Fisheries and Oceans Canada Pêches et Océans Canada
Science Sciences

C S A S **S C C S**

Canadian Science Advisory Secretariat **Secrétariat canadien de consultation scientifique**

1Research Document 2012/060 **Document de recherche 2012/060**

Maritimes Region **Région des Martimes**

Multi-species interactions: Effects of multi-species harvesting on single species harvest rates in NAFO Division 4X

Interactions entre les multiples espèces : effets de la pêche de multiples espèces sur les taux d'exploitation d'une seule espèce de la division 4X de l'OPANO

A. Bundy, J.N. Araújo, and S. Guénette

Fisheries and Oceans Canada, Maritimes Region
Population Ecology Division
Bedford Institute of Oceanography
1 Challenger Drive
Dartmouth, Nova Scotia
Canada B2Y 4A2

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

La présente série documente les fondements scientifiques des évaluations des ressources et des écosystèmes aquatiques du Canada. Elle traite des problèmes courants selon les échéanciers dictés. Les documents qu'elle contient ne doivent pas être considérés comme des énoncés définitifs sur les sujets traités, mais plutôt comme des rapports d'étape sur les études en cours.

Research documents are produced in the official language in which they are provided to the Secretariat.

Les documents de recherche sont publiés dans la langue officielle utilisée dans le manuscrit envoyé au Secrétariat.

This document is available on the Internet at

Ce document est disponible sur l'Internet à
www.dfo-mpo.gc.ca/csas-sccs

ISSN 1499-3848 (Printed / Imprimé)
ISSN 1919-5044 (Online / En ligne)
© Her Majesty the Queen in Right of Canada, 2012
© Sa Majesté la Reine du Chef du Canada, 2012

Canada

TABLE OF CONTENT

Correct citation for this publication:

Bundy, A., Araújo, J.N., and Guénette, S. 2012. Multi-species interactions: Effects of multi-species harvesting on single species harvest rates in NAFO Division 4X. DFO Can. Sci. Advis. Sec. Res. Doc. 2012/060. iv + 26 p.

ABSTRACT

Canada was one of the first nations to adopt an “ecosystem approach” to oceans management, and to this end, has largely focussed on an objectives and ecosystem indicator approach. Ecosystem modelling is a complimentary approach which, until recently, has received little direct support within Fisheries and Oceans Canada. There are many advantages to using ecosystem modelling for an Ecosystem Approach to Management, and the range of established modelling approaches vary from extended single species models to multi-species minimum realistic models, food web models to whole ecosystem models with age and spatial structure. Theme III of the Maritimes Region Gulf of Maine Research Initiative explored the “Quantification of the impact of ecosystem interactions on harvest rates and dynamics of commercially targeted (and non-targeted) species” using a range of ecosystem models. Prior to this work, no multi-species or ecosystem models had been developed for Northwest Atlantic Fisheries Organization Divisions 4X/5Y. This research document presents the results of three ecosystem models of different structure, complexity and levels of aggregation, and explores how they may be used for science advice in support of fisheries management. It provides preliminary estimates of (i) single species fisheries reference points; (ii) aggregate species reference points; (iii) multi-species reference points; (iv) aggregate system reference points; and (v) the level of predation and fishing mortality on key species. These results, and others, indicate that it would be foolish to ignore the broader ecosystem in the science advice provided to fisheries management, particularly in the face of increased environmental change.

RÉSUMÉ

Le Canada a été l'un des premiers pays à adopter une approche écosystémique pour la gestion des océans et, à cette fin, il s'est concentré en grande partie sur une approche fondée sur des objectifs et des indicateurs écosystémiques. La modélisation écosystémique est une approche complémentaire qui, jusqu'à tout récemment, n'a reçu que très peu de soutien direct de Pêches et Océans Canada. Il existe de nombreux avantages associés à l'utilisation de la modélisation écosystémique dans le cadre d'une approche écosystémique pour la gestion. Les approches de modélisation varient de modèles élargis sur une seule espèce à des modèles réalistes minimaux portant sur de multiples espèces, et de modèles sur la chaîne alimentaire à des modèles sur des écosystèmes entiers comprenant la structure spatiale et la structure d'âges. Le thème III de l'initiative de recherche écosystémique du golfe du Maine de la région des Maritimes portait sur la quantification de l'impact des interactions écosystémiques sur les taux d'exploitation et la dynamique des espèces visées par la pêche commerciale et des autres espèces à l'aide d'une variété de modèles écosystémiques. Avant ces travaux, il n'existait aucun modèle écosystémique ou concernant de multiples espèces pour les divisions 4X/5Y de l'Organisation des pêches de l'Atlantique Nord-Ouest. Le présent document de recherche présente les résultats de trois modèles écosystémiques dont la structure, la complexité et les niveaux d'agrégation sont différents et explique comment ils peuvent être utilisés dans le cadre d'avis scientifiques à l'appui de la gestion des pêches. Il fournit également des estimations préliminaires des (i) points de référence pour une seule espèce; des (ii) points de référence pour des espèces regroupés; des (iii) points de référence pour de multiples espèces; des (iv) points de référence regroupés de système et (iv) une comparaison du niveau de prédation et du taux de mortalité par pêche associés à des espèces clés. Ces résultats ainsi que d'autres indiquent qu'il serait ridicule de ne pas tenir compte de l'ensemble de l'écosystème dans les avis scientifiques fournis pour la gestion des pêches, surtout en présence de changements environnementaux accrus.

INTRODUCTION

Canada's east coast oceans have been exploited by commercial fisheries for over 500 years, and although there are still active fisheries, most exploited stocks are mere shadows of their former selves. Fisheries are a major driver of change (Shackell et al. 2012) altering properties of both fished and unfished species, through direct and indirect effects. Direct effects can include reductions in population abundance, age and size structure, biodiversity, community composition and habitat destruction. Indirect effects, including incidental mortality, are transmitted through the ecosystem through trophic interactions and competition, and may result in increased or decreased abundance of prey or predator species, altering community composition. In recognition of the complex, interconnected nature of marine ecosystems, ecosystem approaches have been promoted as a way to improve fisheries assessment and management (Jamieson and O'Boyle 2001; Browman and Stergiou 2004, 2005; Pikitch et al. 2004).

Canada was one of the first nations to adopt an "ecosystem approach" to oceans management (EAM; Curran et al. 2012). Much has been written about the apparent complexities and difficulties of implementing an ecosystem approach (e.g., Murawski 2007; Rice 2011), but arguably it is underway in a number of nations (Pitcher et al. 2009; Link et al. 2011). The Canadian federal government, through Fisheries and Oceans Canada (DFO), leads the Canadian initiative into EAM. To date, DFO's main focus has been on an ecosystem objective and indicator approach (see review in Curran et al. 2012; Jamieson and O'Boyle 2001), which has been supported scientifically through the Oceans Action Plan. Ecosystem modelling also has much to offer EAM (e.g., Link and Bundy 2012; Link et al. 2011; Gaichas et al. 2012a), particularly from a strategic and conceptual perspective. Until recently, ecosystem modelling has not been given great emphasis within DFO. However, a 2007 National Ecosystem Modeling workshop reviewed a range of ecosystem modelling approaches (DFO 2008). More recently, DFO's regional Ecosystem Research Initiative (ERI) program has supported ecosystem modelling work in most of its regional programs. In the Maritimes Region, Theme III of the Gulf of Maine Research Initiative explored the "Quantification of the impact of ecosystem interactions on harvest rates and dynamics of commercially targeted (and non-targeted) species" using ecosystem models. Prior to this work, no multi-species or ecosystem models had been developed for Northwest Atlantic Fisheries Organization (NAFO) Divisions 4X/5Y.

There are many advantages to using ecosystem modelling for EAM. The obvious is that it goes beyond estimates of fishing mortality and stock abundance and provides a more holistic perspective of fisheries productivity by including tropho-dynamic interactions and, in some cases, environmental drivers. One potential benefit of this approach is improved estimates of fisheries reference points (Overholtz et al. 2008; Tyrrell et al. 2011; Gaichas et al. 2012a) and of population trajectories under various future management and climatic scenarios (Holsman et al. 2012). It has long been recognised that interactions among species can affect management references points such as maximum sustainable yield (MSY) or the biomass that supports MSY (B_{MSY}), and that an ecosystem cannot support all single species MSY's simultaneously (Walters et al. 2005; Mackinson et al. 2009; Worm et al. 2009; Gaichas et al. 2012b). Further, the widespread application of single-species MSY policies can lead to severe deterioration in ecosystem structure, including the loss of top predator species (Walters et al. 2005, 2009).

There is a wide-range of multi-species and ecosystem modelling approaches (see Plagányi 2007; DFO 2008; and Townsend et al. 2008 for reviews). They range from extended single species models to multi-species minimum realistic models, food web models to whole ecosystem models with age and spatial structure. All models have strengths and weaknesses: simplicity may entail missing key processes, whereas complexity requires more data, time and resources. These models are most widely used to explore conceptual or strategic questions and to provide advice at these levels. Conceptual models further our understanding of the way that things work, i.e., they address process-driven questions; strategic advice is linked to policy goals and tends to be long-ranged and broad based; tactical advice is shorter term and familiar in the stock assessment world where quotas may be set on a daily to multi-year basis, depending on the stock and management requirements. Most multi-species/ecosystem models are not yet suitable for tactical advice, although they can be used to set single species advice in an ecosystem context. For example, estimates of predation mortality from multi-species models are used in single species assessment models in the North Sea (Sparholt 1990). Multi-species and ecosystem modelling approaches used by DFO are reviewed in DFO (2008). Link et al. (2011) reviewed ecosystem modelling in the Northwest (NW) Atlantic, and Link and Bundy (2012) more specifically reviewed modelling approaches in the Gulf of Maine Area. In eastern Canada, much of the recent ecosystem modelling work has focussed on untangling the seal-cod story using models ranging from two-species models to whole ecosystem models (DFO 2011).

The objectives of Theme III of the Maritimes ERI were quite broad: this document will focus on the main question "Quantification of the impact of ecosystem interactions on harvest rates" in the NAFO Divisions 4X/5Y (Figure 1). In single species stock assessments (e.g, cod or haddock), 4X is usually treated as a single ecosystem (although the catch at age data are developed for the Scotian Shelf and Bay of Fundy areas of 4X separately and then combined into one assessment). In order to make the ERI ecosystem modelling relevant for management purposes, this same scale was used. However, this area includes the western portion of the Scotian Shelf and the Bay of Fundy (BoF) and, although the two areas have a very similar species composition, their dynamics are quite different. The western Scotian Shelf (WSS) is a wide continental shelf area influenced by currents from Labrador and the Gulf of St. Lawrence and is considered part of the Scotian Shelf Large Marine Ecosystem (LME). This LME is limited in the north by the Laurentian Channel and in the south by the Fundian Channel and has a complex topography consisting of shallow banks and basins (Sherman and Hempel 2008, and references therein). The western part of this LME is characterized by warmer waters than the eastern part (Zwanenburg et al. 2002), it has a similar demersal fish fauna (Mahon and Smith 1989; Shackell and Frank 2003; Zwanenburg et al. 2002), but higher growth rates and productivity than the east (Frank et al. 2006; Shackell et al. 2010). The main characteristic of the Bay of Fundy is the magnitude of its tides, which generate intense vertical mixing caused by bottom turbulence and generate high levels of marine productivity and is considered part of the Northeast U.S. LME (Sherman and Hempel 2008).

Alongside the adoption of an ecosystem approach, DFO has committed to using a precautionary approach (PA) when managing fisheries, and is currently in the process of defining single species reference points. However, at the World Summit on Sustainable Development in 2002 (WSSD), Canada also committed to achieving the maintenance or restoration of depleted fish stocks to levels that can produce MSY[1]. Currently, MSY is not widely used for management purposes in Canada and is not currently incorporated into the PA framework. In this document, this question is explored, using three ecosystem models of different structure, complexity and levels of aggregation, and estimates of (i) single species fisheries reference points; (ii) aggregate species reference points; (iii) multi-species reference points; (iv) aggregate system reference points; and (v) the level of predation on key species compared to estimates of fishing mortality. Finally, the effects of the environment on these estimates is explored.

[1] MSY is the long term theoretical catch which is greater than the actual possible constant catch – see Beddington and May (1977) and Sissenwine (1978) for more details.

This research document was presented as a working paper at the DFO Maritimes Centre for Science Advice regional workshop entitled 'Ecosystem Research Initiative (ERI) Synthesis: How can Ecosystem Research Initiative Results be Incorporated into Management Processes and Advice?', held in Dartmouth, Nova Scotia, during October 25-27, 2011.

METHODS

Three multi-species/ecosystem models that span the range of model types described above were used to explore these questions: Multi-Species Virtual Population Analysis (MSVPA), Stock Production Models and Ecopath with Ecosim (EwE). The MSVPA and EwE modelling were the key elements of Theme III of the Maritimes ERI; the Stock Production modelling results are the products of an international collaboration between the U.S., Norway and Canada, largely funded by the U.S. Comparative Analysis of Marine Ecosystems (CAMEO) programme (Link et al. 2010a; Gaichas et al. 2012a; Link et al. 2012).

MODELS

Multi-Species Virtual Population Analysis (MSVPA)

The ERI used a simple multi-species virtual population analysis (MSVPA) centered on herring to quantify the level of predation on herring compared to fishing, and estimate natural mortality at age (Guénette and Stephenson 2012). MSVPA is a multi-species extension of the single species VPA. In a single species VPA, natural mortality, which includes predation mortality, is *a priori* usually assumed to be 0.2; in MSVPA, predation mortality at age "a" (M2) is calculated based on the relative abundance of predators, their consumption rate and the suitability of the prey "a" as prey, and the suitability of other prey groups. Guénette and Stephenson (2012) used a Type II functional response, as described by Magnússon (1995) with respect to its use in MSVPA. This means that as prey 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 prey "a" in the diet of a given predator depends on the abundance of the prey "a" at the moment. As prey "a" abundance decreases, it is assumed the predator will compensate by obtaining a larger portion of its diet from other prey. MSVPA requires more data than stock production models (see below), including a value for the natural mortality (non-predation; assumed here to be M1= 0.01), an estimate for fishing mortality in the last year considered in the analysis (terminal F), and an abundance index for all groups, suitability estimates, weight-at-age, predator ration estimates and diet data. In this model, the abundance of the predators is fixed and it is assumed that there is enough prey for all predators. For further information see Guénette and Stephenson (2012).

Stock Production Models

Production models require only catch and biomass data and, thus, are widely applicable at multiple levels of aggregation (i.e., single species, functional group, aggregate system). Essentially they relate the production of a population to current population size, given an intrinsic rate of productivity and a finite carrying capacity to account for density-dependent effects. As for all models, they are reductionist representations of complex processes and though there has been some debate about their utility in specific applications (Mohn 1980; Ludwig and Walters 1985, 1989; Punt 2003), there is consensus that they play a useful and important role in ecology in general (Mangel 2006) and fisheries science in particular (Ludwig and Walters 1985, 1989; NRC 1998). In addition to their simplicity, production models are useful because: 1) they are robust to various assumptions; 2) they produce standard outputs that are readily comparable; 3) they can be scaled to different spatial and organizational levels; and 4) they can incorporate drivers as covariates. In addition, the outputs of these models can be readily related to commonly used fishery management reference points such as MSY and B_{MSY} or fishing mortality rate corresponding to MSY (F_{MSY}) (Restrepo et al. 1999; Mueter and Megrey 2006). Since data requirements are simple, production models can be applied to target and non-target species and are, thus, capable of providing reference points for non-target species, for functional groups, and for ecosystems. Here a simple linear regression model was applied to cod, herring (Holsman et al. 2012), various functional groups (Lucey et al. 2012) and the aggregate system (Bundy et al. 2012*)* across 13 northern hemisphere ecosystems: the results for NAFO Division 4X are presented here. For further details of the methodology see Mueter and Megrey (2006) or Bundy et al. (2012).

Ecopath with Ecosim (EwE)

EwE has been used widely to quantitatively describe aquatic systems and to explore the ecosystem impacts of fishing (Christensen and Walters 2004; Bundy et al. 2009; Coll et al. 2009). EwE is a modelling framework based on assumptions of mass balance and a system of linear equations describing the average flows of mass and energy between functional groups (Christensen et al. 2005). In the Ecopath module, a base-model for the system is parameterized for a specific period of time (a snapshot). Ecosim is the time-dynamic module of the package that uses parameters from the Ecopath base model as starting point for the dynamic simulations.

The “mass balance” term in EwE means that the model parameters are under the physical constraint that the total flow of mass into a functional group must equal the total flow out of that group. Basic parameter requirements for each functional group include biomass, total mortality, production to biomass ratio, consumption to biomass ratio, biomass accumulation rate, total fishery catch and diet composition. Functional groups can be represented either by aggregate (single) biomass pools or by age/size structured (multi-stanzas) components. Predation and energy transfer in Ecosim is modelled using a “foraging arena hypothesis” where only part of the prey population prey (*i*) is vulnerable to predation by predator (j) at any one time. The flow rate between the two pools is called “vulnerability”, v_{ij}, and is the product of a scaling factor, (k_{ij}) and the ratio of Ecopath base consumption of species (*i*) by species (j) to Ecopath base biomass of species (*i*): $v_{ij} = k_{ij} * Q_{ij,base}/B_{i,base}$. Essentially, *vij* represents the maximum mortality that can be inflicted on prey *i* in the presence of infinite biomass of predator *j*. The parameter *kij*, determines the maximum *Qij*, and is a user-defined input to Ecosim and can vary from 1 to ∞, with a default value of 2. High values result in top-down Lotka-Volterra predator-prey dynamics, low values cause bottom-up control (Walters and Martell 2004; Christensen et al. 2005; Ahrens et al. 2011; Araújo and Bundy 2012).

EwE models have been developed for many of the shelf ecosystems in the NW Atlantic (e.g., Bundy 2001, 2005; Link et al. 2008), but there are no EwE models for NAFO Division 4X (Figure 1). See Araújo and Bundy (2011) for further details of the development of the 4X EwE model.

Araújo and Bundy (2012) analysed the relative roles of fishing, trophic interactions and bio-physical drivers on system dynamics and productivity using a procedure similar to Shannon et al. (2004) and Araújo et al. (2006). The objective was to calculate the total variation explained by the model, using a no-effects model scenario as the base for comparison, and to calculate the relative importance of fishing, tropho-dynamics and primary productivity (a proxy for bio-physical drivers). The Ecosim non-linear tool was used to fit the model biomass estimates to the observed data by estimating the vulnerability parameters for the species interactions (flow rates;

k_{ij}) and a primary production anomaly forcing. The sum of squared deviations (SS) of log observed biomass from log predicted biomass was then recorded from the following scenarios: (i) a no-effects model scenario or baseline situation, in which the model estimates were forced to remain constant at the baseline values; (ii) including all effects (the estimated k_{ij} + fishing + estimated anomaly); (iii) including the effects of the estimated k_{ij} + fishing only (no-anomaly); (iv) fishing only; and (v) k_{ij} only and (vi) anomaly only. The significance of including the anomaly forcing (scenario ii) was tested using the scenario (iii) as the baseline for comparison.

Once trophic flow parameters (k_{ij}) were estimated through time series fitting, MSY and multi-species MSY were estimated using the equilibrium routine in Ecosim. Single species MSY are estimated by running Ecosim to equilibrium using long simulations (100+ years) and increasing and decreasing F incrementally from base values. The biomass of all other species is held constant. The multi-species MSY is estimated by running a long simulation using the single species F_{MSY} for each species simultaneously. See Christensen et al. (2005) for further details.

DATA

For detailed descriptions of the data sources see Guénette and Stephenson (2012) and Araújo and Bundy (2011). The Production models used the same data sources as Araújo and Bundy (2011).

RESULTS

MULTI-SPECIES VIRTUAL POPULATION ANALYSIS (MSVPA)

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 approximately 1.2 million tonnes (t) in 1987 (Figure 2). 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, although some large pulses have been observed (Figure 2).

According to the MSVPA base scenario, predators removed an estimated 130,000 t per year during the period 2000-2006, two-fold less than 1970-1975 when removals 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 (Figure 2). During the same period, the rate of fishing mortality for 5+ herring was predicted to have increased from 0.45 in the 1970s to 0.54 in the 2000s (Figure 2C). 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 particularly for herring of ages 2 and 3 (Figure 2D). 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.

Uncertainty in model estimates was explored under two scenarios that assumed that consumption of herring was either under (minus50) or over (plus50) estimated by 50% for major predators. They predicted a corresponding 48% change in the estimation of herring consumed (Figure 3), and a 25% change in herring population. It is worth noting that the predicted herring consumption with scenario minus 50 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 increasing trend in dogfish biomass, this scenario was the only one resulting in an increasing trend over time for herring biomass, consumption and mortality.

PRODUCTION MODELS

Two single species production models were developed for cod and herring in 4X (Holsman et al. 2012). The biomass of both species decreased across the time period, with a peak in the mid to late 1980s for herring, and the late 1970s for cod (Figure 4a). The Schaefer model provided a good fit to herring Annual Surplus Production (ASP) dynamics (Figure 4b), but not for cod (not shown). The herring model including sea surface temperature (SST), wind speed and cod as a predator provided the best fit (Figure 4c). Positive anomalies in SST were positively correlated with surplus production. The estimated surplus production curve from this model predicted an MSY of 1.97 $t.km^{-2}$, B_{MSY} 10.86 $t.km^{-2}$ and F_{MSY} = 0.18. Note that the model does not fit the data terribly well in the middle years.

Lucey et al. (2012) conducted a similar analysis at the functional group level (without environmental covariates), and estimated MSY, B_{MSY} and F_{MSY} for each functional group where F_{MSY} is estimated as MSY/B_{MSY}, which is equivalent to the r/2 used by Holsman et al. (2012). Lucey et al.'s estimates for their planktivore group (Table 1) and the pelagic group are comparable to Holsman et al.'s estimates in Table 3.

At the aggregate level (all commercial species aggregated together), Bundy et al. (2012) estimated an MSY of 5.38 $t.km^{-2}$, B_{MSY} 11 $t.km^{-2}$ and F_{MSY} = 0.49; the addition of an environmental co-variate, average annual water temperature at 50 m, improved the model fit and changed these values to MSY = 3.48 $t.km^{-2}$, B_{MSY} = 13.37 $t.km^{-2}$ and F_{MSY} = 0.25.

ECOPATH WITH ECOSIM (EwE) MODELS

Overall Model Fit

The EwE model including all effects explained 46% of the observed variation in biomass of the no-effects model (Table 2). The estimated k_{ij} parameters accounted for approximately 23%, fishing accounted for approximately 13% and the anomaly forcing for approximately 10%. Of the total number of estimated time series of biomass the SS of 68% were improved in the all-effects scenario compared to the scenario including only fishing and the k_{ij} parameters. The reduction in the overall SS caused by the inclusion of the primary production anomaly was considered statistically significant ($p<0.01$).

For some groups the SS increased in the all-effect scenario, including sharks, large pelagic fish, flounders and lobsters. The sharks and large pelagic populations are much larger and occur at a much greater geographic scale than the WSS/BoF ecosystem where a small part of the populations occur for a short time of the year. Thus, their dynamics are driven by factors that occur at the larger geographical scale. Flounders and lobster have increased over the period of time analysed herein, a trend opposite to that for most other groups (Figure 5)

Groups that had the largest improvement in fit with the all-effects model included silver hake 4+, macrozooplankton, demersal piscivores, other pelagics, shrimps and cod 1-3. The SS values for these groups in the all-effect scenario were less than 60% of the values in the no-anomaly scenario, indicating that the biophysical forcing was important for these groups, Figure 5. See Araujo and Bundy (2012) for further details of model fitting.

EwE Estimates of MSY

Virtually all of the multi-species estimates of MSY were different from the single species estimates (Figure 6). Many of the msMSY were lower than the ssMSY, but the model predicted a higher msMSY than ssMSY for American plaice, large benthivores, silver hake and mackerel. The sum of the msMSY was less than the sum of ssMSY, indicating that the total multi-species yield from this system is less than would be estimated from single species assessments.

Although theory suggests a negative linear relationship between the ratio of msMSY/ssMSY and trophic level (i.e., lower msMSY for higher trophic levels because of removals of lower level productivity by fishing), the negative linear relationship shown in Figure 7 is not significant. Walters et al. (2005) also found a variable response to this expected relationship and concluded that the direction of change from ssMSY to msMSY is not consistently related to trophic level.

For a variety of reasons, many of the groups in the model are aggregated into functional groups (tractability is a large factor). Focussing on a few key commercial species, cod, haddock, pollock and herring, msMSY is lower than ssMSY for all four species (Figure 6). Current F is lower than Fmsy for cod and haddock, but greater for pollock and herring (Figure 8).

DISCUSSION

In general, these three models make similar predictions about the effect of incorporating the broader ecosystem on estimates of fisheries reference points. They each do this differently: the MSVPA adds estimates of annual predation, summed across predators, to a single species VPA and provides estimates of herring biomass, predation mortality at age and fishing mortality at age; the production models calculate ASP then fit a Graham-Schaefer model to the data, including co-variates (environmental or trophic) and provide estimates of MSY and other reference points; EwE explicitly models tropho-dynamic interactions, and fits the model to time series of data, providing estimates of predation mortality, MSY, other reference points as well as many other products. Unlike the MSVPA, it is constrained by production at lower trophic levels (i.e., there has to be enough prey to support consumption by predators).

These models all have assumptions, caveats and uncertainties. The intention of this document is not to propose any one result for management purposes: rather it is to explore how including the ecosystem into our single species assessments can change our view of the world and affect reference points.

The MSVPA model is perhaps the most accessible since it builds upon the familiar structure of the VPA. Results demonstrated that predation mortality is greater than fishing mortality in most years, and is around 0.6 $year^{-1}$. To accommodate this additional mortality, the herring biomass estimated by the MSVPA is higher than the biomass estimated by the single species VPA. Estimates of predation mortality on herring from EwE are similar (Figure 9): here, predation mortality is estimated to increase over time from around 0.3 $year^{-1}$ in the early 1970s to around 0.5 $year^{-1}$ in 2000, after which it decreased. As would be expected for this major forage species, these multi-species estimates of predation mortality are far greater than the 0.2 usually allowed for in VPAs.

COMPARING MSY REFERENCE POINTS

Herring are a key mid-trophic level species in the 4X ecosystem, transferring productivity from lower trophic levels to higher trophic levels. The most recent assessment for herring did not produce an acceptable analytical model due to discrepancies between the acoustic survey estimate of spawning stock biomass (SSB) and the VPA results, as well as identified problems with aging of 4VWX herring (DFO 2010). Because the production model and EwE are not age structured (although EwE can include age-structure, the implementation used here aggregated herring in one group), these models side-step this problem.

Guénette and Stephenson (2012) used a Thompson Bell yield-per-recruit model to make projections from 2006 to 2025 in order to explore the impacts of different management options for herring. Projections were performed using F0.1 (0.23) and F_{MAX} (0.57) which are currently used for fisheries management, and current F (0.44) estimated from the MSVPA. Their results indicated that the F(0.1) fishing mortality was the only mortality rate considered that would result in some stock rebuilding: current F and F_{MAX} resulted in a lower spawning stock biomass than at the start of the simulation (Guénette and Stephenson 2012; Table 4).

Compared to these results, the estimates of F_{MSY} from the production models and EwE were lower than the F(0.1) value used by Guénette and Stephenson (2012): the values range from 0.12 to 0.24 (Table 3). This suggests that even the estimated value of F(0.1), which was estimated using the estimated partial recruitment from a single species VPA and a yield per recruit model, is too high. As noted by Guénette and Stephenson (2012) "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". Given that herring spawning biomass is at a low level and that there have been drastic reductions in sub-stocks (Power et al. 2010), these results suggest that serious consideration should be given to reducing F below F(0.1).

The results here confirm that single species estimates of MSY can not be achieved simultaneously, and that the sum of multi-species MSY is lower than the sum of single species MSYs. Here we can also compare the estimates of aggregate MSY from the production models with the estimates from EwE (Table 4). The EwE msMSY estimate of 4.26 falls between the two aggregate production model estimates, with and without a co-variate. It is encouraging that these estimates are comparable: Bundy et al. (2012) concluded from their analysis of 11 ecosystems that values of aggregate MSY are typically between 1 and 5 $t.km^{-2}$. This suggests that NAFO Divisions 4X/5Y is one of the more productive ecosystems in the northern hemisphere.

THE ROLE OF THE ENVIRONMENT

The fits of the production models and EwE model were improved when environmental factors were included. In both cases, water temperature was an important driver: the best fitting model for herring included sea surface temperature, wind and cod as a predator (Holsman et al. 2012); the best fitting aggregate surplus production model included temperature at 50 m (Bundy et al. 2012) and the EwE model included a predicted anomaly function which was significantly related to SST, a stratification index and the Atlantic Multidecadal Oscillation (linear regression).

At a broader level across 13 ecosystems, Holsman et al. (2012) concluded that in the northern hemisphere, herring and cod production is related to water temperature and wind speed. Empirical studies of the effects of the environment on fish production are also useful to elucidate these connections and develop mechanistic hypotheses (e.g., Shackell et al. 2012; Link et al. 2010b), but in order to be most useful for fisheries management, we need to incorporate the environment and the broader ecosystem into our advice.

These results, and others, indicate that we cannot afford to ignore the broader ecosystem in the science advice provided to fisheries management, particularly as we face increased environmental change. Further, although increasing the number of variables in models usually is considered to increase observation error in input data, these results suggest that biophysical and trophodynamic covariates can be important sources of process error and that failure to incorporate their effects can lead to underestimates of the uncertainty in biological reference points.

REFERENCE POINTS AND THE PRECAUTIONARY APPROACH

The models presented here were used as tools to explore the impact of including species interactions and the environment on the estimation of reference points. Reference points are critical to a precautionary approach and there have been numerous DFO meetings to discuss these topics (e.g., DFO 2006, 2009). There are guidelines for the definition of stock status and fishery status (DFO 2009; see Table 5) which are based on B_{MSY} and F_{MSY}. The results of the models presented here can start to inform the definition of these reference points from an ecosystem perspective. We do not pretend that these models are currently ready for direct incorporation into advice: all require far greater scrutiny and peer review. However, they demonstrate that methods exist to contribute to these issues.

REFERENCES

Ahrens, R., Walters, C., and Christensen, V. 2011. Foraging arena theory. Fish Fish. 13: 41-49.

Araújo, J.N., and Bundy, A. 2011. Description of three Ecopath with Ecosim ecosystem models developed for the Bay of Fundy, western Scotian Shelf and NAFO Division 4X. Can. Tech. Rep. Fish. Aquat. Sci. 2952.

Araújo, J.N., and Bundy A. 2012. The relative importance of environmental change, exploitation and trophodynamic control in determining ecosystem dynamics on the western Scotian Shelf, Canada. Mar. Ecol. Prog. Ser. 464: 51–67, 2012

Araújo, J.N., Mackinson, S., Stanford, R.J., Sims, D.W., Southward, A.J., Hawkins, S.J., Ellis, J.R., and Hart, P.J.B. 2006. Modelling food web interactions, variation in plankton production and fisheries on the western English Channel ecosystem. Mar. Ecol. Prog. Ser. 309: 175-187.

Beddington, J.R., and May, R.M. 1977. Harvesting natural populations in a randomly fluctuating environment. Science 197: 463–465.

Browman, H.I., and Stergiou, K.I. 2004. Perspectives on ecosystem-based approaches to the management of marine resources. Mar. Ecol. Prog. Ser. 274: 269–303.

Browman H.I., and Stergiou, K.I. 2005. Politics and socio-economics of ecosystem-based management of marine resources. Mar. Ecol. Prog. Ser. 300: 241–296.

Bundy, A. 2001. Fishing on ecosystems: The interplay of fishing and predation in Newfoundland-Labrador. Can. J. Fish. Aquat. Sci. 58: 1153–1167.

Bundy, A. 2004. Mass balance models of the eastern Scotian Shelf before and after the cod collapse and other ecosystem changes. Can. Tech. Rep. Fish. Aquat. Sci. 2520.

Bundy, A. 2005. Structure and function of the eastern Scotian Shelf Ecosystem before and after the groundfish collapse in the early 1990s. Can. J. Fish. Aquat. Sci. 62(7): 1453-1473.

Bundy, A., Heymans, J.J., Morissette, L., and Savenkoff, C. 2009. Seals, cod and forage fish: A comparative exploration of variations in the theme of stock collapse and ecosystem change in four northwest Atlantic ecosystems. Prog. Oceanog. 81: 188–206.

Bundy, A., Bohaboy, E.C., Hjermann, D.O., Mueter, F.J., Fu, C., and Link, J.S. 2012. Common patterns, common drivers: Comparative analysis of aggregate surplus production across ecosystems. Mar. Ecol. Prog. Ser. 459: 203-218.

Christensen, V., and Walters, C. 2004. Ecopath with Ecosim: Methods, capabilities and limitations. Ecol. Mod. 172: 109–139.

Christensen, V., Walters, C.J., and Pauly, D. 2005. Ecopath with Ecosim: A user's guide. Fisheries Centre, University of British Columbia, Vancouver, BC. Nov. 2005 ed. URL http://www.ecopath.org/.

Coll, M., Bundy, A., and Shannon, L. 2009. Ecosystem modelling using the Ecopath with Ecosim approach. *In* Computers in fisheries research. Edited by B.A. Megrey and E. Moksness. Springer, New York, NY. pp. 225–291.

Curran, K., Bundy, A., Craig, M., Hall, T., Lawton, P., and Quigley, S. 2012. Recommendations for science, management, and an ecosystem approach in Fisheries and Oceans Canada, Maritimes Region. DFO Can. Sci. Advis. Sec. Res. Doc. 2012/061.

DFO. 2006. A harvest strategy compliant with the Precautionary Approach. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2006/023.

DFO. 2008. National workshop on modelling tools for Ecosystem Approaches to Management; 22-25 October 2007. DFO Can. Sci. Advis. Sec. Proceed. Ser. 2008/007.

DFO. 2009. A fishery decision-making framework incorporating the Precautionary Approach. Available at [Internet] http://www.dfo-mpo.gc.ca/fm-gp/peches-fisheries/fish-ren-peche/sff-cpd/precaution-eng.htm (accessed 26 June 2012).

DFO. 2010. 2010 assessment of 4VWX herring. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2010/038.

DFO. 2011. Impacts of grey seals on fish populations in eastern Canada. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2010/071.

Frank, K.T., Petrie, B., Shackell, N.L., and Choi, J.S. 2006. Reconciling differences in trophic control in mid-latitude marine ecosystems. Ecol. Lett. 9: 1096–1105.

Gaichas, S.K., Bundy, A., Miller, T., Moksness, E., and Stergiou, K. 2012a. What drives marine fisheries production? Mar. Ecol. Prog. Ser. 459: 159-163.

Gaichas, S.K., Gamble, R., Fogarty, M., Benoît, H., Essington, T., Fu, C., Koen-Alonso, M., and Link, J. 2012b. Assembly rules for aggregate-species production models: Simulations in support of management strategy evaluation. Mar. Ecol. Prog. Ser. 459: 275-292.

Guénette, S., and Stephenson, R.L. 2012. Accounting for predators in ecosystem-based management of herring fisheries of the western Scotian Shelf, Canada. *In* Global Progress in Ecosystem-Based Fisheries Management. Edited by G.H. Kruse, H.I. Browman, K.L. Cochrane, D. Evans, G.S. Jamieson, P.A. Livingston, D. Woodby, and C.I. Zhang. Alaska Sea Grant, University of Alaska, Fairbanks, Alaska. pp. 105-128.

Holsman, K., Essington, T., Miller, T.J., Koen-Alonso, M., and Stockhausen, W.J. 2012. Comparative analysis of cod and herring production dynamics across 13 northern marine ecosystems. Mar. Ecol. Prog. Ser. 459: 231-246.

Jamieson, G., and O'Boyle, R. 2001. Proceedings of the National Workshop on objectives and indicators for ecosystem-based management, Sidney, British Columbia, 27 February – 2 March 2001. DFO Can. Sci. Adv. Sec. Proceed. Ser. 2001/009.

Link, J., and Bundy, A. 2012. Ecosystem Modeling in the Gulf of Maine Region: Towards an Ecosystem Approach to Fisheries. Pages 281-310 *in* R. L. Stephenson, J. H. Annala, J. A. Runge, and M. Hall-Arber, editors. Advancing an ecosystem approach in the Gulf of Maine. American Fisheries Society, Symposium 79, Bethesda, Maryland. © Copyright 2012 by the American Fisheries Society

Link, J., Overholtz, W., O'Reilly, J., Green, J., Dow, D., Palka, D., Legault, C., Vitaliano, J., Guida, V., Fogarty, M., Brodziak, J., Methratta, E., Stockhausen, W., Col, L., Waring, G., and Griswold, C. 2008. An overview of EMAX: The Northeast U.S. Continental Shelf Ecological Network. J. Mar. Sys. 74: 453-474.

Link, J.S., Megrey, B.A., Miller, T.J., Essington, T., Boldt, J., Bundy, A., Moksness, E., Drinkwater, K.F., and Perry, R.I. 2010a. Comparative analysis of marine ecosystems: International Production Modelling Workshop. Biol. Lett. December 23, 2010; 6: 723-726; published online before print July 7, 2010, doi:10.1098/rsbl.2010.0526.

Link, J.S., Yemane, D., Shannon, L.J., Coll, M., Shin, Y.-J., Hill, L., and Borges, M.F. 2010b. Relating marine ecosystem indicators to fishing and environmental drivers: An elucidation of contrasting responses. ICES J. Mar. Sci. 67: 787–795.

Link, J.S., Gaichas, S., Miller, T.J., Essington, T., Bundy, A., Boldt, J., Drinkwater, K.F., and Moksness, E. 2012. Synthesizing lessons learned from comparing fisheries production in 13 northern hemisphere ecosystems: Emergent fundamental features. Mar. Ecol. Prog. Ser. 459: 293-302.

Link, J., Bundy, A., Overholtz, W.J., Shackell, N.L., Manderson, J., Duplisea, D., Hare, J., Koen-Alonso, M., and Friedland, K.D. 2011. Ecosystem-based fisheries management in the Northwest Atlantic. Fish Fish. 12: 152–170.

Lucey, S.M., Cook, A.M., Boldt, J., Link, J.S., Essington, T., and Miller, T.J. 2012. Comparative analyses of surplus production dynamics of functional feeding groups across 12 large marine ecosystems. Mar. Ecol. Prog. Ser. 459: 219-229.

Ludwig, D., and Walters, C.J. 1985. Are age structured models appropriate for catch-effort data? Can. J. Fish. Aquat. Sci. 42: 1066–1072.

Ludwig, D., and Walters, C.J. 1989. A robust method for parameter estimation from catch and effort data. Can. J. Fish. Aquat. Sci. 46: 137–144.

Mackinson, S., Deas, B., Beveridge, D., and Casey, J. 2009. Mixed-fishery or ecosystem conundrum? Multispecies considerations inform thinking on long-term management of North Sea demersal stocks. Can. J. Fish. Aquat. Sci. 66: 1107-1129.

Magnússon, K.G. 1995. An overview of the multispecies VPA - theory and applications. Rev. Fish. Biol. Fish. 5: 195-212.

Mahon, R., and Smith, R.W. 1989. Demersal fish assemblages on the Scotian Shelf, northwest Atlantic: Spatial distribution and persistence. Can. J. Fish. Aquat. Sci. 46: 134–152.

Mangel, M. 2006. The theoretical biologist's toolbox: Quantitative methods for ecology and evolutionary biology. Cambridge University Press, Cambridge, UK.

Mueter, F.J., and Megrey, B.A. 2006. Maximum productivity estimates for the groundfish complexes of the Gulf of Alaska and Eastern Bering Sea/Aleutian Islands. Fish. Res. 81: 189-201.

Mohn, R.K. 1980. Bias and error propagation in logistic models. Can. J. Fish. Aquat. Sci. 37: 276–1283.

Murawski, S.A. 2007. Ten myths concerning ecosystem approaches to marine resource management. Mar. Policy 31: 681-690.

NRC. 1998. Improving fish stock assessments. National Academy Press, Washington, DC.

Overholtz, W.J., Jacobson, L.D., and Link, J.S. 2008. Developing an ecosystem approach for assessment advice and biological reference points for the Gulf of Maine–Georges Bank herring complex: Adding the impact of predation mortality. N. Am. J. Fish. Manage. 28: 247 –257.

Pikitch, E.K., Santora, C., Badcock, E., Bakun, A., Bonfil, R., Conover, D. O., Dayton, P. Doukakis, P., Fluharty, D.,Heneman, B., Houde, E.D., Link, J., Livingston, P.A., Mangel, M., McAllister, M.K., Pope, J., and Sainsbury, K.J. 2004. Ecosystem-based fishery management. Science. 305(5682): 346-347.

Pitcher, T.J., Kalikoski, D., Short, K., Varkey, D., and Pramoda, G. 2009. An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. Mar. Pol. 33: 223–232.

Plagányi, É.E. 2007. Models for an ecosystem approach to fisheries. FAO Fish. Tech. Paper. No. 477. Rome, FAO. Available at [Internet] http://www.fao.org/docrep/010/a1149e/a1149e00.htm (accessed 26 June 2012).

Power, M.J., Fife, F.J., Knox, D., and Melvin, G.D. 2010. 2009 evaluation of 4VWX herring. DFO Can. Sci. Advis. Sec. Res. Doc. 2010/111.

Punt, A.E. 2003. Extending production models to include process error in the population dynamics. Can. J. Fish. Aquat. Sci. 60: 1217–1228.

Restrepo, V.R., Mace, P.M., and Serchuk, F.M. 1999. The precautionary approach: A new paradigm or business as usual? NOAA Tech. Memo. NMFS-F/SPO: 61-70.

Rice, J.C. 2011. Managing fisheries well: Delivering the promises of an ecosystem approach. Fish Fish. 12: 209-231.

Shackell, N.L., and Frank, K.T. 2003. Marine fish diversity on the Scotian Shelf, Canada. Aquatic Conservation: Marine and Freshwater Ecosystems, 13: 305–321.

Shackell, N.L., Bundy, A., Nye, J.A., and Link, J.S. 2012. Common large-scale responses to climate and fishing across Northwest Atlantic ecosystems. ICES J. Mar. Sci. 69(2): 151-162.

Shackell, N.L., Frank, K.T., Fisher, J.A.D., Petrie, B., and Leggett, W.C. 2010. Decline in top predator body size and changing climate alter trophic structure in an oceanic ecosystem. P. Roy. Soc. B.-Biol. Sci. 277: 1353-1360.

Shannon, L.J., Christensen, V., and Walters, C.J. 2004. Modelling stock dynamics in the southern Benguela ecosystem for the period 1978-2002. Afr. J. Mar. Sci. 26: 179–196.

Sherman, K., and Hempel, G. (Editors) 2008. The UNEP Large Marine Ecosystem Report: A perspective on changing conditions in LMEs of the world's regional seas. UNEP Regional Seas Report and Studies No. 182. United Nations Environment Programme. Nairobi, Kenya.

Sissenwine, M. 1978. Is MSY an adequate foundation for optimum yield? Fish. 3(6): 22-42.

Sparholt, H. 1990. Improved estimates of the natural mortality rates of nine commercially important fish species included in the North Sea multispecies VPA model. J. Cons. Int. Explor. Mer. 46(2): 211-223.

Townsend, H.M., Link, J.S., Osgood, K.E., Gedamke, T., Watters, G.M., Polovina, J.J., Levin, P.S., Cyr, N., and Aydin, K.Y. (Editors). 2008. National Marine Fisheries Service Report of the National Ecosystem Modeling Workshop (NEMoW). U.S. Dep. Commerce, NOAA Tech. Memo. NMFS-F/SPO-87.

Tyrrell, M.C., Link, J.S., and Moustahfid, H. 2011. The importance of including predation in fish population models: Implications for biological reference points. Fish. Res. 108: 1-8.

Tyrrell, M.C., Link, J.S., Moustahfid, H., and Overholtz, W.J. 2008. Evaluating the effect of predation mortality on forage species population dynamics in the Northeast US continental shelf ecosystem using multispecies virtual population analysis. ICES J. Mar. Sci. 65: 1689–1700.

Walters, C.J., and Martell, S.J. 2004. Fisheries ecology and management. Princeton University Press, New Jersey, NJ.

Walters, C.J., Christensen, V., Martell, S.J., and Kitchell, J.F. 2005. Possible ecosystem impacts of applying MSY policies from single-species assessment. ICES J. Mar. Sci. 62: 558-568.

Worm, B., Hilborn, R., Baum, J.K., Branch, T.A., Collie, J.S., Costello, C., Fogarty, M.J., Fulton, E.A., Hutchings, J.A., Jennings, S., Jensen, O.P., Lotze, H.K., Mace, P.M., McClanahan, T.R., Minto, C., Palumbi, S.R., Parma, A.M., Ricard, D., Rosenberg, A.A., Watson, R., and Zeller, D.A.F. Rebuilding global fisheries. Science 325: 578-585.

Zwanenburg, K.C.T., Bowen, D., Bundy, A., Drinkwater, K., Frank, K., O'Boyle, R., Sameoto, D., and M. Sinclair. 2002. Decadal changes in the Scotian Shelf large marine ecosystem. *In* Large marine ecosystems of the North Atlantic: Changing states and sustainability. Edited by K. Sherman and H.R. Skjoldal. Elsevier Science B.V., Amsterdam, the Netherlands. pp. 105–150.

Table 1. Estimates of MSY, B_{MSY} and F_{MSY} from production models applied at the functional group level (Sean Lucey, NOAA, pers. comm.).

	Group	B_{MSY} ($t.km^{-2}$)	MSY ($t.km^{-2}$)	F_{MSY}
Habitat	Pelagic	10.8	2.1	0.20
	Demersal	12.2	1.5	0.12
	SUM	23.0	3.6	
Feeding Guild	Benthivore	1.1	0.6	0.51
	Planktivore	8.6	2.1	0.24
	Zoopivore	2.1	1.0	0.46
	Piscivore	2.1	0.6	0.29
	SUM	14.0	4.3	
Size	Small	8.9	2.0	0.22
	Medium	1.9	0.9	0.48
	Large	3.0	1.4	0.45
	SUM	13.8	4.3	

Table 2. Ecosim runs with the respective sum of squared deviations (SS) of log biomass from log predicted biomass. In the no-effects scenario, the model estimates are forced to remain constant at the baseline values; in the all effects, model is run including the estimated k_{ij} (vulnerability multipliers), time series of fishing and the estimated primary production anomaly forcing (Araújo and Bundy 2012).

Scenario number	Scenario description	SS	% variance
1	No-effects	498	
2	All effects	269	46
3	Fishing + k_{ij}	323	35
4	Fishing	434	13
5	k_{ij}	389	23
6	anomaly	450	10

Table 3. Comparison of various estimates of MSY for herring, plus F(0.1) used in MSVPA.

		MSY ($t.km^{-2}$)	F_{MSY} ($t.km^{-2}$)	B_{MSY}
Herring	SS_PM	1.97	0.18	10.86
	Pelagic_PM	1.47	0.12	10.82
	Planktivore	2.09	0.24	8.65
	SS_EwE	2.38	0.12	19.80
	MS_EwE	1.71	0.12	14.21
	MSVPA (F0.1)		0.23	

Table 4. Comparison of estimates of aggregate MSY from the production model with total MSY estimates from EwE.

	MSY
AGG_PM_no covariate	5.38
AGG_PM_with covariate	3.48
Sum EwE SS	5.27
Sum EwE MS	4.26

Table 5. Guidance to Identify Reference Points and Harvest Rules (Annex 1b, DFO 2009, http://www.dfo-mpo.gc.ca/fm-gp/peches-fisheries/fish-ren-peche/sff-cpd/precaution-eng.htm).

Stock Status

In critical zone.
The stock is considered to be in "the critical zone" if the mature biomass, or its index, is less than or equal to 40% of B_{MSY}. In other words: *Biomass ≤ 40%* B_{MSY}.

In cautious zone.
The stock is considered to be in the "cautious zone" if the biomass, or its index, is higher than 40% of B_{MSY} but lower than 80% of B_{MSY}. In other words: *40%* B_{MSY} < *Biomass* < *80%* B_{MSY}.

Healthy.
The stock is considered to be "healthy" if the biomass, or its index, is higher than 80% of B_{MSY}. In other words: *Biomass ≥ 80%* B_{MSY}.

Fishery Status

Harvest at or below removal reference.
The harvest on this stock is considered to be at or below the removal reference if the harvest rate, or the fishing mortality (F), is lower than the provisional harvest rule given below. In other words: F≤ provisional harvest rule.

Harvest exceeds removal reference.
The harvest on this stock is considered to be above the removal reference if the harvest rate, or the fishing mortality (F), is higher than the provisional harvest rule given below. In other words: F > provisional harvest rule.

Provisional Harvest Rule.
In absence of a pre-agreed harvest rule developed in the context of the precautionary approach, a provisional removal reference or fishing mortality (say F_p) could be used to guide management and to assess harvest in relation to sustainability. The provisional harvest rule is as follows:

When the stock is in the "Healthy Zone":	$F_p < F_{MSY}$
When the stock is in the "Cautious Zone":	$F_p < F_{MSY} \times ((\text{Biomass} - 40\% B_{MSY}) / (80\% B_{MSY} - 40\% B_{MSY}))$
When the stock is in the "Critical Zone":	$F_p = 0$

Table 5 continued.

Note on B_{MSY} and F_{MSY}.
In absence of estimates related to the status of the stock and of the fishery at the Maximum Sustainable Yield, options for provisional estimates of B_{MSY} and F_{MSY} are provided below.

Biomass at MSY.
In absence of an estimate of B_{MSY} from an explicit model, the provisional estimate of B_{MSY} could be taken as follows (select the first feasible option):

- The biomass corresponding to the biomass per recruit at F0.1 multiplied by the average number of recruits; or
- The average biomass (or index of biomass) over a productive period; or
- The biomass corresponding to 50% of the maximum historical biomass.

Fishing mortality at MSY.
In absence of an estimate of F_{MSY} from an explicit model, the provisional estimate of F_{MSY} could be taken as follows (select the first feasible option):

- The fishing mortality corresponding to F0.1; or
- The average fishing mortality (or an index of fishing mortality) that did not lead to stock decline over a productive period; or
- The fishing mortality equal to natural mortality inferred from life history characteristics of the species.

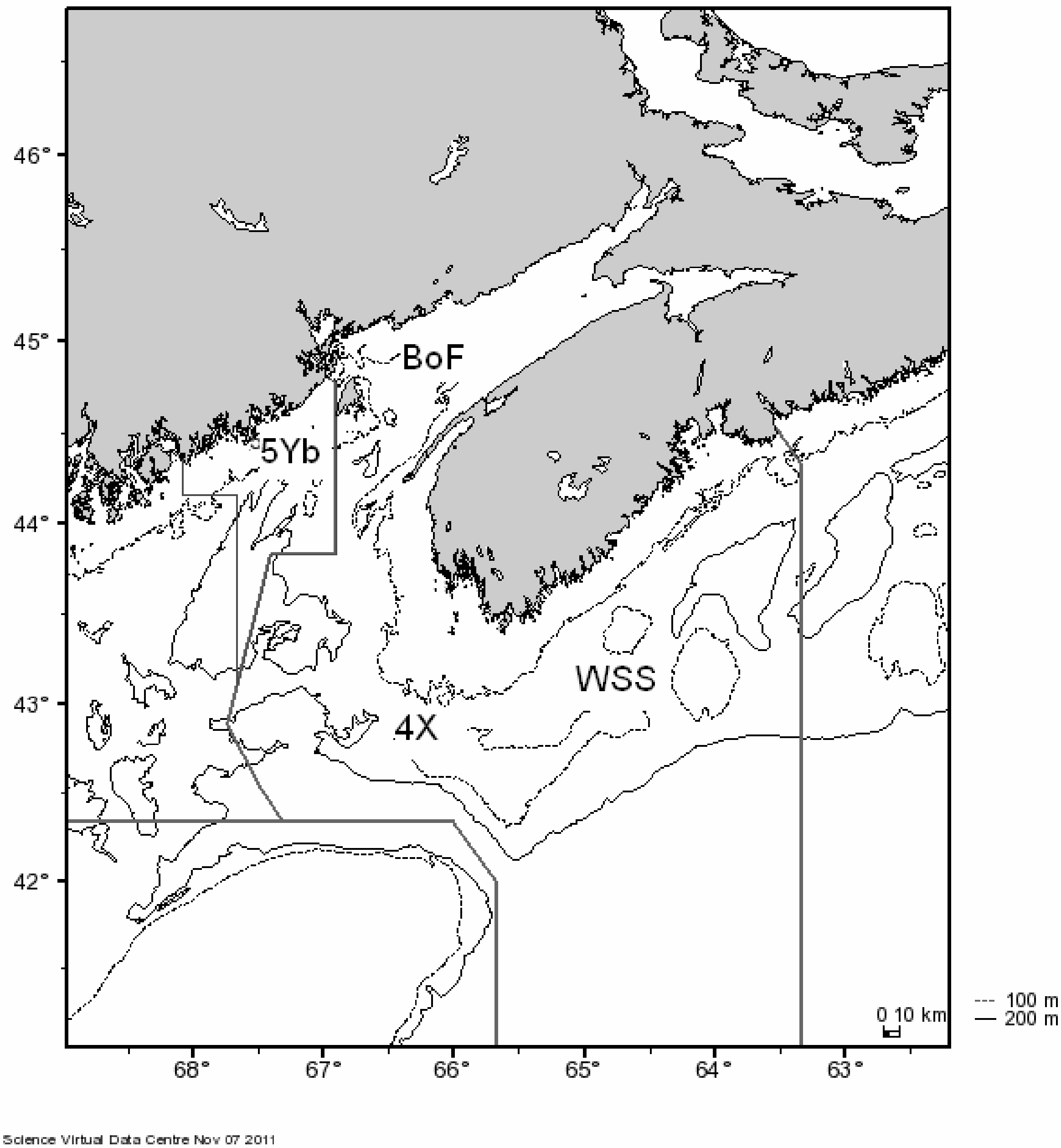


Figure 1. Location of NAFO Divisions 4X/5Y (east coast of Canada).

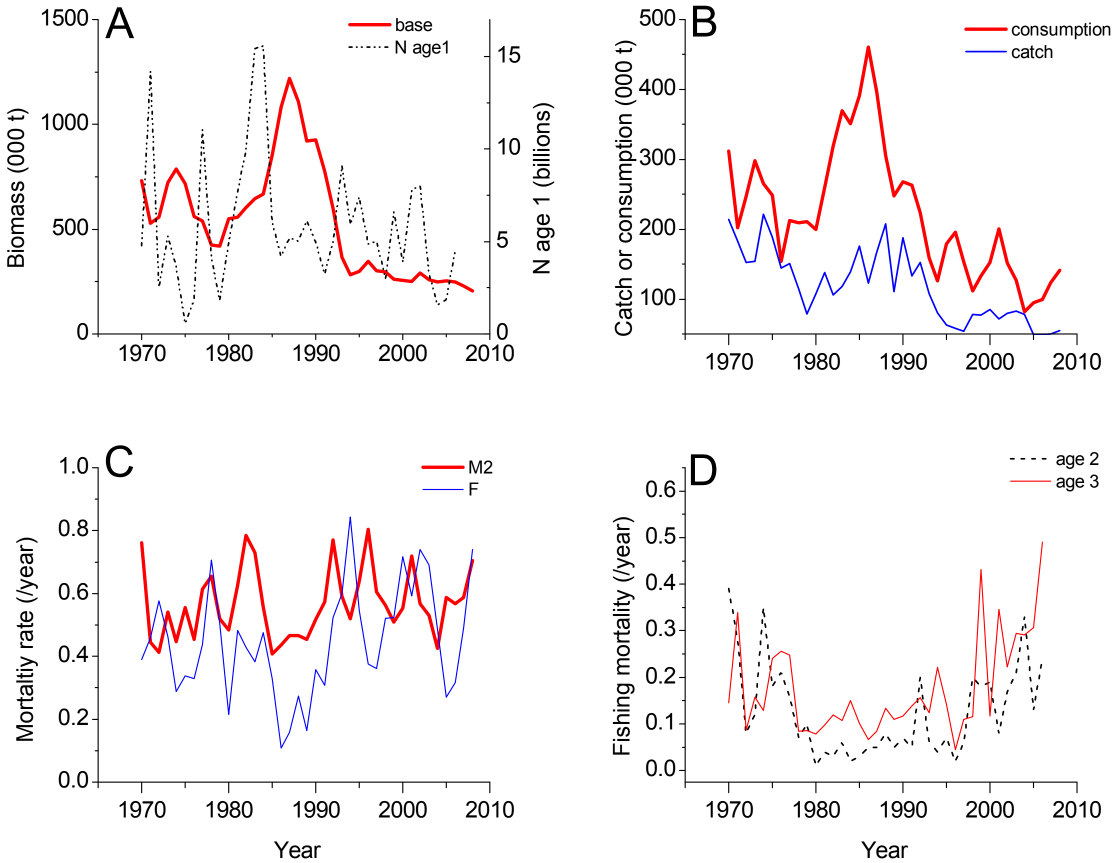


Figure 2. Results from the base scenario MSVPA. A. Herring abundance estimates from the MSVPA, and the number of herring at age 1 estimated by the MSVPA; B. Herring deaths by fishing and predation; C. Herring fishing mortality (for ages 5+) and natural mortality rates; and D. Herring fishing mortality by age group under the base scenario (adapted from Figure 3, Guénette and Stephenson 2012).

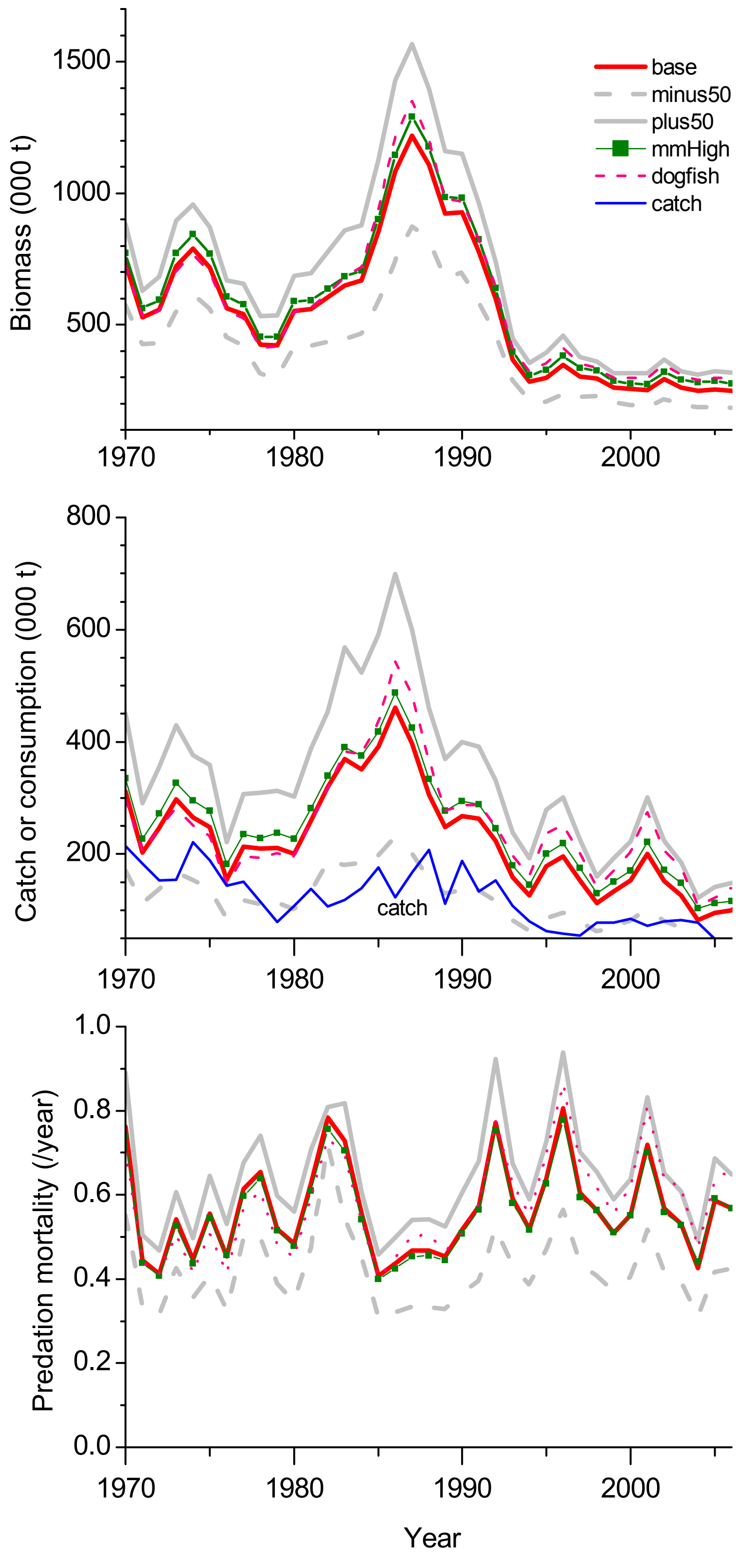
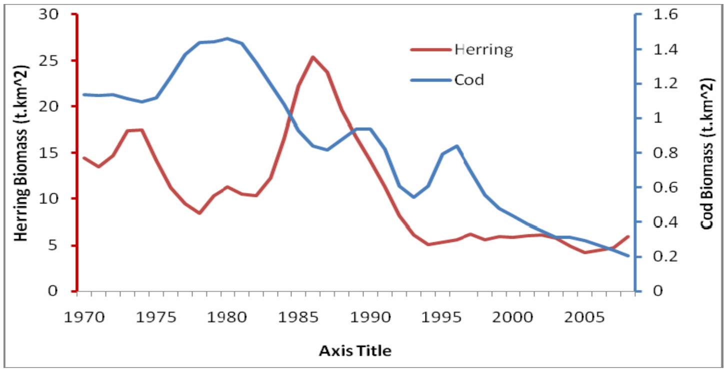
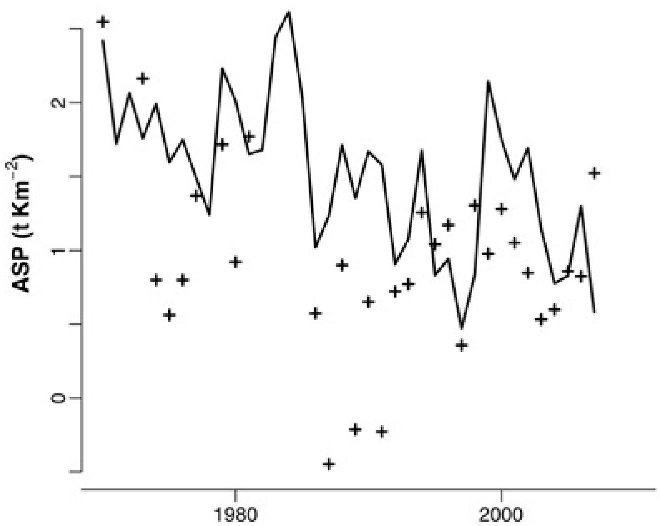


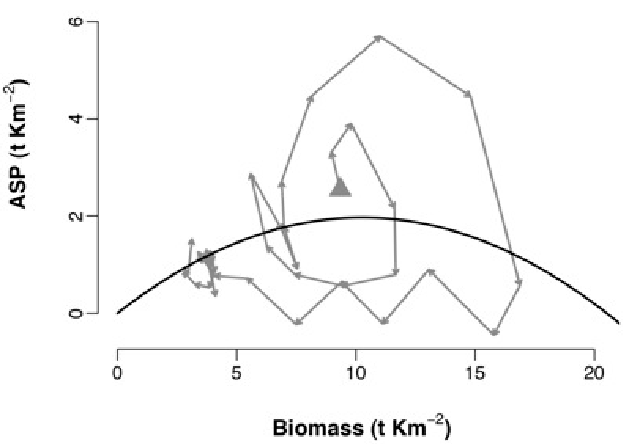
Figure 3. Estimates of herring population, predation mortality and herring consumed for scenarios compared to the base scenario. See text for the definitions of scenarios (adapted from Figure 5, Guénette and Stephenson 2012).



(a)



(b)



(c)

Figure 4. (a) Trends in biomass of herring and cod in NAFO Division 4X/5Y; (b) model estimates of herring annual surplus production (ASP) showing model fit to the data; and (c) observed annual surplus production (ASP) and biomass (t.km^{-2}) of herring (grey line) and ASP curve from covariate model under mean environmental conditions (black line). Figures (b) and (c) adapted from Holsman et al. (2012).

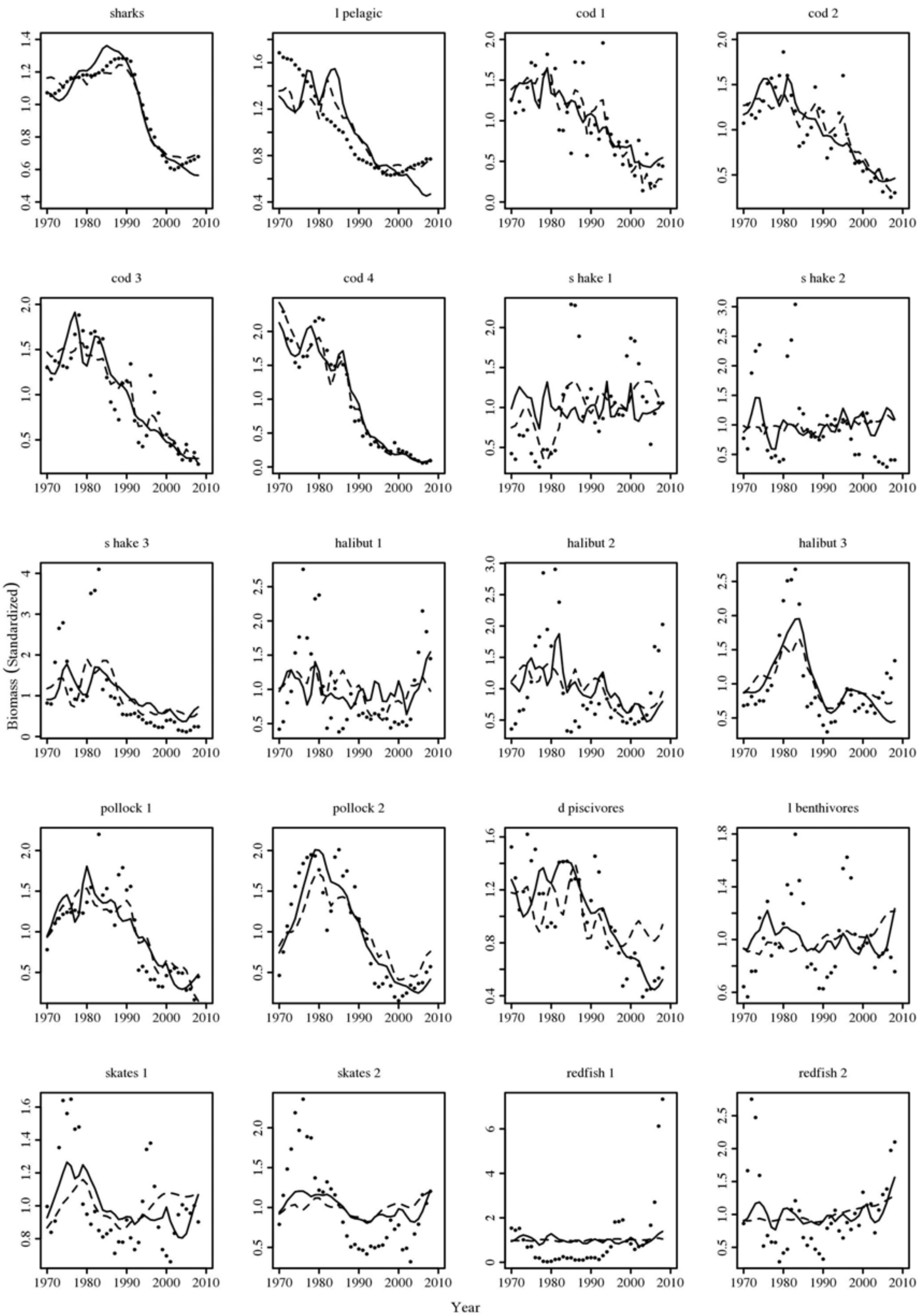


Figure 5. Biomass (standardized) time series estimates from the Ecopath with Ecosim model from 1970 to 2008. Filled circles represent the input estimates; black lines represent Ecosim estimates using a scenario with primary production forcing (all effects model); and dashed lines are for a second scenario using egg production forcing functions for a selected group of species (not discussed here; adapted from Araújo and Bundy 2012).

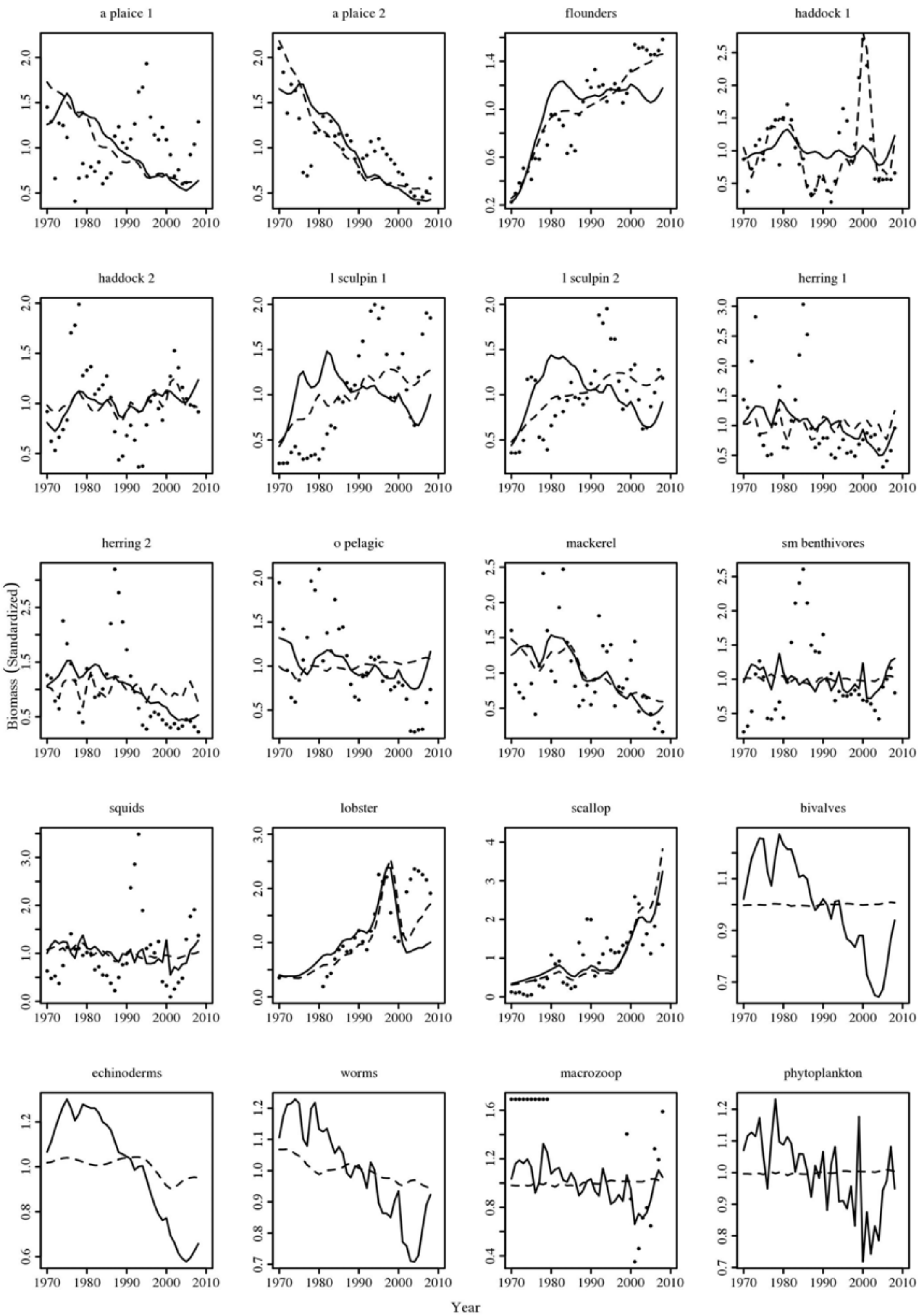


Figure 5. Continued (adapted from Araújo and Bundy 2012).

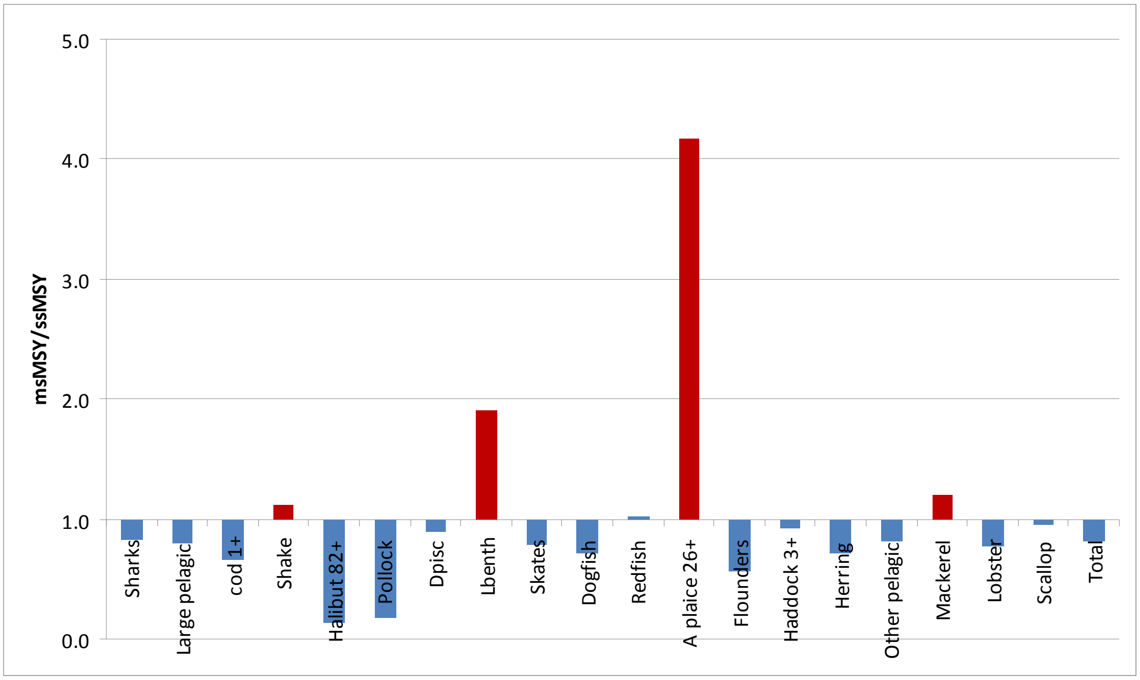


Figure 6. EwE estimates of MSY expressed as a ratio of multi-species (ms) to single species (ss) MSY for fished groups. Red bars=msMSY>ssMSY; blue bars=msMSY<ssMSY.

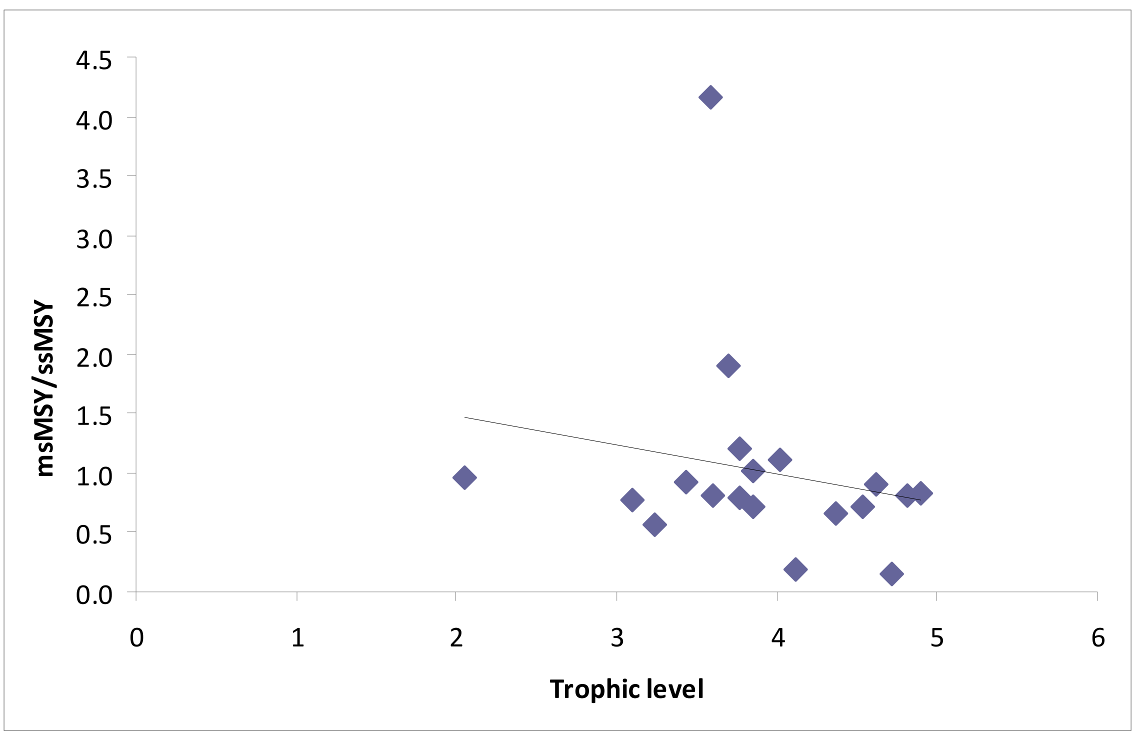


Figure 7. Ratio of multi-species MSY to single species MSY plotted against trophic level. The relationship is not significant.

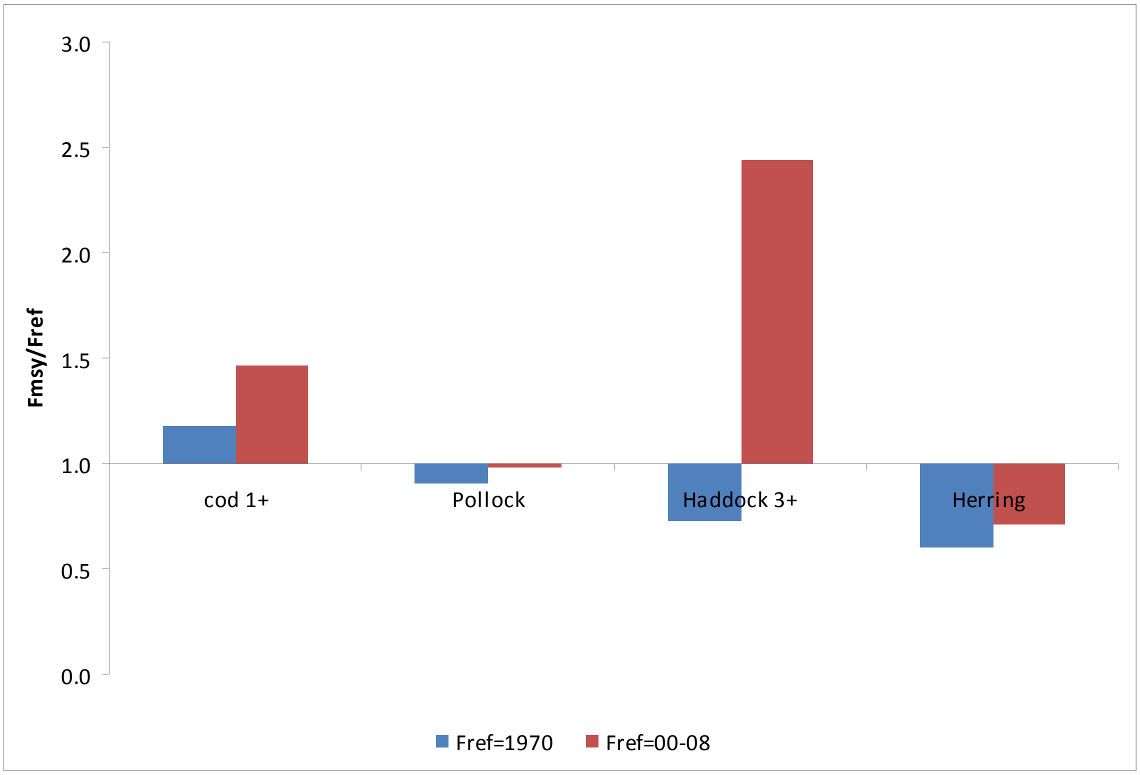


Figure 8. Ratio of F_{MSY} to F_{REF} for 1970 (blue bars) and 2000-2008 (red bars).

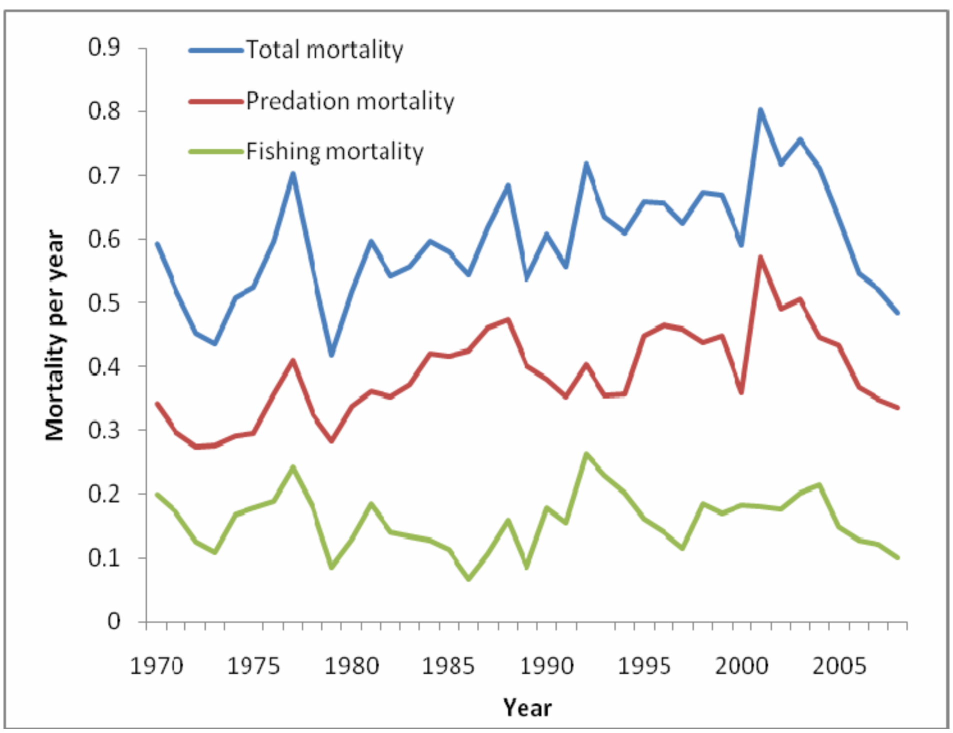


Figure 9. Herring total mortality, predation mortality and fishing mortality estimates by Ecopath with Ecosim.