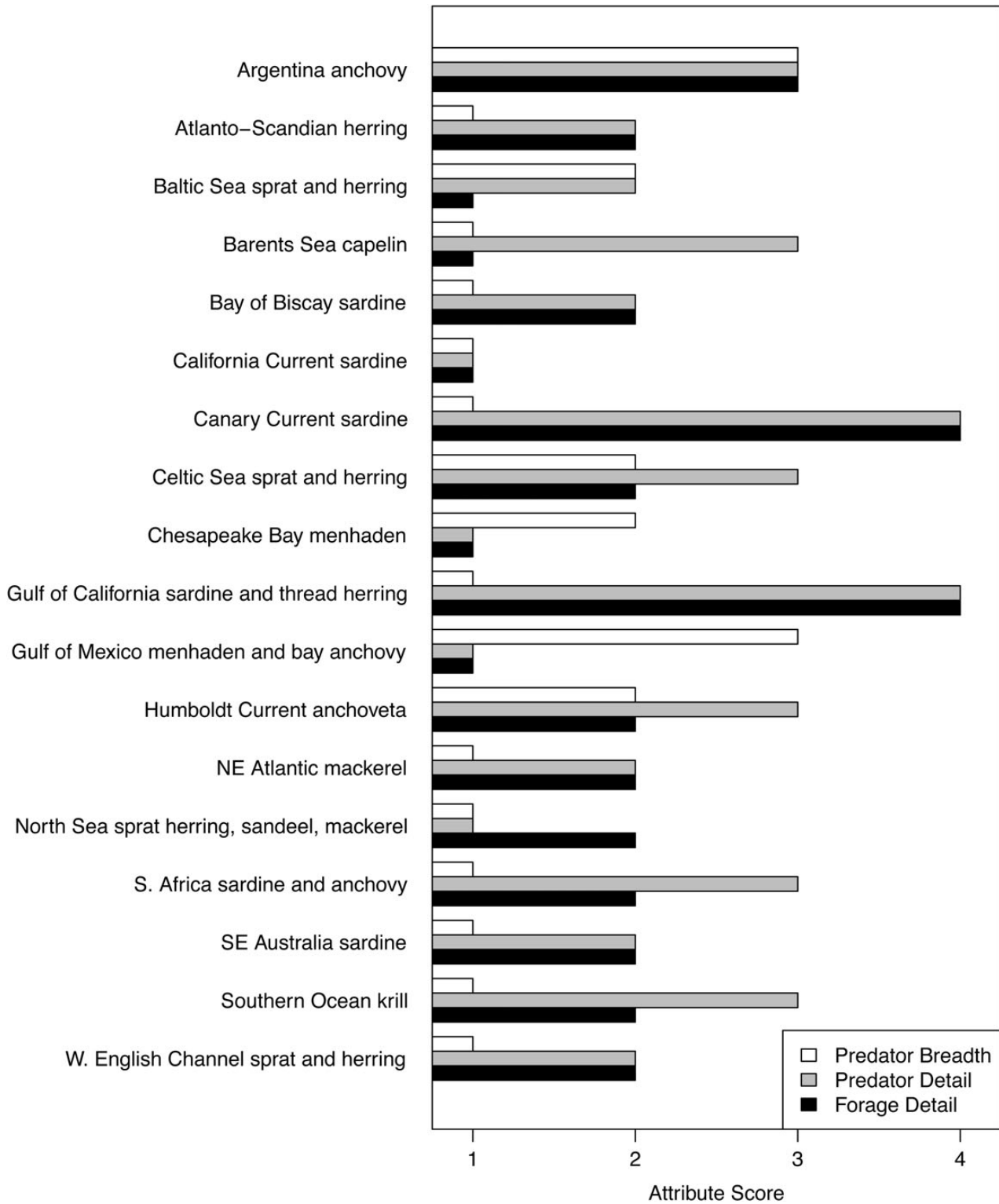
Figure 2. Summary of model structure scores. Predator breadth refers to the representation of main categories of predators (fish, birds, mammals), whereas detail refers to the specificity with which species are represented (Table 2)

decisions that modellers make about the level of aggregation (or detail) with which low trophic level species and their predators are represented. We took a highly detailed model (Northern California Current, Ecopath with Ecosim, and systematically

aggregated model components according to common types of aggregation decisions). For each, we calculated the connectance and the consumer biomass ratio for forage species stocks and drew comparisons across scenarios to determine the types of

Table 3. Summary of model fitting.

Stock	Time-dynamic?	Fitted to data?	Type of stock data	Other data used?	Dynamic Env. variables?	Quality of fit
Argentina Anchovy	Y	Y	1	N	N	1
Atlanto- Scandian Herring	N	–	–	–	–	–
SE Australia Sardine	Y	Y	2	N	Y	1
Baltic Sea Sprat and Herring	Y	Y	1	N	Y	1
Barents Sea Capelin	Y	N	–	–	–	–
Bay of Biscay Sardine	Y	N	–	–	–	–
California Current Sardine	Y	Y	1	N	Y	1
Canary Current Sardine	N	–	–	–	–	–
Celtic Sea sprat and herring	Y	Y	1	N	Y	2
Chesapeake Bay Menhaden	Y	Y	1	N	Y	2
Gulf of California sardine and thread herring	Y	N	–	–	–	–
Gulf of Mexico Menhaden and Bay Anchovy	Y	Y	1	Y	N	1
Humboldt Current Anchoveta	Y	Y	1	Y	Y	2
NE Atlantic Mackerel	Y	Y	1	N	Y	3
North Sea Sprat herring, sandeel, mackerel	Y	Y	1	N	Y	3
S. Africa Sardine and Anchovy	Y	Y	2	N	Y	2
Southern Ocean Krill	Y	Y	1	N	N	2
W. English Channel Sprat and Herring	Y	Y	2	N	Y	1

aggregations that have large effects on the determination of a stock as a key forage species. For reference, Smith *et al.* (2011) suggest that a connectance greater than 4% or the consumer biomass ratio greater than 5% is indicative of a key forage species.

The "base model" indicated that euphausiids and the aggregated "forage fish" group (comprising of herring, eulachon, and N. anchovy) are likely key forage species in this foodweb, based on the high connectance and the high consumer biomass ratio. Sardine and mackerels have connectance and consumer biomass values that are below the threshold values (Figure 3a). When seabird and marine mammal species are aggregated (seabirds as a single functional group, marine mammals into pinnipeds, odontocetes, and baleen whales), there was a modest effect on the connectance scores in general (as indicated by the overall shift in the average connectance of all species), but a mixed effect on potential key forage stocks (Figure 3b). Larger effects were seen when fish predators were aggregated—in addition to the aggregation above, we aggregated some groundfish species into "piscivorous demersal", "rockfish", and "other demersal". This led to all but sardines exceeding the connectance threshold, but had no effect on the consumer biomass ratio. At this level of aggregation, most model groups have a connectance greater than 4%. When we then aggregated forage species (e.g. into "small pelagics"), this tended to increase the overall connectance and the consumer biomass ratio (Figure 3d). At this point, all but one species group in the ecosystem had connectance values that exceed the threshold.

Discussion

Ecological models are used for a variety of purposes, from generating predictions that are used directly in setting harvest guidelines, scenario evaluation, and generating strategic management advice to identifying key uncertainties in our understanding of functional relationships among ecosystem components. The familiar adage "all models are wrong, some models are useful" (Box and Draper, 1987) reminds us that we evaluate models by how useful they are with respect to their intended purposes (Walters, 1986; Starfield, 1997). Thus, our evaluation of models here are in no way intended to be a reflection on their quality, particularly with respect to their intended purpose. Still, there is an increasing trend towards using previously developed models for new purposes, quite possibly those not anticipated by the model developers. The present study examines one example of this type of activity, with the intention of revealing pitfalls, cautions, and caveats that might generally apply. Our study builds on previous studies that have reviewed best practices for constructing ecosystem models (Hill *et al.*, 2007; FAO, 2008) or provide an overview of desirable attributes of ecosystem models, specifically MICE, which are intended for use as ecosystem assessment tools (Plagányi *et al.*, 2012).

This work provides one of the first synthetic and direct comparisons of foodweb models against objective scoring criteria that were based on a single application—ecosystem-based management of forage species. That many of the models that were deficient in one or more of our scoring criteria was not unexpected. Most of these models were not developed to specifically address questions about forage fish fisheries and the evaluation of fishing management. The sole exceptions to this generality were models for Southern Ocean krill, their fisheries, and their predators. This system has been a highly influential case study in ecosystem-based management, beginning with the classic work of May *et al.* (1979) and the subsequent formation of the Convention for the Conservation of Marine Living Resources (CCAMLR). Our review indicates that, in general, pre-existing, "recycled" models will have likely one or more limitations that warrant attention if they are to be used to evaluate the management of forage fish fisheries.

We identified at least two main issues that warrant consideration in using "recycled" foodweb and ecosystem models for prediction or management strategy evaluation. The first is structural—does the model have the appropriate model breadth and depth (*sensu* Walters, 1986) for the question at hand? For the evaluation of ecosystem effects of forage fish fisheries, standard rules of thumb useful for exploring dynamics of foodwebs within a
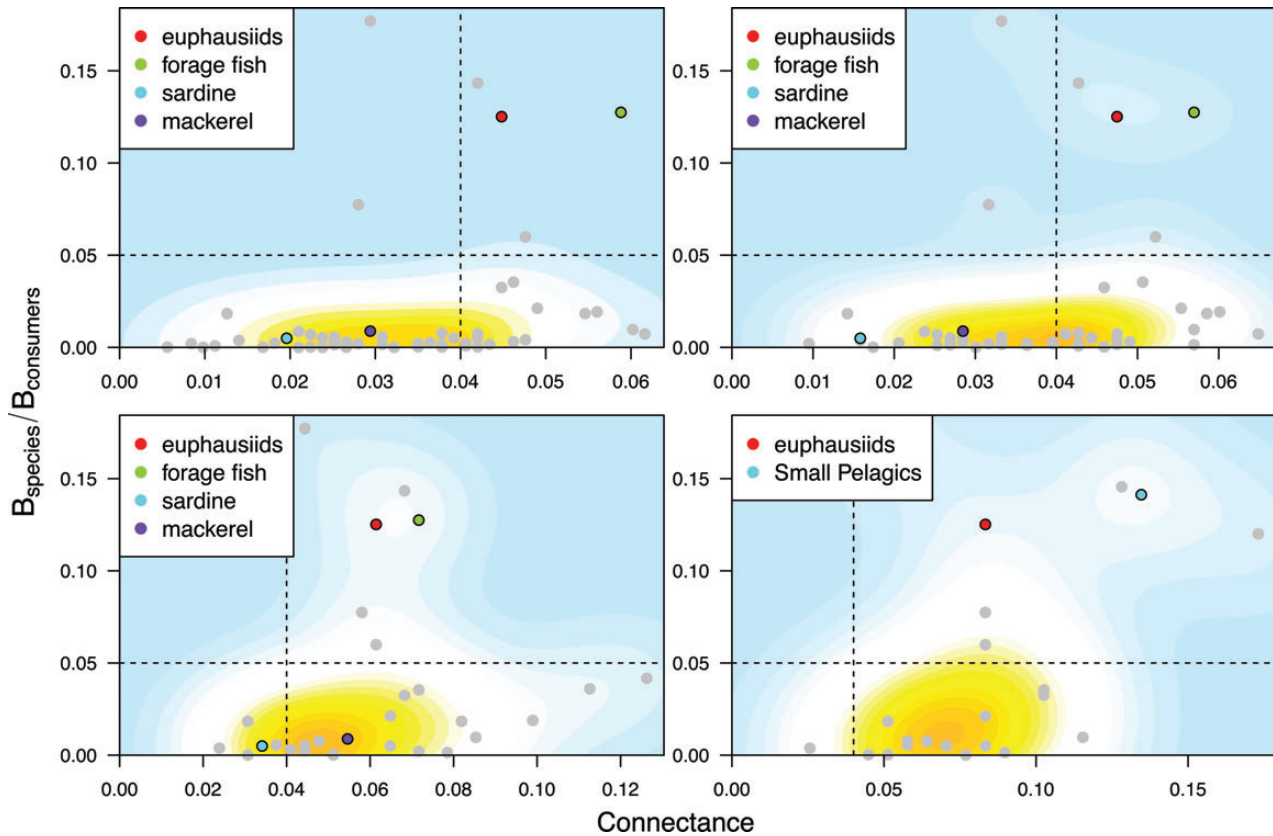
Figure 3. Effects of the sequential aggregation of species into functional groups on indicators of forage species "keyness", applied to the California Current Ecosim model. (a) The base model of the California Current has only two small pelagic groups that have consumer biomass ratios or connectance scores that indicate that they are "key" species. (b). Seabirds, odontocetes, and pinniped species are each aggregated into single functional groups. (c) As in (b), but groundfish are aggregated into three main functional groups. (d) As in (c), but all small pelagic fish are aggregated into a single functional group. Grey lines indicate scores for other species, and background colour depicts a two-dimensional kernel density smoother.

broader ecosystem context (Fulton *et al.*, 2003) may not apply when the focus is on protecting species. That is, by aggregating several predator species, the models essentially predict the response of an "average" predator, whereas the need is to predict the response of the most sensitive predator. These model breadth and depth issues are also highly relevant to applying criteria intended to identify key forage species. Parameters such as connectance are highly sensitive to the detail with which model groups are represented.

The second issue is that models varied considerably with respect to the quality of fits to retrospective data and in the degree to which key sensitivities were indicated. It is not surprising that models might fail to capture historical ecosystem behaviour. For instance, a long-standing axiom in marine fisheries ecology is that population variability is driven by recruitment variation, which in turn is driven by environmental conditions during a critical time window early in life (Houde, 1994; Cushing, 1996). Many models represent recruitment from food availability and predator density, but few do so on the spatial or temporal scale at which the critical period may occur. Moreover, there is still great uncertainty about the nature of functional responses in marine ecosystems (Hunsicker *et al.*, 2011) and elsewhere (Abrams and Ginzburg, 2000). Lastly, model fitting and tuning often involves the adjustment of a subset of all model parameters, so the full uncertainties

in model predictions are not always well known. Indeed, few model publications specifically included alternative model runs that attempted to bracket uncertainty caused by structural and parameter uncertainty.

Based on these common issues, we provide the following sets of guidelines for recycling foodweb models to answer new questions for fisheries management. The first is to ask whether the model structure is appropriate for the question at hand. Simple rules of thumb might include (i) does the model represent the key species of management interest in sufficient detail? (ii) Are all main interacting species included in the model? (iii) Does the model represent a period that is relevant to the current management question? (iv) Can the model represent ecological processes at spatial and temporal scales appropriate for the current question? The second is to ask whether the model has been demonstrated to adequately capture the key features in population trends of species that matter for the management question. That is, it is unnecessary to demand that the model adequately capture all ecosystem dynamics, only those relevant to the current purpose. The third is to ask on what parameters or specifications are models particularly sensitive and to then run alternative model scenarios to bracket the uncertainty caused by these sensitivities. The fourth is to ask whether there are multiple (independent) models available for the question, and if so, do they generally produce consistent

predictions on management strategies (Hollowed *et al.*, 2011)? If models are found to lack any of these rules of thumb, it may be possible to revise them to suit the new purpose. This structured framework for evaluating "recycled" models may provide a useful guide for identifying specific components of models that need revision.

Foodweb models will remain a vital tool for providing ecosystem-based management advice. They provide a way to explore fisheries management strategies, identify potential unexpected outcomes of management decisions, and characterize trade-offs among ecological objectives. In the context of forage fish fisheries management, they are increasingly used to define harvest control rules that can best protect dependent predators. Moreover, they can refine tactical issues in fisheries management, such as improving estimates of biological reference points. For instance, these foodweb models might help better inform "unfished biomass" by reconstructing productivity that would be realized with "pristine" predator population levels or within alternative environmental regimes (Tyrrell *et al.*, 2011; Plaganyi and Butterworth, 2012). Although many existing models may have limitations that raise cautions about their use and interpretation, transparent, and open acknowledgement of these limitations should lead to better inferences and ultimately more robust fisheries management strategies.

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