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2015

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Robert B. Thorpe

Will J. F. Le Quesne

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# Evaluation and management implications of uncertainty in a multispecies size-structured model of population and community responses to fishing

Robert B. Thorpe[1]*, Will J. F. Le Quesne[1], Fay Luxford[2], Jeremy S. Collie[3] and Simon Jennings[1,4]

[1]*Centre for Environment, Fisheries and Aquaculture Science (CEFAS), Lowestoft Laboratory, Lowestoft NR33 0HT, UK;* [2]*School of Mechanical, Aerospace and Civil Engineering, University of Manchester, Manchester M13 9PL, UK;* [3]*Graduate School of Oceanography, University of Rhode Island, South Ferry Road, Narragansett, RI 02882, USA; and* [4]*School of Environmental Sciences, University of East Anglia, Norwich NR4 7TJ, UK*

## Summary

**1.** Implementation of an ecosystem approach to fisheries requires advice on trade-offs among fished species and between fisheries yields and biodiversity or food web properties. However, the lack of explicit representation, analysis and consideration of uncertainty in most multispecies models has limited their application in analyses that could support management advice.

**2.** We assessed the consequences of parameter uncertainty by developing 78 125 multispecies size-structured fish community models, with all combinations of parameters drawn from ranges that spanned parameter values estimated from data and literature. This unfiltered ensemble was reduced to 188 plausible models, the filtered ensemble (FE), by screening outputs against fish abundance data and ecological principles such as requiring species' persistence.

**3.** Effects of parameter uncertainty on estimates of single-species management reference points for fishing mortality ($F_{MSY}$, fishing mortality rate providing MSY, the maximum sustainable yield) and biomass ($B_{MSY}$, biomass at MSY) were evaluated by calculating probability distributions of estimated reference points with the FE. There was a 50% probability that multispecies $F_{MSY}$ could be estimated to within ±25% of its actual value, and a 50% probability that $B_{MSY}$ could be estimated to within ±40% of its actual value.

**4.** Signal-to-noise ratio was assessed for four community indicators when mortality rates were reduced from current rates to $F_{MSY}$. The slope of the community size spectrum showed the greatest signal-to-noise ratio, indicating that it would be the most responsive indicator to the change in fishing mortality $F$. Further, the power of an ongoing international monitoring survey to detect predicted responses of size spectrum slope was higher than for other size-based metrics.

**5.** Synthesis and applications: Application of the ensemble model approach allows explicit representation of parameter uncertainty and supports advice and management by (i) providing uncertainty intervals for management reference points, (ii) estimating working values of reference points that achieve a defined reduction in risk of not breaching the true reference point, (iii) estimating the responsiveness of population, community, food web and biodiversity indicators to changes in $F$, (iv) assessing the performance of indicators and monitoring programmes and (v) identifying priorities for data collection and changes to model structure to reduce uncertainty.

**Key-words:** biodiversity, ecosystem-based management, fish community, fisheries, food web, large fish indicator, power analysis, risk, size-based, trade-off

## Introduction

One facet of the adoption of an ecosystem approach to fisheries is the need to specify and meet objectives for ecosystem components and attributes as well as target species (Garcia & Cochrane 2005; Rice 2011). This requires that objectives can be defined and advice provided on management actions to meet them. In practice, both agreeing on objectives and advising how to meet them have been challenging, in part because advice on consequences of adopting alternate objectives and the probability of meeting them is uncertain and this uncertainty is not quantified.

Several models have been developed to assess fishing effects on community or ecosystem properties (e.g. Plagányi 2007). They have improved understanding of fisheries interactions with ecosystems and are often used to underpin state assessments, but relatively few are used operationally; the main exceptions being those that consider interactions among small groups of species (e.g. Gjosaeter, Bogstad & Tjelmeland 2002).

One reason for limited uptake is that model outputs are assumed to be highly uncertain, and yet there have been few attempts to express uncertainty and its implications (e.g. Hill *et al.* 2007; Espinoza-Tenorio *et al.* 2012; Gaichas *et al.* 2012; Link *et al.* 2012). This contrasts with analysis and advice on single-species exploitation, for which advisers and managers have longstanding experience presenting and addressing uncertainty (e.g. Francis & Shotton 1997).

Effects of parameter uncertainty on complex ecosystem models are poorly understood and may be challenging to address owing to practical constraints such as long run-times and few data for parameterization or validation. However, the complexity of these models is not a prerequisite for addressing many operational management questions (e.g. Fulton, Smith & Johnson 2003; Butterworth & Plaganyi 2004), and a range of less complex models that further simplify ecosystems and/or deal with subsets of ecosystems can be adopted. One class of these models includes size-based multispecies models. These focus on groups of interacting fish species in an otherwise generalized food web and seek to predict changes in species' abundance at size as a function of fishing mortality $F$ (e.g. Hall *et al.* 2006; Rochet *et al.* 2011; Blanchard *et al.* 2014). These models are practical to apply as they are based on measurable properties of species' life histories and dynamics, are relatively straightforward to parameterize with empirical data and have short run-times. They are also suitable for supporting management advice as they predict the effects of changes in $F$ on both size and abundance of species in the modelled community and can therefore be used to estimate single-species fisheries and conservation reference points in a multispecies context as well as a range of community metrics proposed or specified as indicators for evaluating the fishing effects (e.g. Shin *et al.* 2005; EU 2008, 2010; Greenstreet *et al.* 2011; Shephard, Reid & Greenstreet 2011).

To support development of operational management advice on the effects of fishing, we develop and apply a method to assess and describe parameter uncertainty and its consequences in a size-based multispecies model. Parameter uncertainty is explicitly incorporated into the analysis by developing multiple models based on the full range of possible parameter values, rather than applying a single model based on a best estimate of each individual parameter. Emergent population dynamics predicted by each model are then screened against data to generate a reduced set of models displaying credible dynamics. The approach is used to describe uncertainty in single species and community reference points and single species and community responses to alternate management actions. Predicted uncertainties are used to assess the power of monitoring surveys to detect responses in community metrics and to identify where investment in future data collection should be targeted to reduce uncertainty.

## Materials and methods

Analyses were performed using a modified version of the length-based multispecies model initially developed by Hall *et al.* (2006) to represent the Georges Bank fish community and subsequently applied to the North Sea community (Rochet *et al.* 2011). The model represents 21 fish species in 32 equal length classes (each *c.* 5 cm), spanning the full size range of species represented in the model (Table 1). Progression of individuals through length classes is represented using the deterministic von Bertalanffy growth equation (VBGE). Individuals mature when they reach a threshold size defined by a logistic model, with 50% of individuals mature at the length of maturity ($L_{mat}$) defined in Table 1. Reproduction is described with a spawner–recruit relationship, which determines the numbers of recruits entering the smallest size class from the biomass of mature individuals. Species' dynamics are linked via predation mortality ($M2$), which varies with predator abundance, and size and species preference. Size preference is described with a preference function based on a log-normal distribution and species preference with a diet matrix indicating who eats whom (Hall *et al.* 2006). In each length class, individuals are also susceptible to residual natural mortality ($M1$) and fishing mortality ($F$). Other details of model structure and implementation are provided in Hall *et al.* (2006) and Rochet *et al.* (2011).

Expected ranges of parameter uncertainty were expressed for seven key parameters with a default estimate and four alternate values that spanned the range of uncertainty reported in published literature or estimated from data (Table S1). Model variants were established for all combinations of parameter values, generating an initial set of $5^7$ (78 125) variants. Each variant was run under 'no-fishing' and 'historic fishing' scenarios, and outputs were screened to establish whether all species persisted (with no-fishing) and species' biomass was consistent with independent estimates (with 'historic fishing', details below). Model variants that produced outputs that did not meet these criteria were excluded. The remaining model variants were treated as equally plausible and dubbed the filtered ensemble (FE). Evaluations of fishing scenarios were conducted using the FE and thus generated probability distributions of outputs.

Eleven species in the model are assessed by ICES with age-based single-species models; Norway pout *Trisopterus esmarkii* (Nilsson), sandeel *Ammodytes* spp., herring *Clupea harengus* Linnaeus, horse mackerel *Trachurus trachurus* (Linnaeus), sole *Solea solea* (Linnaeus), mackerel *Scomber scombrus* Linnaeus, whiting *Merlangius merlangus* (Linnaeus), plaice *Pleuronectes platessa* Linnaeus, haddock *Melanogrammus aeglefinus* (Linnaeus), cod *Gadus morhua* Linnaeus and saithe *Pollachius virens* (Linnaeus). For these species, default VBGE parameters were calculated by fitting the VBGE to ICES weight-at-age data. For remaining species, VBGE parameter estimates were taken from Rochet *et al.* (2011) and Blanchard *et al.* (2014). Prey species that were not individually modelled were represented as a constant resource of $10^{10}$ g of 'other food' available to all predator length classes at each time step. This equates to a total of $2.5 \times 10^6$ tonnes of other food available to the whole community at each time step. Preliminary sensitivity tests involving 10-fold changes in other food, and varying its size profile, confirmed that while some outputs from the FE were sensitive to the quantity and representation of other food, these choices led to unreasonable $M2$ values (Figs S1–S3, Sparholt 1990). The modelling approach assumes that the modelled community is closed (dynamics determined solely by internal processes), an assumption that is necessarily violated to some extent in a continuous marine environment.

Recruitment was assumed to be deterministic, with individuals recruiting at the mid-point of the smallest length class. The Ricker spawner–recruit relationship used by Hall *et al.* (2006) was replaced with a hockey-stick spawner–recruit relationship (HS-SRR, Barrowman & Myers 2000). This was changed because the Ricker relationship led to a steeper size spectrum slope (SSS) when unfished than at modest

**Table 1.** Default parameter values and species' abundances for modelled species. Spawning stock biomass (SSB) was obtained from survey data or ICES stock assessments, minimum and maximum estimates of SSB from ICES stock assessments were used to filter model variants. $F_{MSY}$ and $F_{HIST}$ were used in simulations. Lifetype indicates if the species is pelagic (P), semi-pelagic and demersal (D1) or strictly demersal (D2), and $a$ and $b$ are the species-specific 'shape' parameters, enabling conversion from length ($L$, cm) to weight ($W$, g) via the relation $W = aL^b$, $L_\infty$ is the asymptotic length, $L_{mat}$ is the length at 50% maturity and $\kappa$ is the von Bertalanffy growth parameter. Maximum SSB was estimated with ICES stock assessment data if these were available; otherwise, estimates were based on ICES North Sea International Bottom Trawl Survey data

| Common name | Species | Lifetype | $L_\infty$ (cm) | $K$ (year⁻¹) | $L_{mat}$ (cm) | $a$ (×1000) | $b$ | Maximum SSB (t) | Minimum SSB (t) | Maximum SSB (t) | $F_{MSY}$ (year⁻¹) | $F_{HIST}$ (year⁻¹) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Sprat | *Sprattus sprattus* | P | 17 | 0·681 | 12 | 5·9 | 3·109 | 380 236 | 78 083 | 380 236 | 1·30 | 1·06 |
| Norway Pout | *Trisopterus esmarkii* | D1 | 24 | 0·849 | 14 | 7·5 | 3·024 | 462 487 | 63 034 | 462 487 | 0·35 | 0·48 |
| Sandeel | *Ammodytes* spp. | P | 24 | 1·000 | 12 | 4·9 | 2·783 | 1 243 757 | 148 723 | 1 243 757 | 0·35 | 0·57 |
| Poor Cod | *Trisopterus minutus* | D1 | 23 | 0·520 | 15 | 8·2 | 3·087 | 786 976 | – | – | 0·72 | 0·55 |
| Long Rough Dab | *Hippoglossoides platessoides* | D2 | 25 | 0·340 | 15 | 5·3 | 3·143 | 46 140 | – | – | 0·60 | 0·55 |
| Dab | *Limanda limanda* | D2 | 32 | 0·536 | 13 | 15·9 | 2·864 | 276 779 | – | – | 0·41 | 0·55 |
| Herring | *Clupea harengus* | P | 33 | 0·606 | 23 | 4·8 | 3·198 | 2 857 134 | 691 752 | 2 857 134 | 0·25 | 0·25 |
| Horse Mackerel | *Trachurus trachurus* | P | 28 | 0·380 | 19 | 31·6 | 2·652 | 193 771 | – | – | 0·50 | 0·63 |
| Lemon Sole | *Microstomus kitt* | D2 | 37 | 0·420 | 27 | 12·3 | 2·971 | 24 915 | – | – | 0·33 | 0·55 |
| Sole | *Solea solea* | D2 | 47 | 0·284 | 21 | 8·9 | 3·017 | 90 300 | 17 600 | 90 300 | 0·22 | 0·53 |
| Mackerel | *Scomber scombrus* | P | 38 | 0·510 | 26 | 5·2 | 3·167 | 248 438 | – | – | 0·32 | 0·63 |
| Whiting | *Merlangius merlangus* | D2 | 52 | 0·323 | 21 | 9·9 | 2·943 | 527 871 | 181 713 | 527 871 | 0·21 | 0·37 |
| Witch | *Glyptocephalus cynoglossus* | D2 | 44 | 0·200 | 29 | 1·7 | 3·389 | 38 129 | – | – | 0·27 | 0·55 |
| Gurnard | *Eutriglia gurnardus* | D2 | 43 | 0·266 | 18 | 16·0 | 2·815 | 166 957 | – | – | 0·27 | 0·55 |
| Plaice | *Pleuronectes platessa* | D2 | 65 | 0·122 | 22 | 12·5 | 2·943 | 500 800 | 182 000 | 500 800 | 0·25 | 0·55 |
| Starry Ray | *Amblyraja radiata* | D2 | 66 | 0·230 | 46 | 10·7 | 2·940 | 74 341 | – | – | 0·15 | 0·55 |
| Haddock | *Melanogrammus aeglefinus* | D2 | 71 | 0·271 | 27 | 9·2 | 3·013 | 524 367 | 63 074 | 524 367 | 0·30 | 0·58 |
| Cuckoo Ray | *Leucoraja naevus* | D2 | 92 | 0·110 | 59 | 2·6 | 3·217 | 7809 | – | – | 0·11 | 0·55 |
| Monkfish | *Lophius piscatorius* | D2 | 106 | 0·180 | 61 | 29·7 | 2·841 | 15 000 | – | – | 0·10 | 0·55 |
| Cod | *Gadus morhua* | D2 | 150 | 0·216 | 54 | 8·1 | 3·050 | 86 562 | 24 534 | 86 562 | 0·19 | 0·88 |
| Saithe | *Pollachius virens* | D1 | 119 | 0·175 | 49 | 8·5 | 3·024 | 327 700 | 105 400 | 327 700 | 0·30 | 0·38 |

fishing pressures, in contrast with theory and observation (e.g. Rice & Gislason 1996; Bianchi *et al.* 2000; Andersen & Beyer 2006) and because the HS-SRR provided a better fit to ICES data which included few recruitment estimates at relatively high spawner biomass. For the default model parameterization, α (the initial slope of the HS-SRR) was estimated as a function of $L_\infty$ from data (Denney, Jennings & Reynolds 2002), and an upper limit for β, the fraction of maximum spawning stock biomass ($SSB_{max}$) above which recruitment is assumed constant, was estimated by fitting the HS-SRR to ICES spawner and recruit abundance data from single-species stock assessments (ICES 2013). Lower values of β were selected from a linear series of successive 20% reductions in β (Table S1). For each species, we then estimated α from $L_\infty$ and the inflection point in the segmented regression as a fraction of maximum spawner biomass ($SSB_{max}$). $SSB_{max}$ for assessed species was defined as the highest species' SSB reported in ICES stock assessments from 1990 to 2010. For non-assessed species, $SSB_{max}$ was each species' highest SSB as estimated from swept area in the North Sea Quarter 1 International Bottom Trawl Survey (IBTS) from 1990 to 2010 (ICES 2012a). Swept area abundance estimates were corrected for selectivity and availability by species, following the method of Fraser, Greenstreet & Piet (2009).

Natural mortality was divided into non-predation (*M*1) and predation (*M*2) mortality, where *M*2 varied with predator abundance. Total predation mortality imposed by each predator was calculated based on the food requirements needed to support growth, taking account of growth efficiency. Predators consumed prey species defined as suitable in a diet interaction matrix that specifies possible interactions among all species. A predator of a given size could consume individuals of an available prey species within a defined predator: prey size ratio (Hall *et al.* 2006; Table S1). All modelled species also consume individuals of a suitable size in 'other food'. Mortality imposed by predators was partitioned among available prey according to relative prey abundance at size. The default diet interaction matrix from Rochet *et al.* (2011) was supplemented with two variants based on empirical diet data, where predators were allowed to consume prey species if the frequency of occurrence of the prey in diet data was greater than a specified threshold (Tables S1–S4). For two remaining variants of the diet matrix, we assumed that all species ate all other species or that pelagic species only ate pelagic species, and demersal species only ate demersal species.

Fishing mortality was applied assuming species-specific relative size selectivity established from fits to ICES (2012b) *F* at length data. Model scenarios were fishing at MSY ($F_{MSY}$, the fishing mortality that produced the maximum sustainable yield, Table 1, Fig. S4) and fishing with 'historic' fishing patterns ($F_{HIST}$, Table 1, Fig. S4). For the $F_{HIST}$ scenario, the *F* applied to each of the 11 modelled and assessed species was mean *F* for 1990–2010 (ICES 2012b). Non-assessed species were grouped with assessed species of similar morphology and behaviour and were assumed to be fished at the same *F* (Pope *et al.* 2000). Poor cod, gurnard *Eutriglia gurnardus* (Linnaeus) and monkfish *Lophius piscatorius* Linnaeus were grouped with haddock, cod and whiting. Long rough dab *Hippoglossoides platessoides* (Fabricius), dab *Limanda limanda* (Linnaeus), lemon sole *Microstomus kitt* (Walbaum), witch *Glyptocephalus cynoglossus* (Linnaeus), starry ray *Amblyraja radiata* (Donovan) and cuckoo ray *Leucoraja naevus* (Müller & Henle) were grouped with plaice. For the $F_{MSY}$ scenario, $F_{MSY}$ values for assessed species were obtained from ICES (2012b). For the non-assessed species, the *F* leading to 40% spawner-per-recruit ($F_{40}$) was assumed to be a proxy for $F_{MSY}$, and $F_{40}$ values were estimated following Le Quesne & Jennings (2012).

Output from each of the 78 125 model variants was tested against two test criteria to filter out variants with parameter sets leading to unrealistic community properties. Each model variant was run for 45 years with *F* = 0 to predict an unfished equilibrium community state, followed by 30 years fishing at $F_{HIST}$. The first test criterion was that all species should persist in the unfished model system, with biomass exceeding 1% of current biomass estimates from ICES assessments or survey data analyses. The second test criterion was applied after 30 years fishing at $F_{HIST}$. It specified that mean predicted biomass of assessed species during the last 15 years of the $F_{HIST}$ simulation should be more than half the lowest, but less than twice the highest, biomass estimate from 1990 to 2010 (assessment data from ICES 2012b). We used a factor of two because biomass estimates from the single-species assessments are uncertain, and because we adopted a deterministic spawner–recruit relationship when environmental trends and stochastic processes influence recruitment dynamics in the real world. Using the screening criteria, each of the 78 125 model variants was classified as plausible or rejected. Consequences of modifying the factor, dubbed the tolerance factor, from 1·5 to 3·0 were examined (Figs S5, S6 and S7). Selected parameter options in the FE were examined to identify which parameters introduced most uncertainty in model outputs, and to assess whether model dynamics were independent of any given parameter choice.

Accepted model variants, referred to as the FE, were run for 30 years into the future under given fishing scenarios to generate ensemble predictions of the potential response of individual species and community metrics to *F*. For the $F_{MSY}$ scenario, all species were fished at their predicted $F_{MSY}$, and at *F* = 0 to *F* = 5$F_{MSY}$, in 0·1 steps. This set of $F_{MSY}$ scenarios did not allow for examination of yield trade-offs among species, as all *F* were scaled equally (same proportional change in F applied to each modelled species), but was used to provide a measure of the relative magnitude of uncertainty in predictions of the single-species reference points $F_{MSY}$, $B_{MSY}$ (the SSB at $F_{MSY}$) and $Y_{MSY}$ (yield at $F_{MSY}$) derived in a multispecies context.

During simulations, biomass and yield of each species and the value of each community metric were calculated annually. Community metrics calculated were as follows: mean length (ML), biomass fraction >40 cm (dubbed the large fish indicator, LFI), slope of the size spectrum (slope of relationship between log numbers in each log size class and log size, SSS) and mean maximum weight by biomass (biomass-weighted mean of species' $W_\infty$; a measure of the life-history composition of the community (mean maximum weight, MMW, Shin *et al.* 2005). Metrics were calculated using abundance estimates for individuals >15 cm, approximating sizes adequately selected in trawl surveys in the North Sea (Fraser, Greenstreet & Piet 2009). When assessing fishing effects on individual species, we referred to species as collapsed when SSB fell below 10% of unexploited SSB. Outputs of simulations are presented as frequency distributions, treating all variants in the FE as equally plausible. Uncertainty was expressed as 50th and 90th percentile intervals within the FE. The 50% and 90% intervals spanned the 25th to 75th and 5th to 95th percentiles of the frequency distribution of output values generated by the FE, respectively.

We calculated the statistical power of the North Sea IBTS Quarter 1 survey to detect predicted changes in community metrics during a transition from fishing at $F_{HIST}$ to $F_{MSY}$. Statistical power was defined as the probability that the predicted change in metric value achieves a statistical significance (here set at 0·10) over a 5- or 15-year period. As well as the statistical significance and sampling period, power depends on the magnitude of the residual variance, which was estimated by taking residuals from a loess smoother (Fryer & Nicholson 1993, span = 1) fitted to a time series (1990–2011) of metrics calculated from North Sea IBTS Quarter 1 survey data. Metrics were calculated from survey data using only the species and size classes included in the multispecies

model. Power to detect the predicted change in each metric during the transition from fishing at $F_{HIST}$ to $F_{MSY}$ was estimated for each model variant in the FE based on 10 000 random draws from a normal distribution fitted to residuals of the empirical time series.

## Results

Individual species' biomass predictions from the models in the unfiltered ensemble (UE), when fishing according to the $F_{MSY}$ scenario, tended to span 2 or more orders of magnitude (Fig. 1), indicating that initial spread of parameter choices provided little constraint on predictions. Uncertainty in predicted biomass is greatly reduced for the FE where uncertainty is reduced to between one-half and one order of magnitude (Fig. 1). Only 188 of 78 125 possible model variants passed the screening criteria. The default model did not pass the screening criteria. Changing the screening criteria for biomass at $F_{HIST}$ to allow a tolerance factor of 1·5× to 3× for the relationship between ICES single-species biomass estimates and model outputs increased the size of the FE (Fig. S5), but had a small effect on outputs and uncertainty and did not change qualitative behaviour and relative performance of species and community metrics (Figs S6 and S7).

To evaluate impacts of parameter uncertainty on estimates of management reference points for each individual species in the multispecies model, the spread of $F_{MSY}$ and $B_{MSY}$ estimates for all models in the FE was compared with mean $F_{MSY}$ calculated across all models in the FE. For $F_{MSY}$, there was 50% probability that a given value was within ±25% of the mean and 90% probability that it was within ±50% of the mean (Table 2), for all species except plaice. For $B_{MSY}$, there was a 50% probability of predicting single-species values to within ±40%; 90% intervals were much wider, typically spanning a factor of 5–10. Probability distributions of outputs from the FE provided information on uncertainty that could

inform decisions such as how to set $F_{MSY}$ to achieve a defined reduction in the risk that true $F$ exceeds $F_{MSY}$. For example, if $F_{MSY}$ for cod was 0·23, there would be 50% confidence that the actual $F_{MSY}$ limit was not breached. However, if the $F_{MSY}$ limit was set to increase confidence to 75%, or 95%, $F_{MSY}$ would need to be reduced to 0·21 or 0·17, respectively (Table 2).

Model $F_{MSY}$ predictions were consistent with single-species estimates for 12 species, higher for eight and lower for sprat (Fig. 2). $F_{MSY}$ estimates were higher for mid-sized species, the likely explanation being predation release as large fish were depleted at high $F$. Relationships between $B_{MSY}$ and $B_0$ followed theoretical expectations of $B_{MSY} \approx B_0/2$ (e.g. Punt *et al.* 2014).

Community metrics showed qualitatively similar responses to changes in $F$ (Fig. 3), although the SSS showed a near linear decline in response to increasing $F$, while the LFI, ML and MMW displayed a nonlinear relationship with $F$. There was considerable uncertainty in predicted values of the community metrics at $F_{MSY}$ and across the full range of $F$ (Fig. 3).

If a community metric is to provide feedback on effects of management action, then it should show a response to these effects and this should be detectable in monitoring data. This requires that the change in metric value following a defined management action should be greater than uncertainty in prediction, and the predicted change in the metric value should be detectable given interannual variation in data. To compare theoretically expected change in community metrics with uncertainty in prediction, the ratio of the metric at $F_{HIST}$: $F_{MSY}$, for each FE variant, was compared with the 50% and 90% uncertainty intervals for the prediction at $F_{HIST}$, where the 50% and 90% uncertainty intervals were expressed as a proportion of the mean metric value (Fig. 4). The mean change in the values of the metrics fell outside the 50% uncertainty interval for MMW, LFI and SSS, but not for ML. For
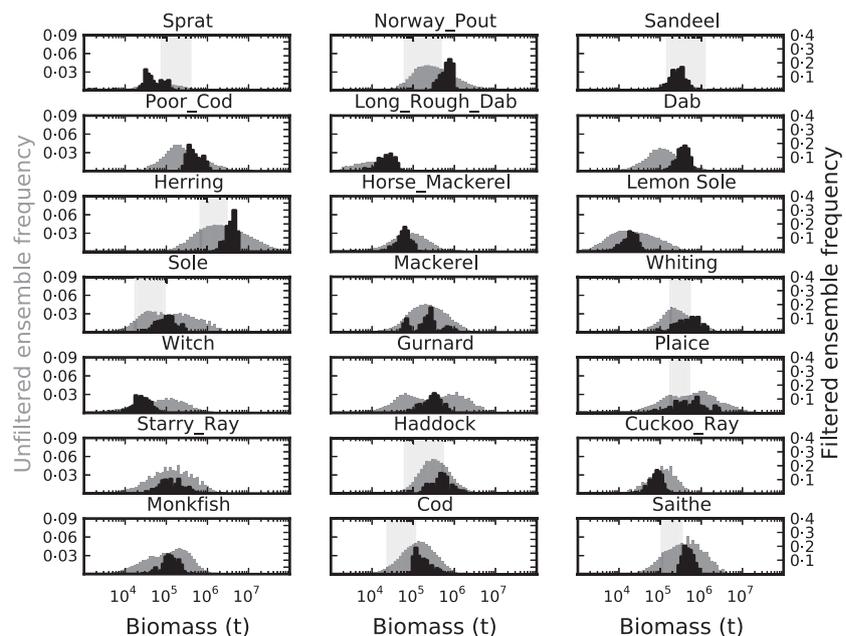


**Fig. 1.** Frequency distribution of estimated biomass of modelled species at $F_{MSY}$. Grey histograms denote the frequency distribution of biomass estimates from models in the unfiltered ensemble (UE), black histograms the biomass estimates from models in the filtered ensemble (FE). The vertical shaded band represents the biomass range reported by ICES for assessed species for years 1990–2010.

**Table 2.** Estimates of single-species fisheries reference points with the filtered ensemble (FE). Mean estimates for $F_{MSY}$ and $B_{MSY}$ are presented as uncertainty intervals (UI) spanning 50% and 90% of the members of the FE. $F_{MSY}$ ratio is the ratio of the $F_{MSY}$ predicted by the multispecies model to estimates of $F_{MSY}$ from ICES single-species stock assessment models

| Common name | $F_{MSY}$ (year$^{-1}$) | 50% UI $F_{MSY}$ (year$^{-1}$) | 90% UI $F_{MSY}$ (year$^{-1}$) | $F_{MSY}$ ratio | $B_{MSY}$ ($\times 10^5$ t) | 50% UI $B_{MSY}$ (proportion) | 90% UI $B_{MSY}$ (proportion) |
|---|---|---|---|---|---|---|---|
| Sprat | 0·78 | 0·65–0·91 | 0·52–1·04 | 0·60 | 1·26 | 0·64–1·32 | 0·46–1·95 |
| Norway Pout | 0·82 | 0·67–0·98 | 0·52–1·16 | 2·35 | 5·07 | 0·76–1·24 | 0·50–1·45 |
| Sandeel | 1·11 | 0·84–1·36 | 0·67–1·57 | 3·18 | 2·13 | 0·69–1·24 | 0·46–1·69 |
| Poor Cod | 0·77 | 0·65–0·86 | 0·50–1·08 | 1·07 | 6·88 | 0·69–1·25 | 0·54–1·66 |
| Long Rough Dab | 0·84 | 0·72–1·02 | 0·54–1·14 | 1·39 | 0·32 | 0·67–1·25 | 0·41–1·69 |
| Dab | 0·85 | 0·70–0·98 | 0·57–1·15 | 2·07 | 3·19 | 0·79–1·19 | 0·53–1·55 |
| Herring | 0·54 | 0·45–0·62 | 0·35–0·73 | 2·17 | 28·3 | 0·82–1·20 | 0·52–1·48 |
| Horse Mackerel | 0·54 | 0·45–0·60 | 0·35–0·80 | 1·07 | 0·80 | 0·64–1·24 | 0·33–1·96 |
| Lemon Sole | 0·48 | 0·36–0·56 | 0·33–0·69 | 1·45 | 0·20 | 0·68–1·29 | 0·40–1·80 |
| Sole | 0·68 | 0·55–0·77 | 0·44–0·99 | 3·08 | 0·90 | 0·62–1·23 | 0·41–2·16 |
| Mackerel | 0·49 | 0·42–0·58 | 0·32–0·67 | 1·55 | 2·59 | 0·47–1·04 | 0·17–3·20 |
| Whiting | 0·75 | 0·59–0·90 | 0·46–1·03 | 3·55 | 4·26 | 0·62–1·23 | 0·39–2·15 |
| Witch | 0·34 | 0·27–0·41 | 0·24–0·49 | 1·27 | 0·28 | 0·57–1·27 | 0·33–2·04 |
| Gurnard | 0·84 | 0·70–0·97 | 0·54–1·19 | 3·12 | 2·81 | 0·65–1·20 | 0·41–1·96 |
| Plaice | 0·31 | 0·23–0·38 | 0·15–0·50 | 1·24 | 8·64 | 0·26–1·06 | 0·13–3·89 |
| Starry Ray | 0·21 | 0·17–0·24 | 0·14–0·26 | 1·38 | 1·49 | 0·42–1·19 | 0·23–2·49 |
| Haddock | 0·48 | 0·36–0·57 | 0·30–0·72 | 1·61 | 4·70 | 0·52–1·19 | 0·26–2·61 |
| Cuckoo Ray | 0·11 | 0·10–0·12 | 0·09–0·13 | 1·02 | 0·86 | 0·70–1·18 | 0·49–1·82 |
| Monkfish | 0·20 | 0·17–0·23 | 0·15–0·25 | 2·01 | 0·90 | 0·65–1·31 | 0·32–1·88 |
| Cod | 0·23 | 0·21–0·27 | 0·17–0·27 | 1·21 | 1·72 | 0·55–1·25 | 0·45–2·44 |
| Saithe | 0·34 | 0·24–0·42 | 0·21–0·45 | 1·13 | 5·54 | 0·69–1·21 | 0·51–1·99 |
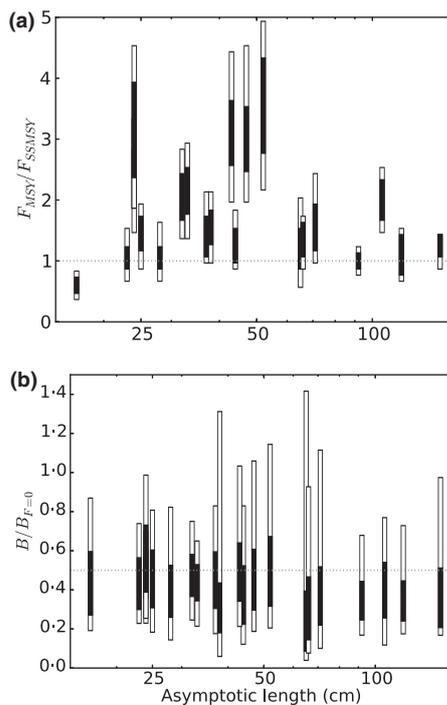


**Fig. 2.** Relationship (a) between $F_{MSY}$ predicted with model variants in the filtered ensemble (FE) as a proportion of $F_{MSY}$ from ICES single-species stock assessment models ($F_{SSMSY}$) and asymptotic length, and (b) between $B_{MSY}$ as a proportion of $B_0$ predicted with model variants in the FE and asymptotic length. Black bars and open bars denote 50% and 90% uncertainty intervals, respectively.

MMW, LFI and SSS, the predicted change in indicator values for the entire FE lay outside the 50% uncertainty interval, and for SSS, predicted change also lay outside the 90% interval.

Thus, SSS is expected to be the most responsive community indicator.

Power of the North Sea IBTS to detect changes in community metrics was assessed by comparing the magnitude of change in mean metric values when model $F$ transitioned from $F_{HIST}$ to $F_{MSY}$ with the magnitude of interannual variation in metric values from IBTS data. For community metrics applied to all demersal species (Table 1, see Lifetype D1), there was a 0·6 to 0·8 mean power of detecting a change in ML, MMW or LFI that occurred over 5 years and a mean power of >0·9 of detecting a change in SSS (Fig. 5). After 15 years simulated monitoring, mean power was still <0·8 for ML, ≈0·9 for MMW, ≈0·95 for the LFI and 1·0 for SSS. For all community metrics except SSS, there was significant variation in power among FE variants after 15 years. For example, for LFI, power ranged from 0·7 to 1·0. Power to predict changes in community metrics for all demersal species was generally a little lower than for strictly demersal species (Table 1, see Lifetype D2), but SSS still provided greatest power after 5 and 15 years (Fig. S8).

Total community yield was maximized when $F \approx 1.6 \times F_{MSY}$ (Fig. 6). Fishing mortality had to be reduced to $F_{MSY}$ or below to avoid all possibility of species' collapse. At $3 \times F_{MSY}$, an approximation of maximum 'historic' $F$ for North Sea target species, mean total yield was predicted to be similar to yield at $F_{MSY}$, or around 85% of the maximum, but two species were expected to collapse, and probability of collapse was >0·1 for another five species. In these fishing simulations, $F$ across all species was increased and decreased evenly, with $F$ ratios among species defined from single-species estimates of $F_{MSY}$. This fishing strategy tends to predict relatively high yields for all species in the community at $F_{MSY}$ (Fig. S9)
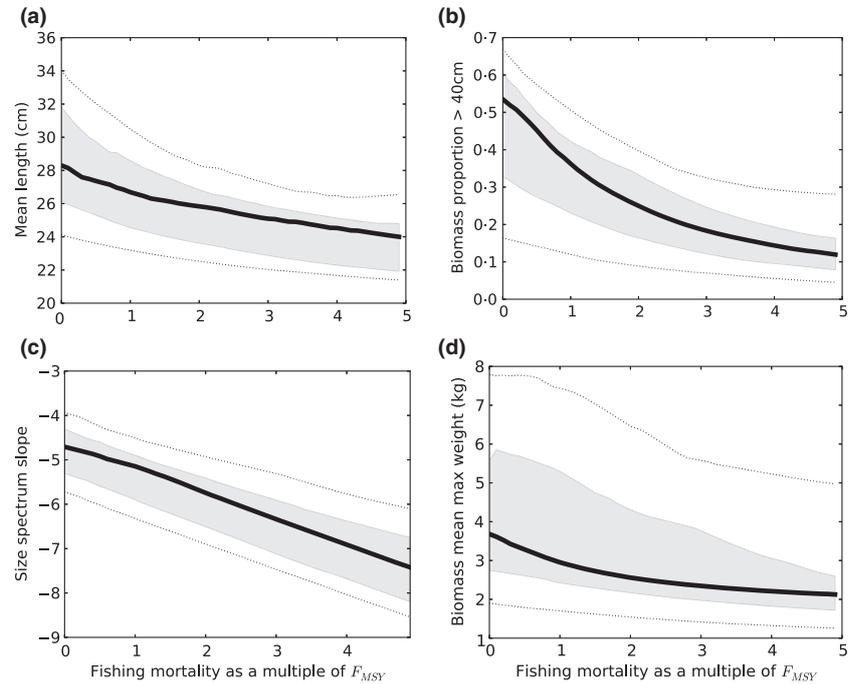
**Fig. 3.** Relationship between the community metrics and fishing mortality predicted by the model variants in the filtered ensemble (FE). Black line is the mean value, and the grey and broken lines denote uncertainty intervals including 50% and 90% of FE members, respectively. Metrics are (a) mean length, (b) proportion of large fish, (c) size spectrum slope and (d) mean maximum weight.
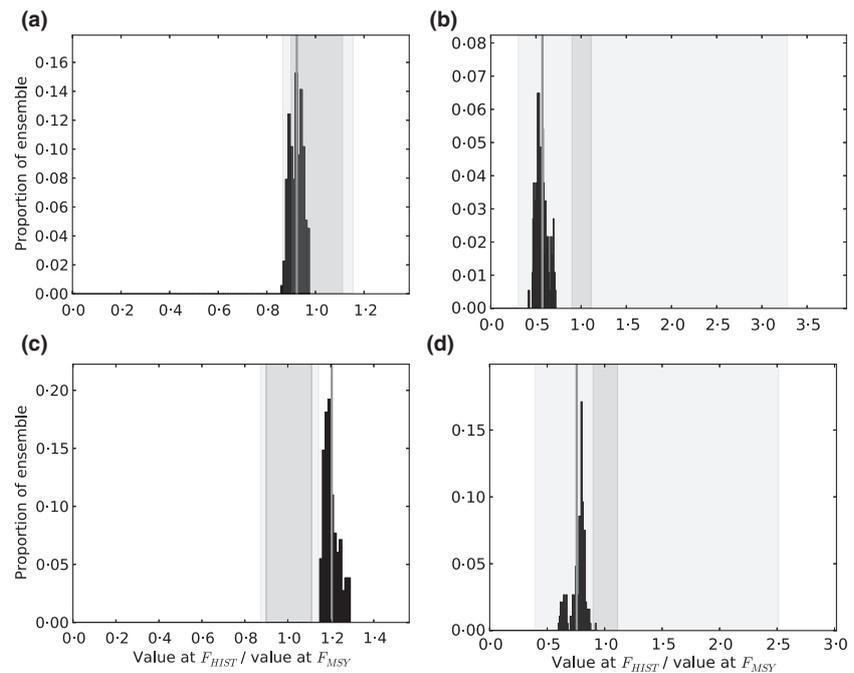


**Fig. 4.** Frequency distribution of community metrics predicted with models in the filtered ensemble (FE) and expressed as the ratio of values of each metric at $F_{HIST}$ and $F_{MSY}$ (black histogram). The vertical grey line is the mean ratio of values of each metric at $F_{HIST}$ and $F_{MSY}$. Vertical shaded bands indicate 50% (dark) and 90% (light) uncertainty intervals for the prediction at $F_{HIST}$, with intervals presented as a proportion of the mean metric value. Metrics are (a) mean length, (b) proportion of large fish, (c) size spectrum slope and (d) mean maximum weight.

and few trade-offs that result from more uneven distributions of $F$ in relation to $F_{MSY}$. The sensitivity of sprat to collapse may be an indication of the differing age at selection applied in our model and the ICES assessment models and/or may reflect relatively high estimates of $F_{MSY}$ in the ICES assessment.

We assessed contributions of each parameter to uncertainty in model outputs (Table 3). Parameters contributing most to overall uncertainty in the UE were $L_\infty$ followed by the diet matrix, where only one option gave successful outcomes. The choices of $M1$, predator–prey size ratio, growth efficiency and HS-SRR parameters all made similar contributions to overall variance in model outputs; in each case, these parameter choices only accounted for about 60% of the variance generated by choice of $L_\infty$. Only the diet matrix of Rochet *et al.* (2011) was incorporated in FE model variants. A single parameter choice was responsible for >50% of successful outcomes for predator–prey size ratio and growth efficiency, while the distribution was more even for $M1$, HS-SRR $\alpha$ and $L_\infty$.

## Discussion

Incorporating plausible parameter uncertainty in model inputs led to many implausible outputs, and screening against data removed >99% of variants. The resulting FE was used to
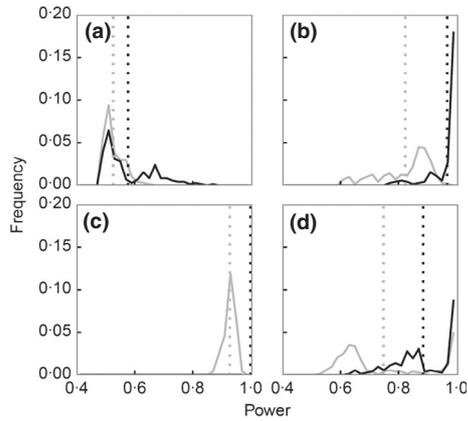
**Fig. 5.** Frequency distributions for the predicted power of the North Sea International Bottom Trawl Survey (IBTS) to detect modelled changes in the values of community metrics for all demersal fishes (Lifetype D1, Table 1) when fishing mortality is reduced from $F_{HIST}$ to $F_{MSY}$ over 5-year (grey) and 15-year (black) periods. Metrics are (a) mean length, (b) proportion of large fish, (c) size spectrum slope and (d) mean maximum weight. Vertical lines denote mean power for all model variants in the filtered ensemble (FE).
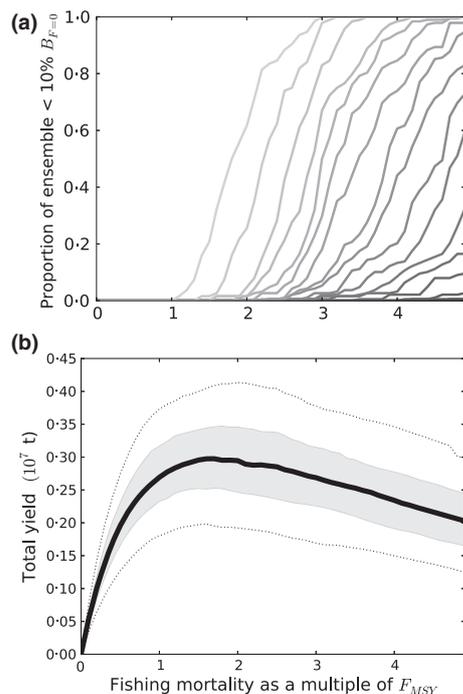


**Fig. 6.** (a) Probability of species' collapse as a function of fishing mortality, with probability expressed as the proportion of variants in the filtered ensemble (FE) which result in collapses ($B < 10\% B_0$). The lightest grey tone indicates probability for the most sensitive species and successively darker tones the probabilities for two or more (up to 14) species. (b) Relationship between mean total yield from the community and fishing mortality. Grey shading and broken lines denote 50% and 90% uncertainty intervals, respectively.

**Table 3.** Proportion of accepted model variants in the filtered ensemble (FE) that incorporated each parameter option, and the relative variance in biomass projections related to each parameter in the unfiltered ensemble (UE). The relative variance describes the variance in biomass projections related to each parameter as a proportion of the variance in biomass projections related to asymptotic length. For parameter choices, see Table S1

| | Parameter choice (FE) | | | | | Relative variance (UE) |
|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | |
| Diet matrix | 1·00 | 0·00 | 0·00 | 0·00 | 0·00 | 0·83 |
| Predation size selection | 0·60 | 0·20 | 0·06 | 0·06 | 0·08 | 0·58 |
| Natural mortality ($M1$) | 0·25 | 0·17 | 0·36 | 0·19 | 0·03 | 0·60 |
| Spawner–recruit ($\beta$) | 0·04 | 0·09 | 0·22 | 0·45 | 0·20 | 0·59 |
| Growth efficiency (Ge) | 0·07 | 0·62 | 0·10 | 0·03 | 0·18 | 0·61 |
| Asymptotic length ($L_\infty$) | 0·15 | 0·08 | 0·16 | 0·23 | 0·38 | 1·00 |
| Spawner–recruit ($\alpha$) | 0·21 | 0·09 | 0·31 | 0·19 | 0·20 | 0·65 |

decisions on the values of reference points to achieve a defined reduction in risk that reference points are not exceeded. Some management frameworks are already designed to take account of information on uncertainty. For example, in the United States, the 'Overfishing Limit' (OL) corresponds to our central estimate of $B_{MSY}$ while the 'Acceptable Biological Catch' (ABC) is set lower than the OL, with the difference between OL and ABC determined by the uncertainty in the OL prediction and the risk that the manager accepts (National Marine Fisheries Service 2009). Outputs from our FE can also be used to assess power of monitoring surveys to detect responses in different community metrics and to identify parameters where investment in future data collection or changes to model structure may be targeted to reduce uncertainty.

Results are clearly contingent on assumptions. In particular, assumed selectivity patterns and relative $F$ among species have a significant influence on estimates of species' abundance, size composition and yield. For simplicity, we scaled $F$ to all species uniformly in relation to $F_{MSY}$. This does not provide an accurate representation of the way $F$ is applied to North Sea stocks, nor did it attempt to reflect a realistic alteration in $F$ in response to management measures, although both issues could be addressed in future. Our preferred emphasis was on evaluating the extent and implications of uncertainty associated with multispecies and community predictions, rather than re-evaluating reference points for North Sea stocks.

Our main measure of indicator performance was sensitivity; the capacity to differentiate responses to changes in $F$. Another relevant measure is the specificity of response to fishing, the extent to which changes in the indicator are driven by fishing as opposed to other causes (e.g. Houle *et al.* 2012). To evaluate specificity would require a model that sought to describe environmental and other influences on the fish community. Our evaluation of the capacity of four community indicators to describe change in state when transiting from historic North Sea $F$ to the current management target of fishing at, or below, $F_{MSY}$ (EU 2013) demonstrated that SSS tracked change with lowest uncertainty. If model predictions of change were realistic then power analysis demonstrated that the NS-IBTS was

assess uncertainty in single species and community reference points and single species and community responses to alternate management actions. Probability distributions of outputs from the FE provided information on uncertainty. This can inform

more likely to detect trends in SSS, as defined here, than any other indicator. Power to detect changes in community indicators for strictly demersal species was better than for all demersal species, likely because the former are more effectively sampled by survey gear (Fraser, Greenstreet & Piet 2009) resulting in lower interannual variation. The more consistent and detectable response of SSS to changes in *F* compared with other community indicators, including the LFI, was consistent with results in Blanchard *et al.* (2014) based on a single parameterization of an alternate multispecies size-structured model that accounted for food-dependent growth and incorporated stochastic recruitment variation. Our power calculations assume that community changes associated with moving from $F_{HIST}$ to $F_{MSY}$ take 5 or 15 years. This allowed relative performance of indicators to be compared. We did not attempt to predict absolute rates of change in community indicators because they are highly dependent on stochastic processes, especially recruitment.

The frequency with which different parameter choices appear in the FE, coupled with relative variance in outputs associated with different parameters, helps identify when investment in refining parameter values may be desirable. Results suggested that the diet matrix was especially influential with neither empirically based diet matrix leading to accepted models. This may be due to limitations of diet studies that may not provide evidence of consumption when some prey species are small, relatively delicate and/or highly digestible, because they quickly become unidentifiable in stomach contents. This could apply to sprat consumption for example. Developing diet matrices likely requires consideration of other evidence for feeding interactions, as attempted by Rochet *et al.* (2011). Further, in models with no explicit representation of spatial structure, the diet matrix also identifies feeding interactions that are not feasible owing to species' spatial separation. The simple assumption that everything eats everything was not supported in our size- and species-based model. Some species require a refuge from predators in the modelled community for their dynamics to mimic those observed in data.

Our conclusions about the effects of parameter uncertainty may not hold for other models with more tightly coupled species dynamics or including food-dependent growth (e.g. Blanchard *et al.* 2014). Further, much density dependence in our model is generated by the HS-SRR which fits quite poorly to data that are already expensive to generate. It is unlikely that more data would reduce our uncertainty about a relationship that has to be used in population models to capture a number of unspecified density-dependent processes that may be impossible to characterize given other sources of variation (e.g. Szuwalski *et al.* 2014). Consequently, the advantages of a model structure that is robust to parameter uncertainty are balanced by disadvantages when model structure omits potentially important processes.

In summary, there have been few attempts to explore effects of uncertainty in multispecies models, which limits interpretation of outputs and their use in management advice. This study has demonstrated one practical method for explicitly introduc-

ing parameter uncertainty in model outputs and describing uncertainty in single species and community reference points and responses to fishing mortality. The research also demonstrates the level of caution that needs to be applied in interpretation of outputs from community models based on single parameter sets.

## Data accessibility

The data used in this article are publicly available via the International Council for the Exploration of the Sea (ICES) Data Portal and from the Centre for Environment, Fisheries and Aquaculture Science (CEFAS) Integrated Database and Portal for Fish Stomach Records (DAPSTOM). More detailed descriptions of data sources are provided in the 'Materials and methods' section of this article.

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## Supporting Information

Additional Supporting Information may be found in the online version of this article.

**Table S1.** Parameter values in the unfiltered ensemble.

**Table S2.** Diet matrix option 1 (Rochet *et al*. 2011).

**Table S3.** Diet matrix option 3 with a cut-off diet fraction by numbers of 0·001 (Pinnegar & Platts, 2011).

**Table S4.** Diet matrix option 4 with a cut-off diet fraction by numbers of 0·01 (Pinnegar & Platts, 2011).

**Table S5.** Diet matrix option 5.

**Fig. S1.** Estimated predation mortality rates ($M2$) in the unfished system based on outputs from model variants in the filtered ensemble (FE) with $10^{10}$ g 'other food'.

**Fig. S2.** Estimated predation mortality rates ($M2$) in the unfished system based on outputs from model variants in the filtered ensemble (FE) with $10^{9}$ g 'other food'.

**Fig. S3.** Estimated predation mortality rates ($M2$) in the unfished system based on model variants in the filtered ensemble (FE) with $10^{11}$ g 'other food'.

**Fig. S4.** Fishing mortality as a function of length for the $F_{HIST}$ (historic average 1990–2010, red line) and $F_{MSY}$ (single species MSY estimates, blue line) scenarios.

**Fig. S5.** The number of models in the filtered ensemble (FE) as a function of the tolerance factor accepted when screening model outputs against ICES stock assessment data.

**Fig. S6.** The effects of changes to the tolerance factor on mean $F_{MSY}$ (red) and associated uncertainty (50% uncertainty intervals: black bars; 90% uncertainty intervals: open bars).

**Fig. S7.** The effects of changes to the tolerance factor on values of four indicators and the associated uncertainty (50% uncertainty intervals: black bars; 90% uncertainty intervals: open bars).

**Fig. S8.** Frequency distributions for the predicted power of the North Sea International Bottom Trawl Survey (IBTS) to detect modelled changes in the values of community metrics for strictly demersal fishes (Lifetype D2, Table 1, accompanying paper) when fishing mortality is reduced from $F_{HIST}$ to $F_{MSY}$ over 5 year (grey) and 15 year (black) periods.

**Fig. S9.** Species' yields as a function of fishing mortality, as predicted with model variants in the filtered ensemble.