Can the Common Fisheries Policy achieve Good Environmental Status in exploited ecosystems: the west of Scotland demersal fisheries example

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Abstract

The latest reform of the Common Fisheries Policy (CFP) which regulates the exploitation of fish stocks in European waters entails a move from the traditional single stock management towards Ecosystem Based Fisheries Management (EBFM). Meanwhile the Marine Strategy Framework Directive dictates that Good Environmental Status (GES) should be achieved in European waters by 2020. Here we apply an EBFM approach to the west of Scotland demersal fisheries which are currently facing several management issues: depleted stocks of cod (*Gadus morhua*) and whiting (*Merlangius merlangus*), increased predation from grey seals (*Halichoerus grypus*), and large bycatch of juvenile whiting by crustacean fisheries. A food web ecosystem model was employed to simulate the outcomes of applying the traditional single stock fishing mortalities (F), and management scenarios which explored F ranges in accord with the CFP recommendation. Ecosystem indicators were calculated to assess the performance of these scenarios towards achieving GES. Our results highlight the importance of considering prey-predator interactions, in particular the impact of the top predators, cod and saithe (*Pollachius virens*), on juvenile cod and whiting. The traditional single stock approach would likely recover cod, but not whiting. Exploring the F ranges revealed that a drastic reduction of juvenile whiting bycatch is necessary for the whiting stock to recover. Predation from grey seals had little impact overall, but did affect the timing of cod and whiting recovery. With the exception of whiting, little difference was observed between the single stock scenario, and the best scenario identified towards achieving GES. The findings advocate for the use of ecosystem modelling alongside the traditional, single stock assessment model used for tactical decision making in order to inform management.

Keywords: Common Fisheries Policy; Ecosystem Based Fisheries Management; ecosystem modelling; Ecopath with Ecosim; Good Environmental Status

1. Introduction

The exploitation of fish stocks in European waters is regulated by the Common Fisheries Policy (CFP). Since its creation in the 1970s this long-standing policy has been through several reforms, the latest of which took effect on January 1st 2014 (EC, 2013). This latest reform proposes a new framework to manage European fisheries, and amongst several new initiatives, it highlights a need to move from traditional singe-stock management towards an ecosystem approach to fisheries (EAF) (Prellezo and Curtin, 2015). EAF originated from the principle of sustainable development and aims at both human and ecosystem well-being (Garcia et al., 2003). The implementation of EAF can vary between an Ecosystem Approach to Fisheries Management (EAFM) in which ecosystem aspects are given consideration when taking management decisions, to Ecosystem-Based Fisheries Management (EBFM) in which ecosystem health becomes a management goal included in trade-offs when pursuing competing management objectives (Patrick and Link, 2015). Most importantly, EBFM prioritises the wellbeing of ecosystems over economic and social objectives since wellbeing is considered a prerequisite for the last two objectives (Murawski et al., 2008).

While the new CFP advocates for the implementation of EBFM, it remains largely unclear how to include conservation objectives within management measures in practice (Prellezo and Curtin, 2015). The CFP currently aims to fish at levels consistent with achieving Maximum Sustainable Yield (MSY) for all exploited stocks (EC, 2011). In northern European waters, these fishing levels are proposed by the International Council for the Exploration of the Sea (ICES) which delivers annual scientific advice for the management of northern European fish stocks. This advice provides biological reference points for each stock, including the level of fishing mortality (F) needed to achieve MSY (F_{MSY}). F_{MSY} is defined on a single-stock approach, meaning that it is calculated individually for a stock based on its own status only, regardless of the status of other stocks. However, this contradicts EBFM (Prellezo and Curtin, 2015), where the interactions between species should be taken into account when defining safe harvest levels for fish stocks. In fact, while F_{MSY} has long been considered a desirable exploitation level for single stocks (Schaefer, 1954), its performance in mixed fisheries, where several stocks are caught simultaneously by the same fleet, has been challenged (Walters et al., 2005), largely due to the fact that it is virtually impossible to apply F_{MSY} simultaneously to all stocks in mixed fisheries (Kumar et al., 2017; Larkin, 1977). Nevertheless, despite this criticism recent empirical studies have shown that the current MSY approach has succeeded in leading European fish stocks towards recovery (Cardinale et al., 2013; Fernandes and Cook, 2013). This suggests that the traditional single stock F_{MSY} values for European stocks may not be too far off the harvest levels needed to achieve sustainable mixed fisheries, potentially facilitating the transition towards EBFM. For example, Froese et al. (2008) have shown that EBFM can be achieved by improving existing single-stock management.

In addition to the traditional advice and corresponding single stock F_{MSY} values, ICES now also provides F_{MSY} ranges for most stocks in European waters, which consist of upper ($F_{MSY\ upper}$) and lower ($F_{MSY\ lower}$) F boundaries around F_{MSY} within which fishing mortality is deemed sustainable (ICES, 2016a, 2015). These ranges are a recent addition to the ICES advice and were requested by the European Commission in order to develop long-term management plans with quantifiable targets. F_{MSY} ranges should be precautionary and also ensure that they deliver no more than a 5% reduction in long-term yield. Whilst they do not originate from a proper multispecies approach such as the one used by the mixed fisheries advice (ICES, 2017), the F_{MSY} ranges do provide some leeway around the single stock F_{MSY} values which are usually difficult to apply simultaneously to all stocks. In mixed fisheries, the Total Allowable Catch

(TAC) derived from F_{MSY} for the least abundant stock is most likely to be reached before the TACs of more abundant stocks are exhausted. Such a situation typically leads to over-quota discarding, a practice no longer allowed as the landings obligation is phased in for European fisheries (EC, 2015a). As a result, it has been proposed that in mixed fisheries the most vulnerable stock with the lowest F_{MSY} should determine the limit of exploitation for all other stocks caught in the same fishery (EC, 2011). However, such an approach is likely to result in a 'choke species' scenario leading to the under-exploitation of other stocks and ultimately jeopardising the fishery (Baudron and Fernandes, 2015).

Another regulation of European waters is the Marine Strategy Framework Directive adopted in 2008 (EC, 2008) which states that all member states should reach Good Environmental Status (GES) by 2020 (EC, 2009). Although achieving GES differs from achieving EBFM, GES measures the performance towards most of the biological and environmental attributes of EBFM (Ramírez-Monsalve et al., 2016). GES is defined by 11 descriptors. Descriptors 1 (biodiversity), 3 (commercial species), and 4 (food webs) directly relate to fisheries and are therefore particularly relevant for EBFM. In order to integrate these GES descriptors into an EBFM framework, indicators are needed to inform whether GES criteria are met for each descriptor. Developing meaningful ecosystem indicators can be challenging due to a lack of relevant data. However, ecosystem indicators for descriptors 1, 3 and 4 can be derived from biomass and/or catch data which are available for most species in ecosystems found in EU waters (Coll et al., 2016; Gascuel et al., 2016; Kleisner et al., 2015; Reed et al., 2017). In addition, the information a single ecosystem indicator can provide is limited: it is therefore preferable to use a portfolio of indicators to fully assess each descriptor (Samhouri et al., 2009). Lastly, GES indicators also need to be informative. Ideally, what constitutes GES should be defined for each indicator in order to assess whether an ecosystem has reached GES or not based on indicator values. For example, Link (2005) proposed reference points for some ecosystem indicators, in which case the examination of indicators' trends relative to the reference point values can then be used as a basis for management recommendations (Jennings and Rice, 2011). However, not all ecosystem indicators have clearly defined reference points, and these reference points are not transferable across ecosystems with different characteristics (Heymans et al., 2014).

EBFM can benefit from ecosystem modelling in order to explore policy options where management objectives (e.g. diversity, abundance of non-target species, etc.) involve multiple species and their trophic interactions which cannot be assessed with single-species models (Christensen and Walters, 2005). Plagányi (2007) reviewed available ecosystem models spanning a wide range of complexity levels from minimum realistic models to whole ecosystem ones. This latter category includes Ecopath with Ecosim (EwE), a food web ecosystem model (Christensen and Walters, 2004). EwE is the most applied tool for modelling marine ecosystems (Colléter et al., 2015) and can be used to investigate marine policy issues such as GES (Piroddi et al., 2015). However, it is crucial to demonstrate that a model can replicate historical trends in ecosystems in order to make plausible predictions in response to novel situations before any management decision can be based upon it (Christensen and Walters, 2005). Of the vast number of EwE models that have been published, only a few have been calibrated using historical time series of data and even fewer have been employed for management purposes (Heymans et al., 2016). One EwE model fulfilling these two criteria was recently published for the west of Scotland ecosystem (Alexander et al., 2015; Serpetti et al., 2017).

The west of Scotland ecosystem (WoS) located in ICES Division VIa is home to numerous valuable species of finfish and shellfish that support four fisheries: an inshore crustacean fishery targeting the valuable Norway lobster (*Nephrops norvegicus*); a mixed demersal fishery targeting cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*) and whiting (*Merlangius merlangus*) on the continental shelf; a fishery for monkfish (*Lophius piscatorius*), hake (*Merluccius merluccius*) and saithe (*Pollachius virens*) in the deeper waters of the shelf edge; and a pelagic fishery targeting mainly mackerel (*Scomber scombrus*) and herring (*Clupea harengus*) (ICES, 2016b, 2016c, 2016d, 2016e, 2016f, 2016g). In 2014, these fisheries contributed to 35% of the total value of all commercial species caught in Scotland, totalling £182.5 million (The Scottish Government, 2015) and are, therefore, important for the Scottish fishing industry. However the WoS fisheries are currently facing several management issues. First, the stocks of cod and whiting are depleted and their Total Allowable Catches (TACs) have been set to zero since 2012 and 2006 respectively (ICES, 2016c). Secondly, the extensive bycatch of juvenile gadoids by the crustacean fishery is thought to jeopardise gadoid stocks, whiting in particular (ICES, 2016c). Thirdly, the population of grey seals (*Halichoerus grypus*), a top predator in the WoS, has been increasing steadily over the last two decades (SCOS, 2015). While Alexander et al. (2015) suggest that excessive exploitation rates rather than an increase in predators were to blame for the collapse of cod and whiting, increased predation from seals seems to have offset the reduction of fishing pressure on VIa cod (Cook et al., 2015) and is likely to hamper the recovery from low stock sizes (Cook and Trijoulet, 2016). The complexity of the WoS food web and the mixed fisheries it supports, coupled with management challenges and the availability of an ecosystem model, makes the WoS an ideal case study to assess the performance of EBFM in achieving specific management goals such as GES.

Here, we reviewed and updated the EwE model for WoS with the latest data available and repeated the calibration procedure to extend the hindcasting period from 1985 to 2013. We used this model to explore the F_{MSY} ranges of the demersal stocks by performing forward simulations of every possible combination of fishing mortalities within these ranges. Additional exploitation scenarios were performed to investigate the impact of juvenile whiting bycatch by the crustacean fishery and grey seals predation. For each scenario, ecosystem indicators related to GES descriptors 1, 3 and 4 were calculated. Outputs from the models were analysed to assess whether the single stock F_{MSY} and/or F_{MSY} ranges implemented by the CFP could achieve GES in WoS the demersal fishery. Management measures required to recover the cod and whiting stocks were also identified.

2. Material and methods

2.1. The model

The model was built using EwE software version 6.5 released in July 2016 (www.ecopath.org). EwE consists of two components: (i) Ecopath, a mass-balance model accounting for energy transfers in the ecosystem which depicts a 'snapshot' of the ecosystem in a given year; and (ii) Ecosim, the dynamic component which allows for temporal simulations based on Ecopath. Ecosim is based on the foraging arena theory (Ahrens et al., 2012), and each prey-predator interaction is defined by a vulnerability parameter that describes whether the interaction is bottom-up (vulnerability < 2), top-down (vulnerability > 2), or neither bottom-up nor top-down (vulnerability = 2) controlled. Both Ecopath (Christensen and Pauly, 1992; Polovina, 1984; Walters et al., 1997) and Ecosim (Christensen and Walters, 2004; Walters and Christensen,

2007) have been documented extensively, and further details can be found in the publications above.

The EwE model for WoS used in this study was first built by Haggan and Pitcher (2005), then updated by Bailey et al. (2011) and Alexander et al. (2015). It was recently updated and extended by Serpetti et al. (2017) who introduced species-specific thermal preference functions in order to drive the model with ocean temperature. The impact of temperature is beyond the scope of this study (see Serpetti et al. (2017) for more details). Here, we built on the model published by Alexander et al. (2015) by applying the same update as done by Serpetti et al. (2017), minus the inclusion of temperature as a driver. The area modelled corresponds to the continental shelf of ICES Division VIa within the 200 m depth contour and covers ~110,000 km^2 (Fig.1). The model comprises 41 functional groups (Table S1) spanning ~ five trophic levels consisting of three marine mammals, seabirds (as a single group), 23 fish, five invertebrate groups, one cephalopod group, two zooplankton, three benthos, two primary producers, and one detritus group. The model has five fishing fleets: demersal trawl, *Nephrops* trawl, other trawl, potting and diving, and pelagic trawl. The cod, haddock and whiting groups are split between juvenile (age 0 and 1) and adult (age 2 and above). The model start year in Ecopath is 1985 (see Bailey et al. (2011), Alexander et al. (2015) and Serpetti et al. (2017) for more details about Ecopath parameters). Ecopath parameter values employed are given in Tables S1-4.

2.2. Update

The update of Ecopath consisted of two steps. Firstly, the 1985 biomass starting values of groups for which data were available were updated using the latest stock assessments (Table

S1) while the total catch of each functional group was updated with the latest landings (Table S2) and discards (Table S3) data (where available). In addition, the growth parameter (i.e. *K* from the von Bertalanffy growth function) used to model the growth of the three multi-stanza groups (cod, haddock and whiting) was updated by fitting a von Bertalanffy growth function to age-length keys obtained from the ICES DATRAS database (https://datras.ices.dk/Data_products/Download/Download_Data_public.aspx) for those three groups. Secondly, the diet matrix used by Ecopath was updated. Adjusting the diet matrix is a powerful and surprisingly underused way to improve EwE models (Ainsworth and Walters, 2015). To improve the model goodness of fit, the diet matrix was updated following these consecutive steps: (i) the data and proxies used by Bailey et al. (2011) and Alexander et al. (2015) to build the diet matrix were reviewed; (ii) the diet composition of each group was checked individually against existing literature for unusual prey; (iii) when unusual prey/predator links were found these were removed and/or amended based on (in the following order): available literature; the DAPSTOM database (Pinnegar, 2014); the diet matrices of the EwE models from two neighbouring and closely related ecosystems, North Sea (Mackinson and Daskalov, 2007) and Irish Sea (Lees and Mackinson, 2007). The updated diet matrix obtained through these three consecutive revisions is given in Table S4. To ensure a coherent and ecologically sound mass-balance, the pre-balance (PREBAL) analysis depicted by Link (2010) was applied to the updated Ecopath model.

To update Ecosim, the time series of biomass, catch, and fishing mortalities driving the model were updated (from 1985 onwards) and extended (up to 2013) for as many groups as possible using the latest data available. While catch time series were handled on an absolute scale in the calibration process, biomass time series are handled on relative scale: having the correct biomass trend is, therefore, more important than having the correct range of values. To this end it was deemed preferable to estimate the biomass time series directly from scientific survey data rather than from assessment model estimates, whenever possible. For demersal and benthic groups, biomass time series were calculated using bottom trawl surveys data obtained from the ICES DATRAS database following the method from Baudron and Fernandes (2015) with the exception of cod, haddock and whiting for which stock assessment estimates (ICES, 2014a) were necessary to obtain separate biomass time series for both stanzas. For Norway lobster, abundance estimates from underwater TV surveys (ICES, 2014a) were summed across the three functional units within the model area (FU 11, 12 and 13) and used as biomass time series. Since pelagic species are not effectively captured by bottom trawl surveys, whenever possible other data sources were preferred to get reliable biomass trends. For herring, total stock biomass estimates from acoustic surveys available for the subarea VIa north which comprises the bulk of the VIa stock (ICES, 2014b) were used. For mackerel, horse mackerel *Trachurus trachurus* and blue whiting *Micromesistius poutassou*, total stock biomass estimates for the western shelf (ICES, 2014c) were scaled down to VIa using the average proportion of landings realised in this area. For grey seals, estimates of pup production from Inner and Outer Hebrides (SCOS, 2015) were summed and used as biomass trend. For harbour seals, pup count values were only available every five years (SCOS, 2015) but were preferred to model estimates as the biomass trend indicator. Abundances values of small (< 2 mm) and large (> 2 mm) zooplankton, and phytoplankton Colour Index (PCI) were obtained from the Sir Alister Hardy Foundation for Ocean Science (SAHFOS). The PCI constitutes a semi-quantitative representation of the total phytoplankton biomass (Batten and Walne, 2011).

Catch time series for both stanzas of cod, haddock and whiting were obtained from stock assessment reports as these include discards and are corrected for misreporting. Contrary to cod and whiting assessed in VIa, haddock is now assessed for both areas IV and VIa (ICES, 2014d). As a result, it was assumed that 9.5 % of northern shelf haddock catches are realised in VIa as this is the threshold managers agreed upon when splitting the TAC between areas IV and VIa (EC, 2015b). For all other groups, 1985-2013 time series of VIa landings were obtained from STATLANT (STATLANT, http://ices.dk/marine-data/dataset-collections/Pages/Fish-catch-and-stock-assessment.aspx) and 2003-2013 discard rates were obtained from STECF (https://stecf.jrc.ec.europa.eu/reports) to estimate the 2003-2013 catch time series. The catch time series for 1985-2002 were estimated by inversely applying 2003-2013 average discard rates to 1985-2002 landings time series. In EwE, F corresponds to the exploitation rate which is the catch to biomass ratio (C/B). To get F time series, biomass time series were adjusted so that the 1985 starting values correspond to the 1985 biomass estimates from Ecopath before calculating C/B to ensure sensible F values: since biomass values resulting from standardised survey sampling are often much smaller than those estimated from stock assessments, the initial value derived from Ecopath was used. Lastly, the "feeding time adjustment rate" was set to 0.5 for mammal groups as suggested by Christensen *et al.* (2008) and to 0.2 for juvenile stanzas which still feed on egg content in early life stages while the default value of 0 was used for all other groups. The time series of biomass, catch, F, and forced catches (i.e. catches used to drive the model for groups for which F could not be calculated due to lack of either C or B) inputs used to fit Ecosim are given in Tables S5-8.

2.3. Parameterisation

For the model to be reliable enough for EBFM it is essential that Ecosim captures the food web processes. This is shown by the ability to reproduce historical trends in biomass and catches when historical fishing mortalities are applied. Ecosim includes a 'fit to time series' module which identifies the prey-predator interactions most sensitive to changes in vulnerability

(Tomczak et al., 2012). The calibration then consists of adjusting these vulnerabilities until the best 'fit' of the model outputs to historical time series is achieved. Goodness-of-fit is assessed by the sum of squared differences between the predicted and observed values on a $\log_{10}$ scale (Christensen et al., 2008). The fitting procedure described in Alexander *et al.* (2015) was applied and the following model scenarios were tested (see Mackinson et al. (2009) for more details):

(i) Baseline: no fishing or environmental forcing and vulnerabilities set at 2

(ii) Baseline + trophic effects: same as (i) except vulnerabilities are adjusted to fit the data

(iii) Baseline + environmental forcing: same as (i) except the 'fit to time series' identifies a time series of values (forcing function) that improves the fit by impacting the predicted biomasses through primary production (subsequent analyses can be performed to link the forcing function to existing environmental drivers). This forcing function is a spline curve, and the maximum number of spline points tested was limited to five so as to not over-parameterise the model (Tomczak et al., 2012), as done by Alexander et al. (2015).

(iv) Baseline + trophic effects + environmental forcing: combination of (ii) and (iii)

(v) Fishing: fishing mortalities are included to drive the model, no environmental forcing and vulnerabilities set at 2

(vi) Fishing + trophic effects: fishing mortalities are included to drive the model and vulnerabilities are adjusted to fit the data

(vii) Fishing + environmental forcing: combination of (iii) and (v)

(viii) Fishing + trophic effects + environmental forcing: combination of (vi) and (vii)

The best candidate was selected with Akaike's Information Criterion (AIC) which identifies the best trade-off between goodness-of-fit and number of parameters (Mackinson et al., 2009). Instead of manually selecting the number of vulnerabilities to adjust prior to running the 'fit to time series' module (Alexander et al., 2015; Tomczak et al., 2012), an automated stepwise fitting procedure (Scott et al., 2016) was used. This 'stepwise fitting' module has been included in the latest release of the EwE software (version 6.5) and allows for testing every possible combination of parameters by automatically running the 'fit to time series' with successive increments of the number of vulnerabilities and/or spline points of the forcing function for each candidate model (ii) to (viii). The stepwise fitting procedure tested 1,990 model interactions based on 28 time-series of relative biomasses, 22 time-series of catches, 22 time-series of F and 9 time-series of forced catches with a total of 1,355 observations (observed data points) estimating a maximum number of 49 parameters (based only on independent time-series). The fitting procedure searched for vulnerability parameters "by predator" for all iterations assuming the same top-down or bottom up control of the predator on all its prey (Scott et al., 2016).

2.4. Management scenario simulations

Once parameterised, the best candidate model was used to explore the possible management scenarios for the WoS demersal fishery which adhere to the current CFP recommendations. The six demersal species considered here for the demersal fishery are cod, haddock, whiting, saithe, hake, monkfish. Saithe and hake are part of larger groups, pollock and large demersals respectively, composed of more than one species (Table S9). According to Bailey et al. (2011), the pollock group is largely dominated by the saithe (97%) and the large demersals group by hake (ca. 60%, although given recent estimates from Baudron and Fernandes (2015), this proportion is likely to be much higher). The groups pollock and large demersals were therefore considered here as being representative of these two single species, and are hereafter referred to as saithe and hake. Forward simulations were performed for a period of 20 years (i.e. 2014-2033) for each scenario. Firstly, a status quo scenario ($F_{status\ quo}$) was performed by keeping F equal to the last historical value (F_{2013}) for all species in the model (Table 1) and used as a reference level. Secondly, a F_{MSY} scenario was performed by applying the single stock F_{MSY} values from ICES (Table 1). Only cod and whiting have stocks with a corresponding F_{MSY} defined for area VIa, in which the model area is located. For other species, the F_{MSY} defined for stock areas which encompass area VIa were used as best available proxies (Table 1). Lastly, the F_{MSY} ranges were explored for demersal species, whilst single stock F_{MSY} values were applied to Norway lobster and pelagic species. Akin to single stock F_{MSY} values, the best available proxies were used when needed (Table 1). The F_{MSY} ranges were explored by simulating, for each species, the $F_{MSY\ upper}$ and $F_{MSY\ lower}$ boundaries and F values in between these two boundaries with a 0.05 increment (Fig. 2a). In order to investigate management strategies likely to recover cod and whiting, the $F_{MSY\ lower}$ boundaries simulated were lowered to F=0.05, this value corresponding to the observed residual F experienced by species not targeted by fisheries (e.g., juvenile cod, see Table S7). Since haddock is also located on the shelf and likely to be caught together with these two species, the cod F_{MSY} range was also applied to haddock (Fig. 2a). The F_{MSY} ranges simulated therefore differed slightly from the ones given by ICES, but did however encompass them (Table 1). To investigate the impact of reducing juvenile whiting bycatch by the crustacean fishery, the F_{MSY} range applied to adult whiting was also applied to juvenile whiting in order to simulate a reduction from $F_{status\ quo}$ of 0.17 (Table S7) down to F=0.05 (Fig. 2a). To investigate the impact of a reduction in predation by grey seals, 5% and 10% culls were simulated by applying Fs of 0.05 and 0.10 to grey seals, in addition of the current no cull (F=0) situation (Fig. 2a). Simulations were carried out for all possible combinations of Fs within the F_{MSY} ranges tested, resulting in 180,000 scenarios being explored in addition to the $F_{status\ quo}$ and F_{MSY} scenarios. These simulations were performed using the Multisim plugin from the EwE software (Steenbeek et al., 2016).

2.5. GES indicators

To assess whether the management scenarios tested achieve GES, and further identify which scenario is most likely to achieve GES, the following ecosystem indicators (hereafter referred to as GES indicators) were calculated using the model outputs for all scenarios.

2.5.1. Biomass

GES implies that all fish stocks are harvested sustainably and therefore within safe biological limits: the spawning stock biomass (SSB, i.e. of adults) should be above biological reference points. The stocks of cod and whiting which are currently depleted are the only two stocks with the biological reference points biomass limit (B_{lim}) and precautionary biomass (B_{pa}) defined for area VIa (cod: B_{lim} =14,000 t, B_{pa} = 22,000 t; whiting: B_{lim} =31,900 t, B_{pa} = 44,600 t) in which the model area is located (ICES, 2016c). The biomass outputs from the model were therefore used as indicators, in conjunction with the biological reference points, to assess whether each scenario led to the cod and whiting stocks remaining depleted (biomass < B_{lim}), being at risk (B_{lim} < biomass < B_{pa}), or recovering (biomass > B_{pa}). This indicator relates to the GES descriptor 3: commercial species.

2.5.2. Shannon's diversity index

Shannon's diversity index (SI) is an indicator of biodiversity commonly used to assess the impact of fishing on food webs (Gascuel et al., 2016). This indicator was calculated following the formula from Shannon (1948):

$$SI = \sum_G \left(P_G . log_2(P_G)\right) \qquad (1)$$

where P_G is the proportion in weight of the functional group G in the biomass. This indicator relates to the GES descriptor 1: biodiversity.

2.5.3. Marine trophic index

The marine trophic index (MTI) is an indicator of the trophic structure of the upper (trophic level 3.25 and above) part of the food web which includes most commercial fish species and therefore is expected to be impacted the most by fishing (Pauly and Watson, 2005). This indicator was calculated as follows:

$$MTI = \sum(TL_G . W_G) \,/ \sum W_G \qquad (2)$$

where TL_G is the trophic level of the functional group G (for groups with a trophic level ≥ 3.25), W_G is the weight of the functional group G in the biomass. This indicator relates to the GES descriptor 4: food webs.

2.5.4. Mean maximum length

The mean maximum length (MML) is an indicator of the species composition of the food web where fishing is expected to lead to a decline in the proportion of large species (Shin et al., 2005). This indicator was calculated as follows:

$$MML = \sum(W_{G}.L_{\infty G})/\sum W_G \qquad (3)$$

where W_G is the weight of the functional group G present and $L_{\infty G}$ is the asymptotic length of the functional group G obtained by averaging L_∞ values obtained from Fishbase (Froese and Pauly, 2017; www.fishbase.org) across species in each functional group (Table S9). This indicator relates to the GES descriptor 4: food webs.

2.5.5. Food web evenness index

The Food Web Evenness index (FWE) is an indicator of biodiversity which, unlike Shannon's diversity index, not only considers the overall diversity of species but also a balanced biomass distribution across trophic levels and evenness of species within each trophic level. This indicator is obtained by inverting either the Canberra or the Bray-Curtis dissimilarity index, *BC*, calculated based on the dissimilarity of the expected and observed biomass of a functional group *G*, as follows:

$$BC = (\textstyle\sum_G |B_{Ge} - B_{Go}|)/\sum_G (B_{Ge} + B_{Go}) \qquad (4)$$

where B_{Ge} and B_{Go} are the expected and observed biomass of the functional group G within its trophic level, respectively. The expected biomass is calculated by defining a reference state of 'food web evenness' in which group biomasses are decreasing with increasing trophic levels, and all groups within a trophic level have equal biomasses (for more details please refer to Appendix A). An advantage of FWE is that it is independent of the total biomass in the system. Therefore FWE only tracks relative changes in species biomasses, i.e. in the compositional diversity of the community. This indicator relates to the GES descriptor 1: biodiversity.

2.6. Identify the best GES scenario

Apart from the biomass indicator for which thresholds (i.e. B_{lim} and B_{pa}) are defined for the depleted stocks of cod and whiting, none of the four GES indicators used to assess descriptors 1 and 4 have clear thresholds defined above which GES is considered reached. Instead, for these four indicators (H, MTI, MML, FWE) it was simply considered that the higher the value the better, and that a scenario achieving high values across these four indicators is more likely to achieve GES than a scenarios achieving lower values (Coll et al., 2016; Kleisner et al., 2015; Reed et al., 2017). Therefore, in order to identify the scenario most likely to achieve GES (hereafter referred to as best GES scenario) the following framework was applied:

(i) To achieve GES, a scenario should recover the depleted stocks of cod and whiting within safe biological limits (i.e. above B_{pa})

(ii) The recovery of depleted stocks should be achieved as early as possible

(iii) Among scenario(s) that satisfy conditions (i) and (ii), the best GES scenario is the one achieving the highest values overall across the four GES indicators H, MTI, MML, and FWE. The best GES scenario was identified through the following three steps:

   a. firstly, the amplitude of the time series of all four GES indicators was standardised by subtracting the mean and dividing by the standard deviation;

b. secondly, for each indicator, the difference between each scenario's value reached in 2033 and the maximum across all scenarios was calculated;
c. thirdly, the best GES scenario is the one with the smallest sum of differences across the four GES indicators.

2.7. Model uncertainty

In order to investigate the impact of parameter uncertainty on the reliability of the model outputs, Monte-Carlo simulations were performed to assess the sensitivity of Ecosim to uncertainty in the following Ecopath inputs: biomass, production to biomass ratio, consumption to biomass ratio, and ecotrophic efficiency (Heymans et al., 2016). The model identified as the best GES scenario was run with the parameter value for each of these inputs randomly selected from within 10% of the original value, as done by Serpetti et al. (2017). 100 runs were performed, and the confidence interval around the time series of biomass outputs were determined by calculating the 5% and 95% quantiles.

3. Results

3.1. Hindcast

Once the updated Ecopath model was successfully balanced, PREBAL (Link, 2010) diagnostics were carried out and confirmed that: the biomass slope on a log scale declines by ca. 5 – 10% with increasing trophic levels; predator/biomass ratios are <1; and vital rates decline with increasing trophic levels (Appendix B). These diagnostics suggest that the Ecopath model is ecologically sound (Link, 2010). The structure of the updated Ecopath food web is depicted in Figure 3, and the final balanced model parameters can be found in Table S1.

The best fitted model with the lowest AIC was achieved when fishing, trophic effects and environmental forcing were applied (Model 8, see Table 2). This model improved the fit by 62% compared to the baseline model. Adding fishing alone improved the fit by 25%, while the combination of fishing and trophic effects reduced the sum of squares by 61%. Adding a forcing function further reduced the sum of squares by 1%, resulting in the lowest AIC. The environmental forcing function on primary producers identified by the fitting procedure is a spline curve with three spline points. Correlations between this forcing function and environmental indices North Atlantic Oscillation (NAO) and Atlantic Multidecadal Oscillation (AMO), as well as the Sea Surface Temperature (SST) were explored with Pearson product moment correlation tests. SST data was obtained from the Hadley Centre HadISST dataset (http://www.metoffice.gov.uk/hadobs/hadisst/), while NAO and AMO data were obtained from NOAA (https://www.esrl.noaa.gov/psd/data/timeseries/). While correlations with SST and NAO were marginally (cor. = 0.107, $p = 0.046$) and not significant (cor. = -0.099, $p = 0.066$) respectively, AMO was the index most correlated with the forcing function with a highly significant correlation (cor. = 0.583, $p < 0.001$, Fig. S1). As a result, a smoothed AMO index obtained by fitting a Loess (local regression) smoother with a span of 0.5 (Fig. S1c) was substituted with the three spline point curve in the model and used as the environmental forcing function on producers.

The best model (model 8, see Table 2) performed fairly well in reproducing the historical biomass trends of most functional groups over the hindcast period (1985-2013), particularly for demersal species such as cod, whiting, saithe and monkfish (Fig. 4). Biomass trends were also fairly well captured for *Nephrops* and pelagic species except in early years (1985-1990) for mackerel and horse mackerel. The historical biomass trends of grey seals was not captured as well, although the model did produce an increasing trend as observed from the historical data. The confidence intervals calculated from the Monte-Carlo simulations were reasonably narrow for a majority of groups, but did reveal large uncertainties around the estimates of cod, haddock and whiting due to the top-down and bottom-up interactions between the adult and juvenile stages of these multi-stanza groups as previously noted by Serpetti et al. (2017). The model also reproduced the observed catch trends for most groups apart from monkfish over the 1990-2000 period (Fig. S2). Catches of hake, mackerel and *Nephrops* were slightly overestimated, while blue whiting catches were slightly underestimated over the 1995-2000 period. The model showed mixed results regarding the ability to reproduce historical trends of GES indicators (Fig. 5). Historical values for the two food web indicators, MML and MTI, were well matched apart from a peak in the mid-2000s largely driven by the large increase in hake biomass (Fig. 4). The two diversity indicators SI and FWE, however, were overestimated by the model, especially SI. Nevertheless, the model outputs did reproduce the shape of the historical trends to some extent, indicating that the GES indicators returned by the model can be used to compare management scenarios to one another.

3.2. Forecast

No forward projections of the AMO index are available. However, this index has been increasing over the model hindcast period (1985-2013), is known to follow a cyclical pattern, and is now approaching a cooling phase (Kotenev et al., 2011). Thus, the mirror values of the smoothed AMO index over 1985-2013 (Fig. S1c) were used as best available proxy and applied as the environmental forcing function of primary producers over the simulation period (2014-2033) when simulating the management scenarios, as done by Serpetti et al. (2017).

The $F_{status\ quo}$ scenario revealed little to no change for most species biomass (Fig. 4) and catch (Fig. S2) levels compared to the last historical year: cod and whiting remained depleted, while other species either remained on par with 2013 levels or quickly reached a plateau, except herring and horse mackerel which kept declining over the simulation period. The F_{MSY} scenario entailed an increase in F for all species expect cod, herring and horse mackerel (Table 1). This led to a recovery of cod SSB above B_{pa} and an increase in horse mackerel biomass but did not stop herring biomass from decreasing despite temporarily curbing the decline. Single stock F_{MSY} values did not recover whiting SSB which remained well below B_{lim}. However, despite experiencing a F three times greater, whiting achieved a higher SSB with F_{MSY} (F=0.18) than with $F_{status\ quo}$ (F=0.06). Similar observations were made for haddock which experienced an increase from $F_{status\ quo}$ = 0.17 to F_{MSY} = 0.19. This is most likely due to a reduction in the predation pressure from the piscivorous top predators saithe, monkfish and hake which all experienced substantial biomass reductions under F_{MSY}. Grey seals also suffered from a reduction in biomass despite experiencing no cull under F_{MSY}, likely due to a reduction in food supply caused by the lower biomass overall across fish species, in particular the important preys saithe and hake (Fig. S3). Catches realised under F_{MSY} were greater than under $F_{status\ quo}$ across all species except *Nephrops*, suggesting that F_{MSY} would lead to higher yield even for species experiencing a reduction in F.

Out of the 180,000 scenarios tested to explore the F_{MSY} ranges, only 260 recovered both the stocks of cod and whiting above B_{pa} by 2033 (Table S10). Out of these 260 scenarios, the earliest date at which recovery above B_{pa} was achieved for both depleted stocks differed among the levels of seal cull considered: 10 scenarios achieved recovery in 2027 with no seal cull, 20 scenarios achieved recovery in 2028 with a 5% seal cull, and 5 scenarios achieved recovery in 2029 with a 10% seal cull. These 35 scenarios are hereafter referred to as recovery scenarios. Culling grey seals had no effect on how quickly the depleted stocks recovered above B_{lim}: cod and whiting reached the threshold in 2021 and 2024 at the earliest, respectively, regardless of the level of culling applied here. However, culling grey seals had an effect on how quickly the depleted stocks recovered above B_{pa}. Cod reached the threshold in 2022 with a 10% cull, a year earlier than with a 5% cull or no cull. In contrast, the recovery of whiting above B_{pa} appeared slower with higher levels of culling, with the threshold reached in 2027 without cull while a 5% and 10% cull led to the threshold being reached in 2028 and 2029 respectively.

The fishing mortalities applied in the 35 recovery scenarios are displayed in grey in Figure 2b and the corresponding biomass trajectories in Figure 4. The recovery of the cod and whiting stocks was achieved with F values within the F_{MSY} ranges from ICES, with the exception of whiting which required a much lower F (Fig. 2b). Although these 35 recovery scenarios did achieve the recovery of both cod and whiting above B_{pa}, for both species the increase in biomass plateaued around 2030 after which it started decreasing again, with the whiting SSB dipping below B_{pa} by 2033 in all recovery scenarios (Fig. 4). Extending the simulation until 2100 as done by Serpetti et al. (2017) revealed that, while the cod SSB remained above B_{pa} after the ecosystem reached equilibrium, the whiting SSB fluctuated around B_{pa} before stabilising between B_{lim} and B_{pa} by 2060 (Fig. S4). This suggests that the scenarios identified as achieving the fastest recovery of cod and whiting above B_{pa} may not maintain whiting within sustainable limits in the long term. The large uncertainty around whiting biomass estimates prevents any firm conclusions, with ca. half of the confidence interval being above B_{pa} (and ca. two thirds above B_{lim}) by 2100. Out of the 35 recovery scenarios, the recovery of both cod and whiting was only achieved when the highest F of the ranges explored was applied to cod (F=0.25) and saithe (F=0.42), and the lowest possible F (0.05) applied to both adult and juvenile whiting. In contrast, recovery was achieved with all possible F values of the range explored for monkfish and grey seals which indicate that these two top predators did not hinder the cod and whiting stocks recovery, although the predation from grey seals had a slight impact on the date when B_{pa} was reached for these two stocks, as detailed above.

The 35 recovery scenarios all resulted in similar values of GES indicators across the simulation period, with the exception of the FWE index which showed more variability across scenarios (Fig. 5). As a result, the scenario identified as the best GES scenario was also the one returning the highest FWE values. Both the best GES scenario and the F_{MSY} scenario produced similar trajectories for all GES indicators over the simulation period, except for the FWE index between 2014 and 2025. However, for all GES indicators the best GES scenario either slightly outperformed the F_{MSY} scenario (e.g. SI), or caught up with it by 2033 (e.g. MML). Both the best GES and F_{MSY} scenarios resulted in lower values than the $F_{status\ quo}$ scenario for the two food web indicators, MML and MTI, although for MTI all three scenario ended up with similar values in 2033. This is likely due to the high biomasses of saithe and hake observed under the $F_{status\ quo}$ scenario, with the abundance of these two large top predator species resulting in high MML and MTI values despite the low biomasses of other large top predators such as cod and whiting. In contrast, the best GES and F_{MSY} scenarios both resulted in higher values than the $F_{status\ quo}$ scenario for the two biodiversity indicators SI and FWE, indicating that these two scenarios led to a more diverse and even species composition of the WoS ecosystem.

The best GES scenario identified via the GES indicators was achieved when the highest F of the ranges explored for haddock (F=0.25) and monkfish (F=0.41) were applied, while an F slightly above the middle of the range explored (F=0.35) was applied to hake (Fig. 2c). While the non-culled biomass of grey seals did not prevent the recovery of cod and whiting, despite slightly impacting the date when this recovery was achieved as explained above, the best GES scenario was achieved when a 5% cull was applied to grey seals. This indicates that, while the predation from grey seals does not prevent stock recovery, it does have an impact, however small, on the food web structure and biodiversity of the WoS ecosystem. Apart from grey seals which experience a 5% cull under the best GES scenario, the best GES and F_{MSY} scenarios produced similar biomass trajectories which were actually closely aligned for most species with one major exception, whiting, which did not recover under the F_{MSY} scenario (Fig. 4). Likewise, apart from cod and haddock which experienced higher F values under the best GES scenario, the catch trajectories produced by the best GES and F_{MSY} scenarios were also similar, even for whiting which experienced a much lower F (0.05) under the best GES scenario the F_{MSY} (0.18) scenario (Fig. S2).

4. Discussion

The results from the model simulations suggest that the single stock F_{MSY} values currently advised by ICES, if applied to all stocks in WoS, would likely recover cod whilst achieving catches on par with historical levels for most species. This management scenario would also lead to an increase in whiting SSB, but would fail to recover this stock to within safe biological limits, suggesting that the current F_{MSY} value for whiting in ICES area VIa is incompatible with this stock's recovery. In contrast, the results from the simulations exploring the F ranges used in this study suggest that it would be possible to recover both cod and whiting stocks by applying F within these ranges. However, two crucial conditions were necessary for the recovery of both these depleted stocks to happen. Firstly, the recovery of whiting required that the lowest possible F (F = 0.05) of the ranges explored was applied to both juvenile and adult whiting. Due to the depleted status of the VIa whiting stock, adult whiting is no longer actively targeted in WoS and is currently experiencing an $F_{status\ quo}$ of ca. 0.06 due to bycatch. Juvenile whiting, on the other hand, is caught as bycatch by the small meshed crustacean fishery targeting the highly valuable *Nephrops* (the crustacean fishery account for 77% of the discards of age 0 and age 1 (i.e., juvenile) groups), and is currently experiencing an $F_{status\ quo}$ of ca. 0.17 as a result (ICES, 2016c). Our results strongly suggest that a substantial reduction in the bycatch of juvenile whiting by the crustacean fishery is essential to the recovery of the VIa whiting stock. This contradicts the previous findings from Alexander et al. (2015) who concluded that there is insufficient bycatch from the crustacean fishery to prevent the recovery of whiting. While measures to prevent bycatch of juvenile whiting by the crustacean fishery could potentially jeopardise one of the most profitable fisheries in WoS, they will soon become a CFP requirement as the landings obligation is being phased in for demersal stocks (EC, 2015a), with whiting already identified to become a choke species for the crustacean fishery in WoS (ICES, 2016c).

The second requirement for the recovery of cod and whiting we identified is that the simultaneous recovery of cod and whiting was achieved only when the highest possible F from the ranges explored were applied to cod (F = 0.25) and saithe (F = 0.42). Both cod and saithe are piscivorous top predators (trophic level ca. 4) of the WoS ecosystem. Saithe, along with mackerel, is one of the main predators of both juvenile cod (Fig. 6a) and juvenile whiting (Fig. 6b), and the increasing saithe biomass over the historical period has led to an increase in predation pressure on these two juvenile stanzas. Scenarios with the highest Fs on saithe therefore resulted in a decrease in predation mortality on juvenile cod and whiting, thus enabling these two species to recover. Likewise, cod is the main predator of whiting (Fig. 6c) and the third most prevalent predator of juvenile cod after saithe and mackerel (Fig. 6a). Applying the highest possible F on cod therefore limited the increase in predation mortality on whiting, thus enabling the recovery of whiting, whilst also limiting cannibalism on juvenile cod and facilitating the recovery of cod. These results suggest that reducing the biomass of saithe, the main predator of juvenile cod and whiting, together with limiting the increase of cod, the main predator of whiting, are necessary to recover both VIa cod and whiting stocks. The fact that the recovery of cod and whiting, two piscivorous top predators, seems unattainable without curbing the increase of another piscivorous top predator, saithe, indicates that it may not be possible to simultaneously maximise the biomass of all demersal piscivorous top predators of the WoS ecosystem (which also include hake and monkfish). Therefore, it may be necessary to identify the optimum balance between these species to achieve sustainable stocks statuses and a healthy food web.

The concept of 'balanced fishing' was first introduced by Garcia et al. (2012) and has gained momentum in recent years as EBFM garnered more attention, although it remains a hotly debated topic (ICES, 2014e). The intricacies and consequences of prey-predator interactions in exploited ecosystems, and the importance of considering them in the management of mixed fisheries are particularly relevant at a time when improved stewardship in the management of European fisheries is leading to the recovery of most commercial stocks (Fernandes and Cook, 2013) resulting in the increase in the biomass of many top predator as they approach their MSY status, with knock-on implications for prey-predator interactions (ICES, 2016h, 2014e). For example, the recovery of the northern hake stock has led to a large increase in the biomass of this top predator across most of northern Europe, including WoS (Baudron and Fernandes, 2015), with repercussions on prey-predator interactions such as the increased competition with saithe for access to their common prey, as documented in the North Sea (Cormon et al., 2016). Although a similar increase has yet to be reported for saithe, the biomass trend from survey data presented here suggest that this species has been increasing continuously from 1985 to 2013 in WoS, whilst fish stock recoveries have been linked to a decline in fishing exploitation and associated harvest rates in ICES area VI overall, and the neighbouring ICES area V for saithe specifically (Jayasinghe et al., 2015). The possible application of 'balanced fishing' in European fisheries and its consequences for ecosystems are currently being investigated by the ICES Working Group on the Ecosystem Effects of Fishing Activities who concluded that, as fish stock recoveries are expected to have significant trophic effects, ecosystem models such as the one employed here could be used to predict the ecological consequences of stock rebuilding (ICES, 2016h).

Implementing a cull of grey seals, the main predator of cod and one of the main predators of gadoid fish species in WoS, had little impact overall on the recovery of cod and whiting. Both species were able to recover when no cull was applied, an observation consistent with the previous findings from Alexander et al. (2015) who concluded that the rise in grey seals biomass had not led to the collapse of these species. This observation contradicts, however, the findings from a recent modelling study which suggests that the sustained high mortality due to increased predation from grey seals is preventing the recovery of the VIa cod stock (Cook et al., 2015). Reducing the grey seals population by 5% every year had no impact of the recovery of cod, however a 10% reduction led to cod recovering within safe biological limits a year earlier. While the difference is small, this observation is consistent with another recent modelling study showing that the VIa cod stock recovery under current levels of grey seals predation is possible although it would remain precarious (Cook and Trijoulet, 2016). Our results showed that a yearly 10% decrease in grey seals biomass led to a slightly earlier cod recovery, suggesting that an increase in grey seals biomass would potentially delay the recovery, a finding consistent with Serpetti et al. (2017) who identified grey seals as exerting a top-down control on their prey. We also showed that a decrease in grey seals biomass could be detrimental for the whiting recovery: the increase in cod biomass associated with a decrease in grey seals biomass would increase predation mortality on whiting, thus delaying its recovery. This potential impact has not yet been reported for whiting in WoS and highlights the need for considering prey-predator interactions in the management of exploited ecosystems, as previously mentioned. Lastly, the best GES scenario identified here included a 5% cull of grey seals, further demonstrating the impact of the abundance of top predators on the food web structure and diversity. However, the small differences observed between scenarios with and without grey seals cull, coupled with the fact that the absence of cull did not prevent the recovery of cod and whiting, do not provide enough support for culling grey seals as a management measure.

The performance of the exploitation scenarios simulated here towards achieving GES was assessed based on five indicators which only related to three out of the eleven GES descriptors: biodiversity (two indicators), commercial species (one indicator) and food webs (two indicators). GES was therefore not comprehensively assessed in this study as many descriptors were omitted from the analyses since it was not possible to model them due to lack of data (e.g., descriptor 10: Marine litter) or lack of processes included in the model (e.g., descriptor 5: Eutrophication). In addition, apart from the biomass indicator for which reference points are defined for the two depleted stocks, the biodiversity and food web indicators employed here have no clearly established thresholds to enable assessing whether GES is reached (i.e., indicator > threshold). This is further complicated by the fact that there is currently no stringent framework that uses indicators in assessing GES criteria (Queirós et al., 2016). Lastly, one of the two food web indicators employed, MTI, was calculated using fixed trophic levels per species, a practice not as efficient as the use of variable trophic levels which better detects the impact of fishing pressure (Reed et al., 2017). These drawbacks were mitigated through the use of two indicators (i.e., diversity and food web) and the use of an ad-hoc approach to identify the best scenario. Notwithstanding these caveats, the use of a food web ecosystem model combined with biomass thresholds enabled the identification of the management measures necessary to recover the depleted stocks of cod and whiting, thus addressing the most pressing environmental issue in WoS fisheries. Whether or not these management measures would also lead to GES for the WoS ecosystem is ambiguous. This is due to the caveats listed above, but also to the fact that, although the two biodiversity indicators increased under the best management scenario identified here compared to status quo, the two food web indicators decreased. This suggests that it might not be possible to simultaneously maximise both the biodiversity and the food web trophic structure (as measured by MML and MTI). With both biodiversity and trophic structure potentially impacting the WoS ecosystem resilience to fishing and other pressures, GES may only be achieved through appropriate trade-offs between these two descriptors. Nonetheless, the approach employed here (i.e., using biodiversity and food web indicators derived from food web ecosystem model simulations) has been successfully used in previous studies investigating the performance of fishing management scenarios towards the contrasting objectives of MSY and GES (Lynam and Mackinson, 2015; Stäbler et al., 2016). Here, the chosen indicators replicated historical trends, suggesting that perhaps they could be used to explore future trends and compare candidate scenarios to one another in order to inform management decisions. Such an approach is employed, for example, when using surveillance indicators for which there is insufficient information to establish a clear target (Shephard et al., 2015). Future work using greater model complexity could achieve comprehensive assessments of GES. For instance, Alexander et al. (2016) have developed a

EwE model for WoS built on their previous work (Alexander et al., 2015) which includes a spatial component. Such a model could allow, for example, mapping trawl fishing activities in WoS and investigating descriptor 6 (Sea-floor integrity), thus improving on the GES assessment presented here.

The Ecopath model presented here entailed an update of the mass balance model from Alexander et al. (2015), as well as extensive changes to the diet matrix. This updated model was recently employed by Serpetti et al. (2017) to assess the long-term impacts of rising sea temperatures on WoS fisheries. In addition, the data time series used to update the Ecosim hindcast period from 1985-2008 to 1985-2013 included biomass trends derived from survey data for saithe and monkfish, where previously proxies derived from stock assessment model estimates were used (Bailey et al., 2011). This improves the credibility of the model since using raw data avoids the uncertainty and possible errors associated with estimates produced by statistical models (Dickey-Collas et al., 2014), especially when these statistical models were designed for different areas than the model area considered here. Another update was the inclusion of biomass time series of zooplankton and phytoplankton used to fit the model. This addition contributes to further improving the credibility of the model by constraining the model calibration at multiple trophic levels, a practice shown to lead to a better and more credible parameterisation especially when both fishing and environmental effects are considered (Mackinson, 2014). Overall, the updated model showed an improvement of the fit, with the hindcast better reproducing the historical biomass trends of most species compared to the hindcast shown in Alexander et al. (2015) whilst being similar to the hindcast shown by Serpetti et al. (2017). Most importantly, the updated model seems to behave more realistically when performing forward simulations. When reducing F, the biomass estimates produced by the updated model showed a gradual increase, as expected in complex ecosystems where trophic interactions may buffer the impact of a decrease in F. In contrast, the results shown in Alexander et al. (2015) showed a sudden increase in the annual biomass of cod and whiting of several thousands of tonnes within a couple of years when a reduction in F was applied. Whilst not disputing the magnitude of the biomass increase observed by Alexander et al. (2015), such an increase within such a short time seems rather unrealistic. The time scale within which the updated model recovers seems more realistic which is a necessary component when testing fishing management strategies and their impact (Lynam and Mackinson, 2015) such as the date when depleted stocks recover, as investigated here.

Ecosystem modelling is a valuable tool for the implementation of EBFM. The inclusion of multiple species spanning several trophic levels and their trophic interactions is necessary to investigate the impact of management strategies on environmental and conservation objectives such as GES (Christensen and Walters, 2005). Yet, as these conservation objectives become a requirement while the latest CFP reform steers European fisheries management away from the traditional approach and towards EBFM, ecosystem modelling tools are still scarcely used in tactical fisheries management which remains very much single stock orientated (Skern-Mauritzen et al., 2015). EwE has benefited from a continuous development spanning over 30 years (Villasante et al., 2016) and has been successfully employed on numerous occasions to investigate marine policy issues (Christensen and Walters, 2004; Colléter et al., 2015), with recent examples including the investigation of the impact of fisheries management strategies on GES (Lynam and Mackinson, 2015; Stäbler et al., 2016), as implemented in this study. However, the use of EwE as a fisheries management tool has been heavily criticised (Plagányi and Butterworth, 2004), since major pitfalls in the application of EwE can produce misleading predictions about the direction of change caused by management strategies simulated, let alone their magnitude (Christensen and Walters, 2004). In addition, it has been shown that EwE models can produce significantly different results from the same analyses depending on how the model has been calibrated (Mackinson, 2014), indicating that such models should be employed with care, particularly when investigating policy issues. The model employed here has been improved four times since its development (Alexander et al., 2015; Bailey et al., 2011; Haggan and Pitcher, 2005; Serpetti et al., 2017). While the model is able to reproduce historical biomass and catch, suggesting that it successfully captures the dynamics of the WoS food web, many assumptions were made during the parameterisation process. Therefore, the model presented here cannot, in its present state, be employed to make tactical management decisions (e.g., setting a Total Allowable Catch) due to the number of uncertainties (e.g., parameter uncertainty) linked to the various processes it describes. Indeed, the sensitivity of the model to parameter uncertainty led to large uncertainties being observed around the biomass estimates of cod and whiting, the two species on which scenario selection was based. In addition, extending the simulation beyond the period of interest until the ecosystem reached equilibrium revealed that the scenarios identified as achieving the fastest recovery of cod and whiting may not maintain whiting within sustainable limits in the long term although no firm conclusions could be drawn owing to the aforementioned large uncertainties around biomass estimates. However, the model could be used to evaluate trade-offs between species, fisheries, and human uses' impacts which is central to the ecosystem approach (Kaplan and Marshall, 2016). We suggest that it is useful in an EBFM context, possibly alongside the use of traditional tactical models (e.g. stock assessment), to explore various 'what if' scenarios, as done here, to inform managers on the likely future trends of biomass and ecosystem indicators.

5. Conclusion

Using a food web ecosystem model to simulate management scenarios accounted for prey-predator interactions whilst investigating biodiversity and food web indicators related to GES descriptors. Our results suggest that the single stock F_{MSY} values currently advised by ICES would recover the VIa cod stock, providing that F_{MSY} is applied to all stocks in VIa, but would fail to recover the VIa whiting stock. The exploration of alternative management scenarios led to the identification of the exploitation levels required to recover both the cod and whiting stocks, and revealed that two conditions are necessary for these recoveries to happen. Firstly, a reduction in the F experienced for juvenile whiting was necessary to recover whiting, indicating that a reduction in the bycatch of juvenile whiting by the crustacean fishery is needed for the VIa whiting stock to recover. Secondly, the simultaneous recovery of cod and whiting was achieved only when the highest possible Fs were applied to both cod, the main predator of whiting, and saithe, the main predator of juvenile cod and whiting, highlighting the need to consider the impact of prey-predator interactions when managing fish stocks. The best GES scenario identified here resulted in biomass trajectories similar to the ones achieved with the single stock F_{MSY} scenario, with the exception of whiting which did not recover under this latter scenario. Likewise, the GES indicators trajectories achieved by the best GES scenario were broadly similar to the ones achieved by the single stock F_{MSY} scenario. Most importantly, the recovery of the cod and whiting stocks were achieved with F values within the F_{MSY} ranges identified by ICES for the six demersal stock considered here, with the exception of whiting. This suggests that the current management measures enforced in European fisheries by the CFP could achieve GES in the WoS ecosystem, provided that existing management issues such as the bycatch of whiting by the crustacean fishery are resolved, and that prey-predator interactions are accounted for, a component which will increasingly be taken into consideration as European fisheries management is evolving towards EBFM.

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8. Tables

Table 1. Fishing mortalities for the main west of Scotland commercial species used in the model simulations with corresponding references. $F_{status\ quo}$ corresponds to the last historical F value observed (i.e. F_{2013}). F_{MSY} corresponds to the single stock F value from ICES supposed to achieve MSY. For demersal species, the $F_{MSY\ lower}$ and $F_{MSY\ upper}$ values from ICES defining the $F_{MSY\ range}$ are also given with their corresponding references (*for monkfish, since no $F_{MSY\ range}$ values are defined for the stock comprising ICES area VIa the $F_{MSY\ range}$ values for ICES areas IIXc and IXa were used instead as best available proxy).

Fishery	Species	$F_{status\ quo}$	F_{MSY}	Reference	$F_{MSY\ lower}$	$F_{MSY\ upper}$	Reference
Demersal	**Cod**	0.60	0.17	ICES, 2016c	0.11	0.25	ICES, 2016a
	Whiting	0.06	0.18	ICES, 2016c	0.15	0.18	ICES, 2016a
	Haddock	0.17	0.19	ICES, 2016d	0.18	0.19	ICES, 2016d
	Saithe	0.07	0.36	ICES, 2016d	0.20	0.42	ICES, 2015
	Hake	0.04	0.28	ICES, 2016g	0.18	0.45	ICES, 2016a
	Monkfish	0.14	0.31	ICES, 2016g	0.18*	0.41*	ICES, 2016a
Pelagic	**Herring**	0.21	0.16	ICES, 2016f			
	Mackerel	0.13	0.22	ICES, 2016e			
	Horse mackerel	0.30	0.09	ICES, 2016e			
	Blue whiting	0.11	0.30	ICES, 2016e			
Crustaceans	**Nephrops**	0.08	0.109	ICES, 2016c			

Table 2. Comparison of the eight candidate models fitted with the stepwise fitting procedure showing the total number parameters estimated (equal to the sum of the number of vulnerabilities and the number of spline points of the forcing function estimated), the model sum of squares (SS), the percentage of reduction of SS compared to the baseline model, and the Akaike Information Criterion (AIC). The best fitted model is highlighted in bold.

Model	**Description**	**Number of vulnerabilities**	**Number of spline points**	**Total number of parameters estimated**	**SS**	**AIC**	**Fitting: % improvement SS**
1	Baseline	0	0	0	1620.04	242.07	-
2	Baseline + trophic effects	0	0	0	1620.04	242.07	0
3	Baseline + environmental forcing	0	5	5	1550.87	192.99	4
4	Baseline + trophic effects + environmental forcing	34	5	39	1177.68	-109.68	27
5	Fishing	0	0	0	1219.31	-142.97	25
6	Fishing + trophic effects	29	0	29	626.61	-985.70	61
7	Fishing + environmental forcing	0	5	5	1113.15	-256.37	31
8	**Fishing + trophic effects + environmental forcing**	**24**	**3**	**27**	**614.30**	**-1016.76**	**62**

9. Figure legends

Figure 1. Shelf area of the west of Scotland (blue) included in the model.

Figure 2. **a**: Fishing mortalities used to perform forward simulations, together with the $F_{MSY\ range}$ from ICES and the $F_{MSY\ range}$ explored with the model. **b**: Fishing mortalities achieving the earliest recovery of cod and whiting above B_{pa} across all levels of seal cull (no cull, 5% cull and 10% cull) together with the $F_{MSY\ range}$ values from ICES. **c**: Fishing mortalities identified for the scenario achieving the best GES indicator values overall together with the $F_{MSY\ range}$ values from ICES.

Figure 3. Food web structure of the model. Nodes represent functional groups within the ecosystem; the size of the node is proportional to the biomass it represents. Biomass flows enter a node from the bottom and exit a node from the top and are scaled to flow proportion. The y-axis indicates the trophic level of the functional groups.

Figure 4. Biomass outputs from the model plotted with the observed biomass data time series used to fit the model (black dots). From 1985 to 2013, the black line shows the outputs from the model hindcast. From 2014 to 2033, outputs from the forward simulation are shown for the status quo scenario (in black), F_{MSY} scenario (in red), scenarios achieving the earliest recovery of cod and whiting above B_{pa} (in grey) across all levels of seal cull (no cull, 5% cull and 10% cull), and the scenario achieving the best GES indicator values overall (in green). Scenarios with the earliest cod and whiting recovery were achieved with only one F for some groups (e.g., whiting), but several possible F values for others (e.g., monkfish, see Fig. 2) resulting in several grey lines over the simulation period. The grey shaded area shows the confidence

interval around the model hindcast from 1985 to 2013, and around the best GES scenario (in green) from 2014 to 2033.

Figure 5. GES indicators calculated from the model outputs plotted with the values calculated from observed data (black dots). From 1985-2013, the black line shows the GES indicators calculated from the model hindcast. From 2014 to 2033, GES indicators calculated from the forward simulations outputs are shown for the status quo scenario (in black), F_{MSY} scenario (in red), scenarios achieving the earliest recovery of cod and whiting above B_{pa} (in grey) across all levels of seal cull (no cull, 5% cull and 10% cull), and the scenario achieving the best GES indicator values overall (in green).

Figure 6. Predation mortality ($year^{-1}$) under the single stock F_{MSY} scenario experienced by juvenile cod (a), juvenile whiting (b) and whiting (c).

Supplementary figure S1. The three spline points forcing function (in grey) from the best model identified by the fitting procedure plotted together with the environmental indices **a**: Sea Surface Temperature (SST), **b**: North Atlantic Oscillation (NAO) and **c**: Atlantic Multidecadal Oscillation (AMO). On each panel, the index smoothed values and the obtained by fitting a Loess (local regression) smoothing curve with a span of 0.5 (thick black line) are shown alongside the raw values (thin black line) for easier visual comparison with the trend of the forcing function.

Supplementary Figure S2. Catch outputs from the model plotted with the observed biomass data time series used to fit the model (black dots). From 1985-2013, the black line shows the outputs from the model hindcast. From 2014 to 2033, outputs from the forward simulation are shown for the status quo scenario (in black), F_{MSY} scenario (in red), scenarios achieving the fastest recovery of cod and whiting above B_{pa} (in grey) across all levels of seal cull (no cull, 5% cull and 10% cull), and the scenario achieving the best GES indicator values overall (in green). Scenarios with the earliest cod and whiting recovery were achieved with only one F for some groups (e.g., whiting), but several possible F values for others (e.g., monkfish) resulting in several grey lines over the simulation period.

Supplementary Figure S3. Comparison of the temporal changes in the diet composition (in % of prey consumed) of grey seals between the status quo scenario (top panel) and the F_{MSY} scenario (bottom panel).

Supplementary Figure S4. Biomass outputs from model simulations extended to 2100 to allow for the ecosystem to reach equilibrium. The observed biomass data time series used to fit the model are shown with black dots. From 1985 to 2013, the black line shows the outputs from the model hindcast. From 2014 to 2100, outputs from the forward simulation are shown for the status quo scenario (in black), F_{MSY} scenario (in red), scenarios achieving the earliest recovery of cod and whiting above B_{pa} (in grey) across all levels of seal cull (no cull, 5% cull and 10% cull), and the scenario achieving the best GES indicator values overall (in green). Scenarios with the earliest cod and whiting recovery were achieved with only one F for some groups (e.g., whiting), but several possible F values for others (e.g., monkfish) resulting in several grey lines over the simulation period. The grey shaded area shows the confidence interval around the model hindcast from 1985 to 2013, and around the best GES scenario (in green) from 2014 to 2100.