Changes of potential catches for North-East Atlantic small pelagic fisheries under climate change scenarios

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Abstract
Small- and intermediate-size pelagic fisheries are highly impacted by environmental variability and climate change. Their wide geographical distribution and high mobility makes them more likely to shift their distribution under climate change. Here, we explore the potential impact of different climate change scenarios on the four main commercial pelagic species in the North-East Atlantic (NEA): Atlantic mackerel (\textit{Scomber scombrus}), European sprat (\textit{Sprattus sprattus}), Atlantic herring (\textit{Clupea harengus}) and blue whiting (\textit{Micromesistius poutassou}). We used a process-based fisheries model (SS-DBEM), where all the target species were exploited at their maximum sustainable yield (MSY), to project future potential catches under a high- and low-future-greenhouse-gas scenario (RCP 2.6 and 8.5, respectively). Two ocean biogeochemical models (GDFL and MEDUSA) were used to force the environmental conditions. Mackerel and sprat are projected to have increases in a potential catch under both scenarios. Herring and blue whiting are projected to have increases in a potential catch under both scenarios. Herring and blue whiting are projected to increase under the RCP2.6, but future projections under RCP8.5 show mixed responses with decreases or no changes forecasted. Overall, the potential catch is projected to increase in the northern area of the NEA but is projected to decrease in the southern area. These projected changes are mainly driven by changes in temperature and primary production. Shifts in the distribution of pelagic resources may destabilize existing international agreements on sharing of straddling resources as exemplified by the dispute in sharing of quota for Atlantic mackerel. Novel climate-ready policy approaches considering full species distribution are needed to complement current stock-based approaches.

Keywords Climate change · Marine fisheries · Modelling · Projections · Uncertainty · Ecosystem approach · Fisheries management · Widely distributed species · Pelagic species

Introduction
The North Atlantic sustains some of the largest populations of commercially exploited fish stocks in the world (Trenkel et al. 2014). Four temperate-boreal pelagic species account for more than 65% of the total catches and 95% of pelagic species catch in the North-East Atlantic (NEA) (Merino et al. 2014; Trenkel et al. 2014): Atlantic herring (\textit{Clupea harengus}), Atlantic mackerel (\textit{Scomber scombrus}), European sprat (\textit{Sprattus sprattus}) and blue whiting (\textit{Micromesistius poutassou}). Atlantic mackerel (\textit{Scomber scombrus}), European sprat (\textit{Sprattus sprattus}), Atlantic herring (\textit{Clupea harengus}) and blue whiting (\textit{Micromesistius poutassou}). These species are widely distributed and subject to large annual migrations (Corten 2002; Ruzzante et al. 2006; Huse et al. 2010). Fisheries catch statistics from International Council for the Exploration of the Sea (ICES) database...
(http://ices.dk/marine-data/dataset-collections/Pages/default.aspx) during the period 2006–2013 show that the eastern ICES areas in the North Atlantic tend to support higher catches of herring (Fig. 1) whereas western ICES areas in the NEA are more dependent on blue whiting. Mackerel catches are more concentrated on the mid-northern ICES areas while sprat catches concentrate in the Baltic Sea (ICES area III). Norway has the highest catches for blue whiting, herring and mackerel but its sprat catches are small. Denmark exploits more sprat and herring while catching a small proportion of the rest of the species. Harvesting countries of blue whiting, herring and mackerel are Norway, Russia, Iceland and Faroe Islands. Historic records show that the Netherlands had a blue whiting fishery in the past. Between 2014 and 2017, the top 5 exploiter countries for mackerel were also the UK, Norway, Iceland, Russia and Faeroe Islands, whereas the main sprat catches were from Denmark, Poland, Russia and Latvia. Despite other boreal/arctic stocks dealt with, such as summer spawning herring off Iceland, capelin stocks (Greenland/Iceland/Jan Mayen and the Barents Sea), sprat and other southern small pelagic stocks (e.g. anchovy, sardine and horse mackerel) dealt with are far less important in terms of biomass catches, but they are important for the local

Fig. 1 The proportion of catches of the four main pelagic species (BWH blue whiting, HER herring, MAC mackerel, SPR sprat) considered per ICES area in relation to total catches for the NEA. It is based on Official Nominal Catches 2006–2013 from ICES database which includes fish, shellfish and algae catch. Country abbreviations are Denmark (DK), Faeroe Islands (FO), Finland (FI), Iceland (IS), Ireland (IE), Netherlands (NL), Norway (NO), Poland (PL), Russian Federation (RU), Scotland (UKM), Sweden (SE) and United Kingdom (UK)
ecology of local economic importance and have a major ecological role to play in the NEA.

Sprats (Sprattus sprattus) follow regular seasonal migrations and occasionally form huge aggregations (Henderson and Henderson 2017). Mackerel performs extensive annual migrations with distinct spawning, feeding, overwintering and nursery areas (Boyd et al. 2018; Brunel et al. 2018). Blue whiting performs seasonal and diel vertical migrations (Gonçalves et al. 2017). Herring also undergo extensive migrations between feeding and spawning grounds (Kotterba et al. 2017). Historical collapses and recoveries of pelagic fish stocks have been attributed to a combination of climate and fishing effects and recruitment success (Planque et al. 2010; Fernandes et al. 2010; Fernandes et al. 2015). Besides, over-exploitation has been found to decrease fish stock resilience to environmental variability and climate change (Anderson et al. 2008; Bates et al. 2014; Hsieh et al. 2006; Ottersen et al. 2006). Small- and intermediate-size pelagic fish species are recognized as key elements in marine food chains (Cury et al. 2011; Kearney et al. 2012) and distribution changes could have significant conservation and management implications (Astthorsson et al. 2012).

The spawning ability and reproductive cycle of adult herring has often been linked to temperature conditions (Jennings and Beverton 1991; Winters and Wheeler 1996; MacKenzie et al. 2007), salinity (Rönkkönen et al. 2004) and food availability (Parmann et al. 1994; Hufnagl and Peck 2011). Herring stocks in the NEA tend to have inshore nursery areas (Geffen et al. 2011). Warmer temperature favoured higher sprat abundance (MacKenzie et al. 2007), whereas low salinity reduces its abundance despite the species’ tolerance to a wide range of salinity (Peck et al. 2012). Adult mackerel also shows affinity to warm water (Beare and Reid 2002; Jansen et al. 2012; Hughes et al. 2014; Bruge et al. 2016). Predation by mackerel is suggested to affect the distribution of blue whiting (e.g. Payne et al. 2012), and changes in distribution and abundance have also been associated with a strong subpolar gyre (Hátún et al. 2009; Payne et al. 2012) and associated oceanographic processes.

Ecosystem and population models based on size-spectrum theory (Blanchard et al. 2012; Jennings et al. 2008), habitat suitability theory (Kaschner et al. 2006; Phillips et al. 2006) or a combination of both (Fernandes et al. 2013a) are regularly used for projecting future scenarios of widely distributed fish species. These models suggest that temperature and primary production are often the main drivers of change in species’ distribution and abundance at global (Jennings et al. 2008; Cheung et al. 2011; Barange et al. 2014; Chust et al. 2014) and regional scales (Fernandes et al. 2017; Fernandes et al. 2016; Speirs et al. 2016). However, despite capturing observed global decadal trends with some success (Fernandes et al. 2013a, b; Jennings and Collingridge 2015), these models are not precise and realistic enough to inform short-term fisheries management (Dickey-Collas et al. 2014). This is partly because these models do not consider all the drivers and local geographical characteristics that impact specific species or stocks (Planque et al. 2011). Despite the complexity of these models, there are still processes that may be underrepresented, such as top-down effects (Kearney et al. 2012), which are often represented as part of a mortality term. Future projections are not only impacted by uncertainties stemming from unresolved and uncertain processes in the models, but also by natural variability and scenario uncertainty (Payne et al. 2016; Cheung et al. 2016a; Mullon et al. 2016). These points highlight the limitations in predicting the future changes in species biomass and distribution, but it also shows that there is an increased capacity to consider long-term scenarios of change and its implications as well as the short-term forecast improvements (Fernandes et al. 2015; Trifonova et al. 2015).

Considering the importance of widely distributed pelagic species, high mobility, their sensitivity to environmental changes and exploitation patterns and their key role as prey to other resources (Trenkel et al. 2014; Cury et al. 2000), this paper aims to assess the impact of climate change on the long-term potential catches of the most abundant pelagic species and the likely implications for fisheries management.

### Methods

A multispecies fisheries model (Fernandes et al. 2013a) which is a combination of a size-spectrum model and a dynamic bioclimate envelope model (thereinafter the SS-DBEM) was used to provide estimates of fish production potential under climate change scenarios. SS-DBEM is driven by the environmental variables obtained from two ocean biogeochemical models that are run under a low- and high-greenhouse-gas-emission scenario (RCP2.6 and RCP8.5) over the twenty-first century considering multiple mechanisms summarized in Table 1 and described in the following sections. Unless otherwise stated, we show 20-year averages of potential catches in the figures and tables to suppress the interannual-to-decadal variability of species, as we are interested in multidecadal to centennial changes. However, figures with time series show yearly values with the full modelled variability. The main data (results of model projections) is in the process of being made public through the European Copernicus service (https://climate.copernicus.eu/).

### Species-based fish model

We used the dynamic bioclimate envelope model (DBEM), a combined mechanistic-statistical approach that has been applied to a large number of marine species globally (Fernandes et al. 2013a; Mullon et al. 2016; Fernandes et al. 2017) and regionally (Jones et al. 2013; Fernandes et al. 2016; Fernandes et al. 2017). This model projects changes in species distribution and
Table 1: Table summarizing the main equations and parameters to consider the species mechanisms in SS-DBEM. Further details in associated references.

| Mechanism                          | Equation                                                                 | Parameters                                                                 |
|------------------------------------|---------------------------------------------------------------------------|---------------------------------------------------------------------------|
| Growth = anabolism – catabolism     | \( G = HW^a - kW \)                                                       | \( H \) = anabolism coefficient \( k \) = catabolism coefficient \( W \) = body weight \( a \) = anabolism exponent (0.5 to 0.95) \( W_e \) = asymptotic weight The coefficients \( g \) and \( b \) were derived from the average \( W_1, K \) and environmental temperature \( T \) of the species reported in the literature. |
| (Pauly 2010; Cheung et al. 2011)    | \( H = g(O_2) \times e^{-0.1T} \)                                       |                                                                          |
|                                    | \( k = h[H^+] \times e^{0.2T} \)                                         |                                                                          |
|                                    |                                                                           |                                                                          |
| Intrinsic population growth rate    | \( G = r \times A \times (1 - (A/KC)) \)                                  | \( W \) = weight \( L \) = length \( E \) = activation energy of metabolism \( k \) = Boltzmann’s constant \( T \) = temperature in Kelvin (°C + 273) \( r \) = intrinsic rate of population increase \( A \) = the relative abundance \( KC \) = population carrying capacity |
| (Hilborn and Walters 1992)          |                                                                           |                                                                          |
| Larval recruitment                  | \( \ln(T) = \frac{\sum \ln(T_i)}{N} \)                                   | \( PLD \) = pelagic larvae duration \( T = \) surface temperature \( T_c = 15 \) C DM is the developmental type of larve (0 lecithotrophic, 1 planktotrophic) \( N \) = number of cells where species occur |
| (O’Connor et al. 2007; Cheung et al. 2008) | \( \ln(PLD) = 0.13(\ln(T/T_2)) - 0.283(\ln(T/T_2))^2 \) \( \beta_0 = 0.739 + 0.739(\ln(T) + 0.714(DM)) \) | \( D \) = diffusion parameter \( u, v \) = velocity parameters \( \text{LAV} \) = larval recruitment |
| Larval dispersal                    | \( \frac{\partial \text{Lav}}{\partial t} = \frac{\partial}{\partial x} \left( \frac{D \partial \text{Lav}}{\partial y} \right) \)\( + \frac{\partial}{\partial y} \left( \frac{D \partial \text{Lav}}{\partial x} \right) \) | \( cm = \) centimetre \( h = \) hour |
| (Hundsdorfer and Verwer 2003; Cheung et al. 2008) | \( \frac{\partial \text{Lav}}{\partial x} = \frac{\partial}{\partial y} \left( \text{Lav} \frac{\partial \text{Lav}}{\partial x} \right) \) \( - \frac{\partial}{\partial y} \left( \text{Lav} \right) \) \( \lambda = \text{Lav} \) | \( \text{Lav} = \) larvae recruitment |
| Adult movement                      | \( Cm = h^{-1} \)                                                         | \( CM = \) centimetre \( h = \) hour |
| Natural mortality                   | \( M = -0.4851 - 0.0824 \times \log(Winf) + 0.6757 \times \log(K) + 0.4687 \times \log(T) \) | \( \text{Winf} = \) asymptotic weight \( K = \) von Bertalanffy growth parameter \( T = \) average water temperature in the animal’s range |
| (Pauly 1980; Cheung et al. 2011)    |                                                                           |                                                                          |
| Fishing mortality at MSY            | \( F_{\text{MSY}} = 0.4 \times M \)                                      | \( M = \) natural mortality |

abundance with explicit consideration of mechanisms of population dynamics, dispersal (larval and adult) and ecophysiology (see Table 1), under changes in ocean temperature, salinity, upwelling, sea-ice extent and habitats (Cheung et al. 2011; Cheung et al. 2016b) considering all the species distribution (not specific stocks of each species separately). Specifically, we employed a multispecies version of the model (SS-DBEM) that incorporated species interactions based on size-spectrum (SS) theory and habitat suitability, the SS-DBEM (Fernandes et al. 2013a). Therefore, the model considers predation and food availability through size-spectrum energy transfer from primary producers to consumers of progressively larger body size. Despite yearly outputs, pelagic species have two internal time steps to account for interannual seasonality and both bottom and surface environmental drivers are considered since these species have pelagic and demersal life stages. For example, herring is a demersal spawner (Lambert and Ware 1984; Axelsen et al. 2000) whereas mackerel, sprat and blue whiting are pelagic spawners (Coombs et al. 1981; Lambert and Ware 1984; Fréon et al. 2005).

Fishing practices are considered in relation to the concept of maximum sustainable yield (MSY; Table 1). MSY is defined as the highest average theoretical equilibrium catch that can be continuously taken from a stock under average environmental conditions (Hilborn and Walters 1992). In our application, the fishing mortality under MSY \( (F_{\text{MSY}}) \) values were obtained from FishBase (www.fishbase.org) assuming that \( F_{\text{MSY}} \) is approximately equal to 0.4 × \( M \) (natural mortality). This equation is used instead of the \( F_{\text{MSY}} \) from stock assessments because SS-DBEM models the whole species distribution (both inside and outside NEA) instead of modelling individual stocks. Therefore, the whole species distribution (both inside and outside NEA) is modelled given that widely distributed species do not follow human management boundaries (Baudron et al. 2020). However, this equation
values are in general consistent with values from stock assessments as reviewed below in the discussion section.

The larvae recruitment depends on temperature and currents that are known mechanisms affecting the mortality or success of herring, mackerel, blue whiting and sprat (Alvarez and Chifflet 2012; Fernandes et al. 2015; Huse 2016; Martin et al. 2016; Henderson and Henderson 2017). The model calculates larval dispersal through ocean currents and diffusion assuming that pelagic larvae disperse passively from surrounding ‘source’ areas through ocean surface current and diffusion (Cheung et al. 2008; Table 1). The magnitude of larval recruitment is dependent on pelagic larval duration (PLD), strength and direction of ocean currents and diffusivity (Table 1). PLD, expressed in days, is calculated from an empirical equation established from a meta-analysis of PLD from 72 species of fish and invertebrates (O’Connor et al. 2007). Based on the calculated PLD and ocean current velocity data, the model calculates dispersal of pelagic larvae over time through diffusion and advection. Diffusion and advection of ocean currents are important factors determining dispersal of pelagic larvae of marine organisms (Possingham and Roughgarden 1990; Gaylord and Gaines 2000; Bradbury and Snelgrove 2001; Gaines et al. 2003). The temporal and spatial patterns of pelagic larval dispersal were modelled by a two-dimensional advection-diffusion equation (Table 1; Sibert et al. 1999; Gaylord and Gaines 2000; Hunsdofer and Verwer 2003).

Ocean biogeochemical models

Ocean environmental conditions from two ocean biogeochemical models of different complexity were used to force the species-based fisheries model SS-DBEM: the NEMO-MEDUSA (NEMO, Nucleus for European Modelling of the Ocean; and MEDUSA, Model of Ecosystem Dynamics, nutrient Utilisation, Sequestration and Acidification, Yool et al. 2013) and the GFDL ESM2M (GFDL, Geophysical Fluid Dynamics Laboratory; and ESM2, Earth System Model z coordinate, Dunne et al. 2012; Dunne et al. 2013). Both ocean biogeochemical models simulate changes in physical and biogeochemical ocean conditions over the historical period and the twenty-first century under two different greenhouse gas scenarios (RCP2.6 and RCP8.5). These ocean conditions include seawater temperature, salinity, oxygen, alkalinity, primary production and horizontal and vertical water currents and affect the ecological processes in the SS-DBEM model such as the life history, habitat, population dynamics and dispersal (Fernandes et al. 2013a; Queirós et al. 2016).

Nucleus for European Modelling of the Ocean- Model of Ecosystem Dynamics, nutrient Utilisation, Sequestration and Acidification biogeochemical model

The NEMO-MEDUSA is a global ocean model (Yool et al. 2013) with a half-degree ocean resolution. It simulates the physical environmental conditions and a size-structured ecosystem of small phytoplankton and zooplankton. The NEMO-MEDUSA explicitly includes the biogeochemical cycles of nitrogen, silicon and iron nutrients as well as the cycles of carbon which are not used directly by the fish model but influence the primary production. The NEMO-MEDUSA model was initialized using standard biogeochemical climatological fields (Garcia et al. 2010; Key et al. 2004) and simulated under surface atmospheric forcing derived from the Met Office Unified Model simulations (HadGEM2 configuration).

Geophysical Fluid Dynamics Laboratory biogeochemical model

The GFDL ESM2M is a global coupled atmosphere-ocean general circulation model (Dunne et al. 2012, 2013) including a marine biogeochemistry model. The global ocean model has an approximately 1° horizontal resolution and it describes the cycles of carbon, nitrogen, phosphorus, silicon, iron, oxygen, alkalinity and lithogenic material and considers three phytoplankton functional groups.

Greenhouse gas emission scenarios

The two ocean biogeochemical models were run under two greenhouse gas emission scenarios (Moss et al. 2010): a strong mitigation scenario (Representative Concentration Pathways RCP2.6) with an atmospheric CO₂ concentration of 421 ppm by the end of the twenty-first century, and ‘the business as usual’ high-greenhouse-gas-emissions scenario (RCP8.5) with atmospheric CO₂ concentrations of 936 ppm by the end of this century.

Validation of the species projections

Standardised and long-term fisheries surveys from 1977 to 2007, collated by Simpson et al. (2011), were used to cover the breadth of fisheries for this study. The survey data available for validation included AFBI Irish Sea Q1 and Q3, Cefas Celtic Sea, Cefas North Sea (autumn) and ICES IBTS North Sea (spring) datasets. The raw data are now freely available at the ICES website (www.ices.dk-marine-data/data-portals/Pages/DATRAS.aspx). Data from demersal surveys are widely used to estimate abundance (e.g. ICES (International Council for the Exploration of the Sea) 2014d; Peck et al. 2013) and distribution (e.g. Huse et al. 2008; Jansen et al. 2012) of pelagic species. Given that blue whiting, sprat and herring have all been shown to undertake diel migrations (Cardinale et al. 2003; Post et al. 2003) and that survey trawl data has been shown to be a reliable indicator of pelagic species abundance (Fig. 4, Suppl. Mat.; Montero-Serra et al. 2015; ICES 2015; Peck et al. 2013) and distribution (Huse 2016).
et al. 2008; Jansen et al. 2012), the dataset used in this study are considered more reliable when compared to landings (herring) and acoustic (sprat) data (Fig. 4, Suppl. Mat.; Montero-Serra et al. 2015). Despite these studies, it is acknowledged that bottom trawl surveys are not designed to target pelagic species, and that additional abundance estimates should be extracted from acoustic surveys (e.g. ICES (International Council for the Exploration of the Sea) 2015). However, acoustic data is not consistently available over the study area and timeframe despite recent important advances in its harmonization and public availability are promising for its use in ecosystem modelling and validation. For example, the ICES Working Group on Acoustic and Egg Surveys for Sardine and Anchovy in ICES areas 7, 8 and 9 (WGACEGG) is developing a not-yet-public protocol with the title ‘Manual for Acoustic Surveys in Ices Areas 6, 7, 8 and 9’.

The total number of individuals for each species was tallied across size classes for each survey haul. An average catchability estimate was applied to similar pelagic species for all size groups using total biomass and catch estimates from Sparholt (1990):

Corrected abundance = uncorrected abundance × (1/catchability)

To control for the differing effort between surveys, the swept area for each haul was calculated using estimates of wing-spread for Grand Ouverture Vertical (GOV) trawls from Fraser et al. (2007) multiplied by the distance over which the hauls were undertaken scaled to square kilometres following the equation:

\[
\text{Area swept km}^2 = ((6.85 \times (\log(\text{depth}))) + 5.89) \times \text{distance} / 1,000,000
\]

Depth and distance were measured in metres, being the distance based on haul duration and speed (assumed tow speed 4 knots for duration of haul, except for North Ireland data where knots for duration of haul, except for North Ireland data where

All the survey data is aggregated into a cell grid of 0.5 × 0.5 degrees to match the fish model grid. To compare projected changes with observations from surveys, a time series per cell for each species have been compared. Besides, time series aggregating all the spatial cells with survey data for each species is compared to estimate the long-term performance of wider areas. The time series are normalized between 0 and 1 to consider that the model projects relative change and to have error estimates in an interval that are easier to understand. Since multiple species at multiple cells were considered, we needed to ensure that results are comparable; therefore, time series of survey data with more than 3 years of missing data were not included. As a result, not enough data to assess the simulated changes in blue whiting was available. Then, time series from the models were extracted for those years, species and cells where there was commonly available data from the surveys at the 0.5 × 0.5 degrees and yearly resolution. This restricted the data that could be analysed to the period from 1982 to 2007 (26 years) and the following 3 pelagic species (out of the 4 modelled here): herring, mackerel and sprat. These time series were produced for both the survey data and the model projections, and were compared by calculating the average error (AE) between them:

\[
\text{res}_{op, \text{SS-DBEM}} = \frac{2 \times \text{std.dev}(\text{Bio}_{\text{DSuit}})}{\text{mean}(\text{Bio}_{\text{DSuit}})},
\]

where \( p \) is the biomass projected in the SS-DBEM model in a particular year for each species, and \( s \) is the biomass from the survey and \( j \) is the number of years with data.

### Results

#### Performance of the fish projections

General trends are well simulated by the models when aggregating over big areas (Table 2 and Fig. 2). However, local scale simulations are more complicated (grid cell level). Errors at cell level range between 0.43 and 0.49, whereas errors aggregating all the cells with survey data drop to the range 0.23 to 0.41 (Table 2). In general, projections for herring and mackerel using GFDL biogeochemical model tend to have lower errors than projections using NEMO-MEDUSA’s output. However, sprat projections using the NEMO-MEDUSA model at both cell and aggregated levels are higher than those projected with the GFDL. Moreover, none of the differences between biogeochemical model projections is statistically significant (paired \( t \) test). Therefore, it cannot be concluded that projections driven by a particular biogeochemical model are better than the other.

The large variance in the error highlights the need to consider the model projections as averages over periods (e.g. 20-year averages) instead of considering the modelled interannual variability. This can be observed in Fig. 2 where variability projected

| Species | Time series | GFDL | MEDUSA |
|---------|-------------|------|--------|
| Herring | By cell     | 0.43 ± 0.15 | 0.44 ± 0.17 |
|         | Cell sum    | 0.27 ± 0.17 | 0.23 ± 0.16 |
| Mackerel| By cell     | 0.46 ± 0.15 | 0.49 ± 0.16 |
|         | Cell sum    | 0.30 ± 0.29 | 0.41 ± 0.19 |
| Sprat   | By cell     | 0.49 ± 0.17 | 0.47 ± 0.16 |
|         | Cell sum    | 0.31 ± 0.29 | 0.26 ± 0.22 |
by the model and that observed in surveys do not match at the year by year scale; however, similar decadal trends are observed. This outcome drives our approach of showing results as an ensemble of both model projections (Fig. 3) to show the inherent uncertainty from the projection of environmental variables in the biogeochemical models into the species model.

**Projections of potential catch and its distribution under two emission scenarios**

Herring catches were projected to vary by ±5–10% relative to the present day (2015) averaged over the North-East Atlantic, but the projections diverge between the two gas emission scenarios from 2070 onwards (Fig. 3(a)). Under the low-greenhouse-gas-emission scenario, herring is projected to increase by up to 15% by the end of this century, while under the high-greenhouse-gas-emission scenario, herring is projected to decrease by up to 35%. On the contrary, mackerel was projected to increase in both future scenarios: up to 5–10% until 2040 (Fig. 3(b)) and much higher increases 15–30% by the end of the century. The model simulates increases in catch trends for mackerel (Fig. 3(b)) and sprat (Fig. 3(c)). However, projections of sprat have larger uncertainties in the first decades than other species projections. Sprat would be benefited from changes in environmental conditions under a low-gas-emission scenario and showing the most significant differences across. Catch of blue whiting is projected to increase by up to 10% in a low-emission scenario but it is projected to decrease by up to 40% under a high-emission scenario (Fig. 3(d)).

![Fig. 2](#) Time series of projections of 3 pelagic species with the two different biogeochemical model forcing in the same cells where there is survey data. The abundance time series are normalized and the values have been smoothed using a 5-year moving average.

![Fig. 3](#) Relative change of potential MSY catches for the four pelagic species in ICES areas II to VII, relative to 2015 catches. RCP2.6 (green) and RCP8.5 (blue) represent a low- and high-future-greenhouse-emission scenario, respectively. Both biogeochemical models (GFDL and NEMO-MEDUSA) were used to drive two fisheries projections for each scenario. Then, the mean value is used to calculate the straight-line trend and the shaded ranges show the difference between both projections.
3(d)). However, there is considerable uncertainty in projections under the high scenario and the results need to be analysed carefully.

Catch projections for all the pelagic species show consistent increases under a low-emission scenario by the end of the century (Fig. 3). The projections under the high-emission scenarios, however, are more uncertain than low-emission scenarios and sometimes do not even agree on the sign of changes. Herring and blue whiting show the lowest uncertainty in the medium term to 2040 (Fig. 3(a, d)), whereas mackerel and sprat show higher uncertainty in the medium term between 2040 and 2080 (Fig. 3(b, c)). The uncertainties in potential catches are mainly driven by the high uncertainty in primary production projections (Table 3). Despite a general increase in temperature in all ICES areas, northern areas experience generally lower increases than southern areas in the shorter term (2020–2039). Primary production increases in areas II and III and decreases everywhere else despite high uncertainty. Area III, which contains the highest increases, shows the lowest uncertainty in terms of primary production estimates.

Figure 4 shows the changes in species distributions for the periods 2020–2039, 2050–2069 and 2080–2099 relative to the present baseline period (2000–2019) under the high-emission scenario. Blue whiting, herring and mackerel have a potential for habitat displacement towards the Arctic Ocean, with reductions in the southern area of distribution, especially for herring and blue whiting. For capelin, there are no agreements between the models since GFDL projects in general slight increases for both emission scenarios with decadal oscillations, whereas MEDUSA shows strong declines in the high-emission scenarios for most of the ICES areas (higher declines in northern areas of NEA). Sprat shows a decline in catch potential in areas IV, VI and VII, but not a corresponding increase in northern regions. There is a consistent pattern showing that northern latitudes will benefit whereas southern areas will be negatively impacted in terms of all species catches. ICES area IIB is the area that shows the highest increases with gains in blue whiting, herring and sprat. Sprat increases are concentrated at the more southerly end of the northern regions IIA and V, areas that show other three species. Herring is expected to show the highest changes by 2020–2039 and 2050–2069, both positive and negative, in terms of the amount of area experiencing changes. Mackerel will be the second species more impacted by 2050–2069. Blue whiting is projected to increase and sprat to decrease, with a few local exceptions. Herring and mackerel show the highest changes with dramatic northern shifting by the end of the century. Overall, the lower emission scenarios show similar spatial patterns in terms of areas of highest increases and decreases for each species with some differences: (i) slower rates of changes with almost no change by 2020–2039, (ii) changes in 2050–2069 similar to the high-emission scenario between 2020 and 2039 and (iii) changes by the end of the century similar to the mid-century projections under the high-emission scenario. The sparse catches have concentrated mostly in the south-western areas. However, the potential catches decrease projected by the model in the future are consistent with the trend observed in the Baltic (ICES area III) catch data. Therefore, the model cannot be trusted to forecast the distribution of species precisely, but it is good for general trends over wide areas (e.g. ICES areas).

Discussion

We show that under a MSY, the SS-DBEM projects a general increase in the potential catch for all widely distributed pelagic species in the short (by 2020) and the medium term (by mid-century), but a decrease for herring and blue whiting by the end of the century under a high-emission scenario (RCP8.5). Potential catch increase is projected in northern areas but decreases in southern areas mainly due to changes in temperature and primary production (Table 3) simulated by two ocean biogeochemical models in response to climate change. For example, both models simulate a decrease in the Atlantic Meridional Overturning circulation under global warming, which leads to an overall reduction of ocean heat transport to the high North Atlantic and as a result to a reduced warming south of Greenland (Winton et al. 2013).

Potential catch increases of pelagic species are projected over the twenty-first century, despite negative trends in total primary production in NEA southern areas. However, total fish biomass can decrease as shown in other studies (Cheung et al. 2009; Lotze et al. 2019) while biomass of small fish species increases (e.g. pelagic species considered in this study). This is because at a higher temperature, the steepness of the relationship between the primary production and fish abundance will increase (Fernandes et al. 2016a). Using a simple size-spectrum approach based on temperature and primary production (Jennings et al. 2008), an increase of 2°C in temperature (and at the same primary production level) can trigger a 20% decrease in total biomass, but an increase of smaller size fish abundance and biomass (Fernandes et al., 2016). This is consistent with higher trophic and benthic species projected to decrease as a result of warming and ocean acidification in southern areas of the NEA (Queirós et al. 2015; Fernandes et al. 2017; Lotze et al. 2019). The two biogeochemical models show agreement on the main trends and areas of impacts. The differences in the simulated physical and biogeochemical conditions between the two models under the same future scenario may be explained by internal natural variability uncertainty or model uncertainty (Walters et al. 2005; Hawkins and Sutton 2009; Frölicher et al. 2016; Cheung et al. 2016a; Frölicher et al. 2016).

Figure 2 shows that we can trust more the model for herring and sprat because they are more able to reproduce the
historical catches (goodness of fit), also considering that the SS-DBEM is not a statistical model where catch data has been used to drive the model (generalization power). However, it seems the model is less reliable for mackerel in comparison with the other species. Figure 2 does not show high uncertainty in the historical projections since there is small difference between the model run trends except for sprat that shows higher differences. Nevertheless, higher uncertainty is shown in the projections (Fig. 3), so that scenario uncertainty is expected to be higher than model internal variability. These study results are in agreement with empirical work which confirms projected increases of mackerel in the Svalbard Archipelago (Berge et al. 2015). Furthermore, evidence of mackerel distribution changes in association with warmer temperatures across the North Atlantic have been recorded (Overholtz et al. 2011; Hughes et al. 2014; Montero-Serra et al. 2015(Montero-Serra et al. 2015). Sprat abundance and size have been declining since 1980 (Henderson and Henderson 2017) due to changes in temperature and global climate patterns in the Bristol Channel which contains spawning and overwinter areas. A distribution shift of fish species can occur due to changes in habitat suitability, but there are additional processes involved (e.g. geographical attachment, species interactions or demographic structure) which are needed for a shift to happen or which can limit it (Planque et al. 2011). Therefore, a species may be able to move to new areas (Nøttestad et al. 2016) where it has not been previously observed or species can increase their abundance dramatically in areas where they have rarely observed due to density changes (Petitgas et al. 2012; Punzón et al. 2016). Both situations would produce a shift in the centroid of the species distribution. The SS-DBEM model used considers many of these processes as detailed in Planque et al. (2011). The modelled shift in distribution is determined by changes in predicted carrying capacity of the species in each grid cell, while its shifts are the result of the spatial population dynamics (Cheung et al. 2016b). Firstly, carrying capacity of the population in a cell is positively related to habitat suitability and the energy from primary production that is available for the species with consideration of competition between species in the same cell which can limit shift speed (Fernandes et al. 2013a, b). Secondly, diffusion of the populations is also related to the gradient of habitat suitability between adjacent cells, resulting in an increase in net diffusion out of the cell if habitat suitability in adjacent cells is higher. Thirdly, population recruitment is determined by the dispersal of larvae which is dependent on surface ocean advection and pelagic larval duration; the latter is a function of temperature. Thus, as ocean conditions change, these three processes result in the change in the distribution of abundance of the species.

The SS-DBEM model used do not account for the adaptation capacity of the species to changes in environmental conditions, due to the sparse knowledge available. This could slow the rate of changes or limit the spatial occurrence of these changes. Engelhard and Heino (2004) demonstrated substantial changes in phenotypic growth and maturation in herring, but a weak evolutionary response. While some studies investigate differences in spawning herring populations using genetic proxies (Jørgensen et al. 2005; King et al. 1987; McPherson et al. 2003), herring spawning tactics also appear to be influenced by phenotypic variability (Jennings and Beverton 1991; Winters and Wheeler 1996). Herring have
probably adopted different spawning seasons as one of their survival strategies (Melvin et al. 2009; Sinclair and Tremblay 1984). Although it is established that in herring the process of first maturation is triggered by a combination of physiological (size condition of fish) and environmental cues such as temperature (Winters and Wheeler 1996) and photoperiodic cycles (McPherson and Kjesbu 2012), the relative influence of genetics and environment on herring spawning behaviour (i.e. seasonal strategy) remains unresolved. Similarly, no evidence of climate change–induced genetic selection was found by Heath et al. (2012).

Pelagic species have lower geographical barriers and higher capacities to change their distribution (Cheung et al. 2008; Trenkel et al. 2014) due to their wide distribution and pelagic habitat but they are also highly vulnerable to environmental variability, including multidecadal regime shifts (Alheit et al. 2009; Barange et al. 2009; Chavez et al. 2003; Fernandes et al. 2010; Hsieh et al. 2009). This sensitivity to environmental variability is often linked to impacts on the recruitment success of pelagic species (Ibaibarriaga et al. 2007; Hátún et al. 2009; MacKenzie et al. 2012; Payne et al. 2013; Brugel et al. 2016; Tsoukali et al. 2016). Atlantic herring, mackerel and blue whiting are migratory species that occupies nursery grounds during the early life stages and then migrates to feeding grounds (Blaxter and Holliday 1963; Corten 2002; Ruzzante et al. 2006; Volkenandt et al. 2015). The areas of highest negative impact contain many of the key spawning areas of these pelagic species. For example, known herring spawning areas are in the East of Scotland and the Celtic sea (van Damme and Bakker 2014). Although mackerel spawning areas from Iberian Peninsula to the West of Ireland and Scotland (Brunel et al. 2018) would be less affected,

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**Fig. 4** Left panel, model projections of current species distribution (20 years average from 2000 to 2019). Right panel, projected relative changes of potential catches (average of two biochemical models) under the high-emission scenario (RCP8.5) in ICES areas II to VII. The columns represent differences between different futures (2020–2039, 2050–2069, 2080–2099) and the present (2000–2019).
strong impacts in the southern spawning areas around Iberian Peninsula are projected. Blue whiting spawning areas in the West of Ireland and Scotland (Gonçalves et al. 2017) would be less affected with potential positive impacts on local spawning in the Norwegian coast and the coast of Iceland. Similarly, local herring spawning in Norwegian coast might become more suitable (Berg et al. 2017). These results agree with the increase of northern spawning areas already forecasted decades ago (Johnson 1977). Geographical attachment of migratory pelagic species to nursery and spawning areas has been observed and discussed (Petitgas et al. 2006; Brunel et al. 2018). The life history characteristics of the pelagic species considered in this study render them having low to moderate vulnerability to fishing (Cheung et al. 2005; see also www.fishbase.org). These vulnerabilities to environmental variability and fishing complicate the assessment of the impacts of long-term climate change on these species, in terms of both distribution shifts and mortality (Petitgas et al. 2012; Shephard et al. 2014). One of the main examples is the anchoveta (*Engraulis ringens*), whose catches are highly variable and strongly dependent on the state of the El Nino Southern Oscillation index (FAO 2016). The intensive fishing impacts the variations of the stock abundance contributing to the amplification of the change magnitude (Fréon et al. 2008).

Mackerel increases in northern latitudes have had direct economic consequences for Iceland, Greenland, Norway and Scotland and have raised disputes on catches allocations between these countries (Bazilchuk 2010; Cendrowicz 2010; Astthorsson et al. 2012; Jansen et al. 2016; Spijkers and Boonstra 2017). Given the projected trends in mackerel and that other species may follow similar patterns, our work suggests that further disputes for widely distributed pelagic species quotas may occur, as other recent studies (Baudron et al. 2020). Brexit and politics can add difficulties to these tensions depending on the final agreement between UK and EU (Lubchenco and Grorud-Colvert 2015; Boyes and Elliott 2016). This could also have implications for fisheries of higher trophic level species that forage on pelagic species such as cod or haddock, marine mammals and seabirds (Köster et al. 2001; Mullowney and Rose 2014). In addition, these species could follow the pelagic species shift causing further management issues, but also conservation problems with polar species (Renaud et al. 2012).

Small pelagic fisheries in the Northeast Atlantic are managed partly by North-East Atlantic Fisheries Organization (NEAFC) and partly by the EU Common Fishery Policy, whose objective is to maintain or rebuild fish stocks to levels that can produce their MSY. This policy is already yielding stock improvements on European Atlantic fisheries (Cardinale et al. 2013), particularly in relation to widely distributed pelagic species (Fernandes and Cook 2013). For example, Atlantic herring seems to have recovered from overexploitation faster than expected (Nash et al. 2009), though later studies have attributed the recovery to natural variability in the climate system (Drinkwater et al. 2014).

ICES advice works on the stock level for each species giving different values of fishing mortality under MSY for some of the evaluated stocks. Herring historical estimates of sustainable fishing mortality advice (i.e. $F_{MSY}$) for most of the stocks is 0.25 except for a couple of stocks with 0.15 and 0.35 values respectively (ICES Stock Summary Database; http://www.ices.dk). ICES summary database was used to report a value of 0.22 for the mackerel stock ‘mac-nea’, the 2014 ICES advice reports a value of 0.25 (ICES (International Council for the Exploration of the Sea) 2014b) and the latest 2019 report shows a value of 0.23 (ICES (International Council for the Exploration of the Sea) 2019a). For sprat, the 2014 ICES advice report suggests the values 0.29 and the ranges 0.26–0.32 (ICES (International Council for the Exploration of the Sea) 2014a) with the latest advice decreasing these values to the range between 0.19 and 0.27 (ICES (International Council for the Exploration of the Sea) 2019b). The value for blue whiting was increased from 0.18 to 0.30 (ICES (International Council for the Exploration of the Sea) 2014c) and further to 0.32 recently (ICES (International Council for the Exploration of the Sea) 2016a). The fishing mortality $F_{MSY}$ for each of the species used in this study (based on natural mortality), for herring, mackerel, blue whiting and sprat, is 0.21, 0.26, 0.18 and 0.49 respectively. Most of these $F$ values for the whole species distribution are close to the ones reported based on individual stock assessments above, despite for many stocks there is not yet a stablished $F_{MSY}$ value in the stock assessment. There are still many mackerel, sprat and blue whiting stocks exploited above MSY levels or of unknown status (ICES (International Council for the Exploration of the Sea) 2016a; ICES (International Council for the Exploration of the Sea) 2019a, b). Furthermore, research highlights the economic and ecological benefits of fishing levels below MSY (Shephard et al. 2013; Voss et al. 2014; Da-Rocha and Mato-Amboage 2015; Merino et al. 2015).

Due to the difficulties and lack of some stocks data aforementioned, the European Commission is developing a proposal for a multi-stock multiannual plan for the management of fisheries in the Baltic aiming at cod, herring and sprat (amending Council Regulation (EC) No 2187/2005 and repealing Council Regulation (EC) No 1098/2007). The need for multispecies approaches is not only recognized ecologically, but also for an effective ecosystem management approach (Möllmann et al. 2014). The SS-DBEM model considering trophic interactions for 49 species in the North Atlantic has shown a 20% slower latitudinal shift of species than in the single-species approach (Fernandes et al. 2013a). A recent study (Thorpe et al. 2015) highlights that multispecies MSY values can differ by 25–40% compared to the current single-species approaches which confirms previous concerns raised
in Walters et al. (2005). Recent research suggest that forecast of climate change areas of impact can be used to inform multidisciplinary local spatial planning and stakeholder’s actions for of climate-ready management (Fernandes et al. 2017; Queirós et al. 2016; Queirós et al. 2018). These climate-ready approaches use multiple ecosystem models and statistical analysis to identify potential areas where higher changes might occur to be considered by managers and industries in their planning. However, this adds further complexity to the need of transboundary agreements (ICES (International Council for the Exploration of the Sea) 2016b; Krysov et al. 2017) which need to contemplate long-term considerations.

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