

Sustainable fishing can lead to improvements in marine ecosystem status: an ensemble-model forecast of the North Sea ecosystem

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1	Sustainable fishing	can lead to	improvements	in marine	ecosystem	status: an	ensemble-

2 model forecast of the North Sea ecosystem

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- 15 Running page head: Ensemble-modelling of marine indicators
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1 Abstract

2 To effectively implement Ecosystem Based Fisheries Management, tools are needed that are 3 capable of exploring the likely consequences of potential management action on the whole ecosystem. Quantitative modelling tools can be used to explore how ecosystems might 4 5 respond to potential management measures, but no one model can reliably forecast all 6 aspects of future change. To build a robust basis for management advice, a suite of models 7 can be used, but the interpretation of the joint output of multiple models can be difficult. We 8 employ a newly developed ensemble approach to integrate five different ecosystem models 9 and estimate changes in ecosystem state within a single probabilistic forecast. We provide 10 evidence on the response of ecosystem state (measured using ecological indicators relating to plankton, fish and top predators) to potential fisheries management scenarios. We 11 demonstrate that if future fishing mortality is consistent with MSY policy, the North Sea fish 12 community will recover in terms of its size-structure and species composition. However, there 13 14 is currently large uncertainty in trends of future fish biomass, plankton and top predators. We 15 conclude that a) this ensemble approach can be applied directly to policy-relevant questions 16 and add value for decision-makers as multiple aspects of uncertainty are considered; b) future research should prioritise improvements in model skill via a reduction in uncertainty 17 surrounding biomass estimates; c) fisheries management that leads to sustainable fishing 18 19 levels can be considered appropriate for two crucial aspects of fish biodiversity; species 20 composition and size structure.

Key words: ecosystem models, ensemble modelling, indicators, maximum sustainable yield,
 multispecies models, North Sea, uncertainty.

1 **1. Introduction**

2 Many national governments have a policy to implement an ecosystem-based approach to 3 fisheries management (EBFM); and the parties to the Convention on Biological Diversity (www.cbd.int) have agreed Strategic Goals with specific targets to "Reduce the direct 4 pressures on biodiversity and promote sustainable use" and "Improve the status of 5 biodiversity by safeguarding ecosystems, species and genetic diversity". An overarching aim 6 7 is to reduce direct pressures on biodiversity and promote sustainable use of our seas. To 8 measure progress towards these goals, ecological indicators have been developed for 9 national (e.g., HM Government 2019) and regional (OSPAR 2017a b c d, HELCOM 2018) 10 assessments including indicators of pressure (e.g., fishing mortality, noise, contaminants) and state (species, habitats and communities; McQuatters-Gollop et al. 2019). In the European 11 Union, indicator-based assessments also support the policy objectives of the Marine Strategy 12 Framework Directive (European Commission 2008) and the reformed Common Fisheries 13 14 Policy (European Union 2015). In the UK, indicator assessments support the UK Marine Strategy Regulations (HM Government 2010) and the ecosystem-based approach is 15 embedded within the UK Fisheries Act (HM Government 2020), with an explicit objective of 16 achieving good environmental status (GES), where the seas are clean, healthy, safe, 17 productive and biologically diverse (2008/56/EC). To implement EBFM, the indicator 18 19 assessments are followed by a review of monitoring required to capture human impacts on 20 the marine environment and recommendations for a programme of measures to achieve GES. This process is called the indicator assessment cycle (Lynam et al. 2016). 21

Fishing is a major pressure on marine ecosystems (Jennings & Kaiser 1998, Pauly et al. 1998)
and numerous indicators have been developed to describe how changes in state have, and

will, respond to fisheries management scenarios (Fulton et al. 2005, Collie et al. 2013, Coll et 1 2 al. 2016, Rossberg et al. 2017, Fu et al. 2019). Some indicators consider differences in habitat, e.g., relative benthic status (Hiddink et al. 2020), whereas others consider the effect that 3 fishing might have on charismatic species of conservation interest, e.g., seabird breeding 4 5 success (Eerkes-Medrano et al. 2017). Here, we are particularly interested in changes in 6 ecosystem structure and function, specifically functional group indicators, as they provide 7 integrative measures that can capture the wider effects of fisheries on ecosystems (including 8 indirect impacts that may propagate up through the food web, e.g., to piscivorous seabirds and marine mammals) and of ecosystem change on fisheries (e.g., through changes in the 9 biomass of plankton at the base of food web; Tam et al. 2017). Community level indicators of 10 11 size structure (Bianchi et al. 2000, Greenstreet et al. 2012, Brose et al. 2017, Tu et al. 2018) and species composition (Fisher et al. 2010, Houle et al. 2012, Bell et al. 2018) have previously 12 13 been shown to respond to fishing pressure. These ecological indicators have been 14 incorporated into assessments by OSPAR and, within the UK, they are used to monitor changes in food webs and biodiversity, alongside stock-specific information for commercial 15 fish and shellfish (see UK Marine Online Assessment Tool, UKMMAS 2019). We consider a 16 selection of these indicators of GES for which current published ecosystem models are able 17 to inform on: i.e., the biomass of zooplankton, the ratio of phytoplankton to zooplankton 18 19 biomass, the biomass of fish and top predators (birds and mammals), along with measures of 20 species composition and size structure within the fish community.

Within the indicator assessment cycle, modelling should be employed to assess the performance of indicators in relation to pressure and forecast any likely change in state following the implementation of existing and/or planned management measures (Lynam et al. 2016). For example, limiting fishing opportunities (catch quotas or days at sea) in order to

achieve maximum sustainable yield (MSY), the highest yield achievable in the long-term while 1 2 maintaining reproductive capacity of the stock (ICES 2021). Several possible ecosystem models could, and have, been used for this task. For example, Blanchard et al. (2014) used a 3 size-based model to investigate how length-based indicators (including the Large Fish 4 5 Indicator, LFI) respond to past and future fishing pressure. Speirs et al. (2016) demonstrated that fishing at MSY could facilitate recovery in LFI to levels that represent GES in the North 6 7 Sea, where the indicator target was defined by Greenstreet et al. (2011). However, Queirós 8 et al. (2018) also found through spatial and size-based modelling that any recovery in LFI would be limited by climate effects on fish body size. Additionally, Ecopath with Ecosim has 9 been employed to test the relevance of trophic indicators (including the slope of the biomass 10 11 spectrum) to EBFM in numerous ecosystems (Bourdaud et al. 2016). Despite such efforts, it is important to note that individual ecosystem models, despite being built with similar 12 13 processes and fitted to similar data, can often produce conflicting predictions (Plagányi 2007, 14 Collie et al. 2016, Tittensor et al. 2017, Rosenberg et al. 2018). Consequently, the predicted outcome of any management decision under consideration is sensitive to the model chosen. 15 This sensitivity to model choice, coupled with the inherent uncertainty in modelling the 16 17 complex dynamics of whole ecosystems has, in part, led to a lack of uptake of ecosystem models in the advice process (Hyder et al. 2015). 18

When giving advice it is important to quantify uncertainty in predictions in a robust manner (Harwood & Stokes 2003). Uncertainty within individual ecosystem models has previously been explored (e.g., Thorpe et al. 2015, Spence et al. 2016, 2021, Mackinson et al. 2018), however, these attempts are likely to underestimate the uncertainty, as uncertainty across models is often ignored. Management advice should ideally consider the outputs of a suite of models, however, combining them is not as easy as averaging over their outputs or ascribing

a simple weight to each as we do not expect the discrepancies of each of the models to be 1 2 independent (Rougier 2007, Christiansen 2020). Further, each model's relative utility is often question-sensitive (Dickey-Collas et al. 2014), and no such model will be uniformly better than 3 the rest (Chandler 2013). To counteract this problem, Spence et al. (2018) developed an 4 5 ensemble model that synthesises empirical and modelling studies to give a single probabilistic forecast with robust quantification of uncertainty, however, this approach is yet to be 6 employed to investigate indicators of pressure or ecological state. To supplement our 7 8 approach, we also assess 'model skill' (here using information), defined as the inverse of uncertainty, as a metric. This is an emergent feature in the recent literature (e.g., Olsen et al. 9 2016, Geary et al. 2020) and provides a useful narrative via which this comparison of 10 11 uncertainty can be made.

In this study, we address an explicit policy question that follows from the OSPAR 2017 (OSPAR 12 2017a b c d, HELCOM 2018) and UK 2019 assessments of biodiversity (HM Government 2019), 13 14 namely: will fisheries management measures that aim to maintain productive commercial 15 fisheries also lead to improvements across the ecosystem? The long-term impact of fishing on an ecosystem includes the depletion in the proportion of large or old individuals 16 (Christensen 1996, Pauly et al. 1998, Greenstreet et al. 2011), a reduction in the biomass of 17 fished populations (Palomares et al. 2020) and subsequent declines in the abundance of apex 18 predators (mammals, birds, large fish and elasmobranch species) that are dependent on the 19 20 availability of prey (Lynam et al. 2017). A recovery in size-structure (using typical length 21 and/or LFI) is considered important for food web functioning as large individuals are typically effective spawners (Barneche et al. 2018) and top predators play an important role in 22 23 stabilising ecosystems (Casini et al. 2012, Lynam et al. 2017, Landi et al. 2018). A simultaneous 24 recovery in size-structure and species composition, as measured by mean maximum length

(MML), is considered a strong measure of improved biodiversity since species with a high 1 2 maximum body length are typically slow-growing and most vulnerable to fishing impacts 3 (Andersen 2019). So, a question that arises from managers seeking an improved ecosystem state is: will fish communities (as measured by their size-structure and species composition) 4 5 and the biomass of birds and mammals improve as stocks are fished at their target mortalities, 6 or will more restrictive mortality targets be required? Similarly, a related question posed is: will an increase in fish biomass lead to a depletion in zooplankton production or is the 7 8 ecosystem productive enough to allow species to recover? To provide an integrative 9 understanding of the likely impacts of fisheries management on the North Sea ecosystem, we investigate changes in the aforementioned indicators of GES through the ensemble modelling 10 approach of Spence et al. (2018). We consider a range of management scenarios (including 11 fishing at single species or multi-species targets for MSY), five different ecosystem models 12 13 (Ecopath with EcoSim, mizer, FishSUMS, StrathE2E and LeMans) and an empirical analysis of 14 the indicators of interest, all within a single ensemble model. This approach allows us to (a) make use of information from numerous ecosystems models; (b) identify the likely outcomes 15 of each fisheries management scenario with robust and quantifiable uncertainty; (c) discuss 16 how changes in fishing pressure will influence ecosystem structure and function; (d) assess 17 the model skill of the suite of ecosystem models and make targeted recommendations for 18 model development. 19

- 20 2. Materials & Methods
- 21 **2.1.** North Sea

The North Sea is a semi-enclosed sea situated on the continental shelf of North WesternEurope. Many offshore fisheries operate in the North Sea and are regulated in accordance

with the EU Common Fisheries Policy (European Union 2015) and by Norway and the UK 1 2 through coastal state agreements. The environmental policy for the North Sea is managed by 3 national governments and OSPAR (e.g., OSPAR 2009). Demersal fishing effort (beam trawling 4 and demersal otter trawling, Couce et al. 2020) has been reduced since the 2002 Common 5 Fisheries Policy reforms (European Union 2015). Since then, a recovering trend in the 6 demersal fish community has been observed (Engelhard et al. 2015), however, there is substantially less food (through a reduction in discards) for surface feeding and scavenging 7 8 seabirds (Sherley et al. 2020).

9 **2.2.** Indicators of state

10 The following indicators of GES were considered:

Total biomass of marine birds and mammals (*BM*, tonnes). *BM* will respond to fishing
 indirectly via shifts in food availability and/or directly due to mortality caused by human
 activities.

Total biomass of the demersal fish community (*TFB*, tonnes). *TFB* will be impacted directly
 by ecosystem productivity and fishing pressure.

Size-structure of the demersal fish community was measured using the large fish indicator
 (*LFI*), which is the proportion of biomass in the community that is composed of individuals
 larger than a set threshold: here 40 cm, i.e.,

19
$$LFI = \frac{\sum_{i=1}^{n} B_i I(l_i \ge 40)}{\sum_{i=1}^{n} B_i}$$
(1)

This indicator has been used for many years in the ICES and OSPAR communities (ICES 2009, OSPAR 2009, 2017c) and was developed to be responsive to fishing pressure. *LFI* will decline as large individuals are fished out of the ecosystem (Greenstreet et al. 2011).

Size-structure within the whole fish community was investigated through the Typical
 Length indicator (*TyL*, cm, the geometric mean length of fish weighted by biomass; OSPAR
 2017d), which has been suggested to have more favourable statistical qualities than the
 ratio indicator *LFI* (ICES 2014). The typical length indicator is

$$TyL = \exp\left[\frac{\sum_{i=1}^{n} B_i \log(l_i)}{\sum_{i=1}^{n} B_i}\right]$$
(2)

5

6 where B_i is the biomass and l_i is the length (cm) of the *i*th fish in a sample of *n* fish. *TyL* 7 will rise and fall as the size-structure responds to fishing pressure and other pressures 8 including climate change impacts (Houle et al. 2012, Tu et al. 2018).

9 5. Species-composition within the whole fish community was measured using the Mean
10 Maximum Length indicator (*MML*, cm). The Mean Maximum Length indicator is calculated
11 as:

12
$$MML = \frac{\sum_{j=1}^{m} B_j L_{inf,j}}{\sum_{j=1}^{m} B_j}$$
(3)

where B_j is the biomass of each species j for a community of m species and $L_{inf,j}$ is the 13 growth-model asymptote known as length at infinity. MML will decrease as communities 14 are fished down (Houle et al. 2012). The rationale for this is that large, slow-growing, slow 15 to mature species are more sensitive to fishing pressure impacts than smaller-bodied, 16 rapidly growing, fast to mature species. Under the "fishing-down-the-food-web" 17 18 hypothesis (Christensen 1996, Pauly et al. 1998), larger bodied species (typically more 19 valuable species) are targeted preferentially by fisheries, which would lead to a direct decline in *MML* and the size-structure of the fish community. For the model studies $L_{inf,j}$ 20 is specific to each model and for the empirical analysis estimates were taken from 21 22 FishBase (Froese & Pauly 2021).

The ratio of zooplankton to phytoplankton biomass (*Z:P*, dimensionless). *Z:P* will respond
 to top-down cascades when they are strong (Rossberg et al. 2019) and may also respond
 to climate change (OSPAR 2017a). We expect *Z:P* to respond indirectly to fishing pressure
 via the recruitment of fish larvae and the consumption of zooplankton by
 zooplanktivorous fish (Gorokhova et al. 2016).

7. Total zooplankton biomass (*ZB*, tonnes). *ZB* represents the transmission of bottom-up
effects, e.g., related to nutrient concentrations, to zooplanktivorous fish (Gorokhova et
al. 2016). We expect *ZB* to respond directly to ecosystem productivity and indirectly to
fishing pressure as the numbers of potential predators in the ecosystem change.

The historical change in each indicator was examined from 1986 to 2013 based on available data and/or empirical studies (see Section 2.3). From 2014 to 2050, each indicator was forecasted using all available ecosystem models (not all ecosystem models were capable of forecasting each indicator) and the fisheries management scenario under consideration.

14

2.3.

Empirical studies

Continuous Plankton Recorder (CPR) data for zooplankton (abundance of copepods; 15 individuals per n.mi²) and phytoplankton (total counts of diatoms and dinoflagellates; 16 individuals per n.mi²) were taken from the study of Lynam et al. (2013). Z:P was calculated by 17 division of annual records for each component. ZB was determined from abundance data for 18 29 taxa using wet weight factors and corrections for under-sampling of the CPR relative to 19 vertical towed nets (WP-2), using the conversion factors of Pitois & Fox (2006) (see 20 21 Supplement 1, Table S1). We modelled the seasonality and long-term change in ZB using a 22 generalized additive model (see Supplement 1).

The *MML* for all species, *TyL* for all species, *TFB* and *LFI* were calculated from the International 1 Bottom Trawl Survey (IBTS) data from Quarter 1 (January – March). Processed data were 2 3 obtained from the quality assured OSPAR Groundfish Survey Monitoring and Assessment Data Product for the northeast Atlantic area (Greenstreet & Moriarty 2017, Moriarty et al. 4 5 2017), which is based on national datasets uploaded to the ICES DATRAS database. Walker et 6 al. (2017) developed catchability correction factors for species and size classes of fish in the IBTS and we apply them here to better estimate fish biomass and community indicators. A 7 8 second order polynomial regression was fitted to log transformed MML, TyL and TFB to smooth the data and calculate uncertainty values (see Supplement 2 for more details). 9 Similarly, a second order polynomial regression was fitted to the logistic transform of the LFI. 10 BM was taken from species distribution model predictions based on ~2 million kilometres of 11 at-sea cetacean and seabird surveys between 1985 and 2015 (Waggitt et al. 2020; see 12 Supplement 1). 13

14

2.4.

Future management scenarios

The likely change in the ecosystem state (as measured by the indicators described in Section
2.2) was forecasted to the year 2050 under four different fishing management scenarios:

Status quo: continue to fish at the 2013 fishing mortality rates for all stocks (Table 1) as
 defined by Thorpe et al. (2017).

Maximum sustainable yield (MSY): fish 21 species at the fishing mortality rates
 recommend by ICES for obtaining MSY, while all other species are fished at 2013 levels.
 MSY is the maximum yield of a single species that can be sustainably caught and is
 calculated on a species by species basic. All values are taken from Thorpe et al. (2017).

See Supplement 3 for the specific definition of MSY and Table 1 for species-specific fishing
 mortality rates.

3. Nash equilibrium (Nash): fish 21 species at mortality rates according to the Nash
 equilibrium solution computed by Thorpe et al. (2017), while all other species are fished
 at 2013 levels. The Nash equilibrium is a potential multispecies MSY, where each of the
 individual species are simultaneously at their respective single species MSYs. See
 Supplement 3 for the specific definition of the Nash equilibrium and Table 1 for species specific fishing mortality.

9 4. Closure (termed no fishing throughout): no fishing at all

Each scenario was implemented from the year 2014. For more information on the definitionsof each scenario please see Supplement 3.

12 **2.5.** Ecosystem models

Management effects on the indicators were predicted using the five ecosystem models that
were available to us. They were:

15 1. Ecopath with EcoSim (EwE) is a whole ecosystem model that was first developed in 1984 by Polovina (1984) and has been updated since to include temporal (Ecosim) and spatial 16 (Ecospace) dynamics (Christensen & Walters 2004). Ecopath is currently used extensively 17 across the globe to simulate historic changes in ecosystems and their response to 18 pressures (Heymans et al. 2016). The EwE model used here is the North Sea model 19 reviewed by ICES (2016). It contains > 10 fishing fleets and > 60 functional groups, some 20 21 of which are split into multiple age stanzas. We used parameter uncertainty from Mackinson et al. (2018). EwE was able to predict ZB, Z:P, BM and TFB from 1991 until 22 23 2050.

2. The multispecies size spectrum model (mizer) was developed to represent the size and 1 2 abundance of all organisms from zooplankton to large fish predators in a size-structured food web. A proportion of the organisms are represented by species-specific traits and 3 body size while others are represented solely by body size. The core of the model involves 4 5 ontogenetic feeding and growth, mortality, and reproduction driven by size-dependent predation and maturation processes (Hartvig et al. 2011, Scott et al. 2014). Blanchard et 6 al. (2014) developed and applied a version of mizer for the North Sea that is used here. 7 8 Mizer was able to predict MML, TyL, LFI and TFB from 1986 until 2050.

3. FishSUMs (Speirs et al. 2010, 2016) represents the population dynamics of a set of key 9 tropically-linked predator and prey species. For each species the state variables are 10 biomass by length class. In discrete time steps the state variables are updated through 11 increasing length, density-dependent mortality, and losses due to fishing and predation 12 13 by explicitly modelled species, and seasonal reproduction. Additional food resources, not 14 modelled at the species level, are characterised by three biomass spectra representing zooplankton, benthos, and "other fish". The model was initially configured for the North 15 Sea with a set of nine structured species (Speirs et al. 2010), and subsequently extended 16 to include plaice and saithe so as to include the eight most abundant demersal species 17 that make up > 90% of the North Sea biomass (Speirs et al. 2016). FishSUMs was able to 18 predict MML, TyL, LFI and TFB from 1986 until 2050. 19

The Strathclyde end-to-end (StrathE2E) marine food web model is a whole ecosystem
 model designed to simulate regional scale, macroscopic top-down and bottom-up
 cascading trophic effects (Heath et al. 2014). The mathematical formulation is based on a
 network of coupled ordinary differential equations representing the entire food web in
 the water column and seabed sediments from nutrients and microbes though

zooplankton and fish, to birds and mammals, including the effects of advection, mixing
 and active vertical migrations. Living components are represented at low taxonomic
 resolution, focussing on fluxes of nitrogen between coarse functional groups, and
 simulating the general "shape" of the food web rather than the detail. StrathE2E was able
 to predict *ZB*, *Z:P* and *BM* from 1991 until 2050.

5. LeMans is a modified form of the length-based multispecies model initially developed by
Hall et al. (2006) to represent the Georges Bank fish community, which was subsequently
adapted for use in the North Sea by Rochet et al. (2011). The model represents 21 fish
species in 32 equal length classes of around 5 cm each, spanning the full-size range of
species represented into the model (Thorpe et al. 2015, Spence et al. 2020). LeMans was
able to predict *MML*, *TyL*, *LFI* and *TFB* from 1986 until 2050.

Each model was run with historical fishing mortalities applied annually until 2013 and then the four future scenarios were implemented until 2050. From 2014-2050, fishing mortality rates were fixed according to the management strategies described above (see Section 2.3 and Table 1). A summary of each ecosystem model and their outputs with respect to this study can be found in Supplement 4.

17 **2.6. Ensemble model**

Following the approach of Spence et al. (2018), the five ecosystem models (see Section 2.5) were combined with empirical studies (see Section 2.3) in an ensemble model. The ensemble model is a statistical model that describes the structure of the suite of models and the empirical studies. It separates model discrepancies (or biases) between "shared discrepancies", i.e., discrepancies shared between models, and "individual discrepancies", which are specific to each model. It combines the model outputs and discrepancies with

probabilistic estimates from the empirical studies to generate a prediction of the true value
 of the indicators, with quantifiable uncertainty. A more thorough description of the ensemble
 model is provided in Supplement 5.

4 The ensemble model was fitted using a Bayesian framework in order to quantify the uncertainty (Spence et al. 2018). For prior specification see Supplement 5. The likelihood of 5 6 the ensemble model is intractable and so we sampled from the posterior distribution using 7 Markov Chain Monte Carlo (MCMC). Due to the high dimensionality and correlation of the 8 uncertain parameter space, we fitted the model using No U-turn Hamiltonian Monte Carlo (Hoffman & Gelman 2014) in the package Stan (version 2.21.2, Carpenter et al. 2017) in R 9 (version 4.0.2, R Core Team 2020). The ensemble model was fitted by running the MCMC 10 algorithm for 2000 iterations. The first 1000 iterations were discarded as a burn-in. 11

12 **3. Results**

13 **3.1.**

. Ecosystem model runs

The ability of each ecosystem model to follow the empirical studies and then forecast under an MSY scenario is illustrated in Figure 1 (see Supplement 6 for alternative scenarios). Only two of the models (EwE and StrathE2E) were able to estimate *ZB*, *Z:P* and *BM* and both were unable to replicate the historical trends. There is also substantial disagreement between models when forecasting, both in terms of trends and absolute values. StratheE2E predicted an increase in *ZB* until 2050 whereas EwE did not. Similarly, *BM* and *Z:P* were estimated to increase (albeit slightly) by EwE, whereas StrathE2E estimated no change.

Three models (mizer, FishSUMS and LeMans) were able to estimate *TyL* and *LFI*. All three models followed historical trends, with both *TyL* and *LFI* increasing over 1986-2013. This increase was then accelerated, followed by a plateau in all three model forecasts. Alongside

TyL and *LFI*, mizer, fishSUMS and LeMans, as well as EwE, were able to estimate changes in
species composition through *MML*. Trends in *MML* during the historic period did vary
between models, however, they all suggested some degree of increase. This increase of *MML*is analogous to the empirical studies despite large uncertainties. All four models then
estimated further increases in *MML* under an MSY scenario.

All five models were able to estimate changes in *TFB*. While they agreed unanimously that *TFB* has increased during the historic period, this trend was not reflected in the highly
uncertain empirical studies. When forecasting to 2050 under a MSY scenario, StrathE2E
projected an increase whereas the other four model estimated little additional change.

10 **3.2.** Uncertainty quantification

The ensemble model predictions for each of the seven indicators are illustrated in Figure 2. *MML, TyL* and *LFI* were forecasted to increase and then plateau under an MSY scenario. The uncertainty for these indicators was low throughout and did not grow appreciably over time. In comparison, forecasts for *BM, TFB, ZB* and *Z:P* were much more uncertain. In *BM* and *TFB* this uncertainty increased into the future, whereas for *ZB* and *Z:P* the uncertainty was large throughout. These patterns were consistent across scenarios (see Supplement 6).

17 **3.3.** Scenario comparison

In general, the MSY and Nash scenarios produced similar results for all seven indicators (see Figures 2 and 3 and Supplement 6, Figure S11) so here we focus solely on MSY scenarios. For *MML, TyL, LFI* and *TFB*, the no fishing scenario resulted in the largest absolute indicator values, followed by MSY and then status-quo (Figures 3 and 4). The species composition and size-structure of the fish community should recover under fishing scenarios (MSY and Nash),

with MSY leading to an expectation of a 5%, 19% and 8% smaller *MML*, *LFI*, and *TyL* respectively, relative to the unfished state (Figure 4). *BM* was predicted to be higher under the status-quo and no fishing scenarios than it was for MSY. For *ZB* and *Z:P*, the estimates were associated with large uncertainty independent of the scenario and we are unable to say conclusively which scenario would lead to the largest increases. That said, we estimate that there is a 0.985 and 0.986 probability that the MSY and Nash scenarios will lead to a higher *Z:P* ratio than under the status quo scenario.

8 **4.** Discussion

9 Marine ecosystems face a range of pressures from climate change to the cumulative effects 10 of a growing range of human activities, such as shipping, pollution, or the renewable energy sector (Willsteed et al. 2018). Given our limited understanding on the resilience or 11 vulnerability of ecosystem function, a risk averse strategy would be to minimise human 12 13 impacts and allow ecosystems to recover to natural levels. However, the socio-economic 14 demands of society also require us to maintain sustainable use of the marine environment. 15 Consequently, biodiversity assessments have been developed to monitor ecological indicators which are designed to track changes in ecosystem state at a range of hierarchical 16 levels (species, communities, and ecosystems). Our study compares strategies for sustainable 17 fishing to demonstrate whether these strategies are likely to lead to decreases in ecological 18 19 risk. Specifically, this work addresses an explicit policy question that follows from the OSPAR 20 2017 (OSPAR 2017a b c d, HELCOM 2018) and UK 2019 assessments of biodiversity (HM 21 Government 2019), namely: will fisheries management measures lead to improvements in 22 fish communities (size-structure and species composition) and the biomass of birds and 23 mammals?

1 The intermediate assessment of OSPAR (2017c d) suggested that the size structure (LFI and TyL) of demersal fish communities should continue to improve if current fishing levels in the 2 North Sea are maintained and our results confirm that this should be the case in each 3 management scenario. In contrast, empirical estimates by OSPAR (2017b) of MML (measured 4 5 separately for pelagic and demersal communities) indicate minimal recovery based on survey 6 estimated relative abundance. Here, we modelled the absolute biomass of the whole fish 7 community, not spilt into pelagic and demersal communities, and demonstrate that MML and 8 *TyL* should increase in each management scenario. The biomass of birds and mammals should also improve, but we do not expect it to reach the levels seen in the early 1990s. Further, all 9 10 of the scenarios should lead to improvements in TFB and BM. An aim of EBFM is to support top predators while maintaining productive fisheries (Hill et al. 2020). We show here that 11 fishing on pelagic fish at lower levels than MSY (and Nash), as is the case in status quo and no 12 13 fishing, will probably lead to greater improvement in BM and TFB. This demonstrates a trade-14 off between yield and biodiversity, which is important evidence to inform potential management action. 15

16 The no-fishing scenario demonstrates the recovery potential of the ecosystem, assuming 17 constant productivity, from the current state and so provides a basis for the quantitative 18 assessment of GES (sensu Rossberg et al. (2017)). The forecast values of each indicator, with their uncertainty, provides an estimate of the 'natural range' of each indicator when the 19 20 ecosystem is unperturbed. Quantitative targets for indicators can then be made relative to 21 this 'natural range' and this is a particularly useful approach when associated with estimates of temporal responsiveness of change (Houle et al. 2012, Collie et al. 2013, Rossberg et al. 22 23 2017). We suggest that this model-based approach is preferable to the simpler empirical 24 approach in which indicators are compared to historic baselines that may no longer be

appropriate (McQuatters-Gollop et al. 2019). Nevertheless, long-term change in abiotic
conditions can alter the recovery potential of an ecosystem (for example due to a decrease in
ecosystem productivity as suggested by Marshall et al. (2016) and Capuzzo et al. (2018)).
Consequently, indicator targets should be iteratively re-evaluated as part of the assessment
cycle (Lynam et al. 2016). Similarly, once an indicator target is accepted and management
measures to reach the target are considered, then the social and economic effects of the
measures should also be considered within the assessment cycle (Lynam et al. 2016).

8 Ensemble model

9 Previous investigations into management effects of ecosystem indicators have used only one ecosystem model per environment and could not account for uncertainty within model 10 projections, potential systematic differences between different modelling approaches or 11 quantify the confidence they have in their prediction (Fu et al. 2019). As a result, the findings 12 of a study can be sensitive to the choice of ecosystem model. As no one model is uniformly 13 14 better that the rest (Chandler 2013) and indeed all are subject to some degree of structural 15 error, we used a suite of models, combined using the ensemble model of Spence et al. (2018), to generate projections of management effects on ecosystem indicators. This model 16 17 synthesised information from empirical and modelling studies, exploiting the strengths, while discounting the weaknesses (Chandler 2013), to give probabilistic estimates of the indicators 18 19 of interest. It integrated multiple sources of uncertainty, including observational, parameter 20 and model uncertainty, to allow a much more robust assessment of risk which is important 21 when giving management advice (Harwood & Stokes 2003).

In this study we used five ecosystem models to investigate the effect of different
 management strategies on seven indicators of GES. These ecosystem models were chosen as

they were the only models available to us at the time, and all gave some information about the true state of the ecosystem, both in the past and in the future, under the different fishing scenarios. If more models were available to us (e.g., an Atlantis model, Audzijonyte et al. 2019), there would be more information about the true state of the system and therefore we would have included it in the ensemble study, further reducing the uncertainty of the true value of the indicator and increasing the skill of the suite of models.

7 In model weighting schemes, e.g., Bayesian model averaging (e.g., Banner & Higgs 2017, 8 Dormann et al. 2018), or where a single model is used, the uncertainty is limited to the models in the suite, regardless of whether they are good at describing the indicator or not. However, 9 10 the uncertainty from the ensemble model of Spence et al. (2018) is dependent on the ability of the models to represent the indicators. This is evident when looking at the LFI, where the 11 12 relative changes in uncertainty for the 2013 and 2050 predictions are small when compared to the TFB, where the relative uncertainty is larger. This is despite only 3 of the models being 13 14 able to predict LFI but all the models being able to predict TFB. The ensemble model provides 15 a way of combining multiple sources of information, both modelling and empirical studies, to predict the outcomes of management scenarios with greater certainty. If managers can be 16 more confident in the outcome of their actions, then they can make better decisions (Zhang 17 2006). 18

19 Model skill

The uncertainty from the ensemble model can be considered a metric to assess the skill of the suite of models as a collective, with higher uncertainty indicating lower skill. In this study we found that the models were relatively good at predicting the length-based indicators, as the uncertainty remained relatively constant into the future, suggesting higher levels of skill.

When compared to the empirical studies, the three size-based models followed the trends of 1 2 each length-based indicator well, despite errors in magnitude. Models with predictable errors 3 are often more useful than ones with less. For example, a model that predicts the trends of 4 the system right, despite getting the absolute values wrong, may be more useful, and have 5 greater "model skill", than one that gets the absolute value right, but the trends completely 6 wrong (Spence et al. 2018). Size-based theory has a long history (Sheldon et al. 1972) and the 7 mechanisms that govern growth and size structure are well understood and supported by 8 data (Andersen 2019). The size-based models, mizer, FishSUMs and LeMans, explicitly describe the fish community, with the rest of the ecosystem being implicit, and are all based, 9 10 to varying degrees, on these mechanisms. On the other hand, EwE and StrathE2E are whole ecosystem models, with the fish community embedded in the wider ecosystem. In these 11 models the processes that act on the size-structure of the fish community are less detailed, 12 13 suggesting they cannot directly inform the size-based indicators. Instead, they ought to better 14 represent flows of energy in and out of the fish community, which allows one to answer questions about the impact of the whole ecosystem, rather than just the fish community. 15

For the higher and lower trophic level indicators, the uncertainty is much larger. This is 16 17 because the models' predictions of zooplankton, phytoplankton, birds and mammals neither reproduced the past adequately, nor agreed with each other when forecasting forwards. In 18 these circumstances, the uncertainty is large, and the forecasts disagree with each other, so 19 20 the ensemble has lower predictive skill. The lack of skill in the ensemble model for indicators 21 describing the top and bottom of the food web is somewhat understandable. For example, for birds and mammals, their large spatial distribution, migration, and complex behaviour, are 22 23 less well understood than fish growth (e.g., Newton 2005, Hays et al. 2016). Furthermore, 24 modelling the whole ecosystem, in a computationally tractable way, requires that

simplifications be made and therefore the accuracy of its individual components will be reduced (Skogen et al. 2021). Further development should prioritise the models' abilities to describe the processes governing top and bottom trophic levels. Alternatively, existing models could be coupled (e.g., physics, biogeochemistry and fish habitat models, Garcia-Garcia et al. (2021, this issue)), therefore taking advantage of detailed mechanisms at different levels of the ecosystem.

7 The work presented here uses the ecosystem models designed and built with the data in their 8 respective publications, with the skill reflecting the suite of models as opposed to each individual model. Although the whole ecosystem models do not appear to be as useful as the 9 10 size-based models in this study, there are many other questions where they may provide useful output, for instance trophic indicators (Bourdaud et al. 2016), that would better 11 demonstrate the utilities of these models. However, in this study we were interested in using 12 multiple sources of information to inform the effects of different fishing strategies on these 13 14 seven indicators, as opposed to an inter-model comparison, which is beyond the scope of this 15 work. Since this study, there has been further developments of StrathE2E (Heath et al. 2021), which are likely to improve the performance of this ensemble model, particularly for birds 16 and mammals. 17

18 Summary and conclusions

Empirical data analyses currently provide the basis for ecosystem assessment as they represent the key means to monitor change within ecosystems. Data analyses can also demonstrate ecological responses to changes in pressure and the effect of different management measures. However, quantitative modelling tools are the most appropriate basis to explore potential ecological responses to newly planned management measures.

Unfortunately integrating different model outputs is difficult because a) differences in model structure may make it impossible to simply average outputs in a meaningful way, b) the model discrepancies will not be independent of one another, c) uncertainty in the projections can be hard to quantify. The ensemble approach of Spence et al. (2018) directly addresses these issues and provides a powerful framework for integrating a range of models and providing robust advice based on these models and information on their past performance at forecasting quantities of interest, with quantifiable uncertainty (Jardim et al. 2021).

8 We have used this framework to make projections for seven ecosystem indicators of GES, covering key parts of the North Sea food web from phytoplankton to top predators. All 9 10 indicators are projected to increase when fishing the community in accordance with MSY 11 principles, and we have high confidence in this recovery for the size-based indicators (LFI, TyL 12 and MML). However, our confidence in the future response of TFB, BM, Z:P, and ZB is low, with a significant chance that these might not increase with MSY-consistent fishing. 13 14 Performance with respect to TFB is particularly disappointing as this is a key quantity of 15 interest that is forecast by all five ensemble constituents. Future research should be prioritised to reduce uncertainty in biomass estimates by ecosystem models and empirical 16 analyses because these are important indicators of state which are poorly forecast despite 17 18 good understanding of size-structure and species composition.

We demonstrated that this new ensemble methodology can be directly applied to address policy-relevant questions and add value for decision-makers because all uncertainties are considered together. From this work we conclude that current fisheries management objectives to achieve MSY can be considered appropriate for aspects of fish communities

(species composition and size structure) as monitored by The Convention for the Protection
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1 Tables

Table 1. Species list and species-specific fishing mortality rates for MSY and Nash scenarios
taken from Thorpe et al. (2017), with some species having a decreased fishing mortality in
MSY and Nash, relative to status quo and others having an increased mortality. Species are
ordered by asymptotic size.

Species (Latin name)	F _{MSY}	F _{Nash}	F _{Status quo}
Sprat (Sprattus sprattus)	1.30	0.78	1.06
Norway pout (Trisopterus esmarkii)	0.35	0.63	0.48
Sandeel (Ammodytes tobianus)	0.35	0.63	0.57
Poor cod (Trisopterus minutus)	0.72	0.58	0.55
Long rough dab (Hippoglossoides platessoides)	0.60	0.54	0.55
Dab (<i>Limanda limanda</i>)	0.41	0.57	0.55
Herring (Clupea harengus)	0.25	0.50	0.25
Horse mackerel (Trachurus trachurus)	0.50	0.55	0.63
Lemon sole (Microstomus kitt)	0.33	0.48	0.55
Sole (Solea solea)	0.22	0.42	0.53
Mackerel (Scomber scombrus)	0.32	0.45	0.63
Whiting (Merlangius merlangus)	0.21	0.42	0.37
Witch (Glyptocephalus cynoglossus)	0.27	0.38	0.55
Grey gurnard (Eutrigla gurnardus)	0.27	0.46	0.55
Plaice (Pleuronectes platessa)	0.25	0.20	0.55
Starry ray (Amblyraja radiata)	0.15	0.32	0.55
Haddock (Melanogrammus aeglefinus)	0.30	0.42	0.58
Cuckoo ray (<i>Leucoraja naevus</i>)	0.11	0.14	0.55
Monkfish (Lophius piscatorius)	0.10	0.23	0.55
Cod (Gadus morhua)	0.19	0.21	0.88
Saithe (Pollachius virens)	0.30	0.30	0.38

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Figure 1. Indicator estimates from the combination of all five ecosystem models and the
empirical studies until 2013. Starting in 2014, each ecosystem model is run under the MSY
fishing scenario. The uncertainty envelope represents ± 2 standard deviations. The black
vertical line indicates the last year for which empirical studies were considered.





- 2 uncertainty envelope surrounding each prediction represents ± 2 standard deviations. Note,
- 3 the empirical studies end in 2013 as illustrated by the black vertical line.



Figure 3: Violin plots of the ensemble model's predictions for the indicators in 2013 and 2050
under the different management scenarios (SQ = Status Quo, MSY = Maximum Sustainable
Yield, Nash = Nash equilibrium, NF = No fishing).

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Figure 4. Difference between management scenarios (NF = No fishing, SQ = Status Quo, MSY
= Maximum Sustainable Yield) in the long-term predictions for each indicator (see separate
pane titles) from the ensemble model. The pane in the top left-hand corner demonstrates
how to read the plots i.e., areas where one scenario leads to a better solution than another.
The circles encapsulate the uncertainty in the outcomes showing the 25th, 75th and 95th
percentiles. The MSY and Nash scenarios lead to very similar results and so only MSY is shown
here.











