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 (*Scomber scombrus*), European sprat *(Sprattus sprattus)*, Atlantic herring (*Clupea harengus*) and blue whiting (*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 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.

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 (Clupea harengus), Atlantic mackerel (Scomber scombrus), European sprat (Sprattus sprattus) and blue whiting (Micromesistius poutassou). Atlantic mackerel (*Scomber scombrus*), European sprat *(Sprattus sprattus)*, Atlantic herring (*Clupea harengus*) and blue whiting (*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 2014-2017, the top 5 exploiters countries for mackerel were also the United Kingdom, Norway, Iceland, Russia and Faeroe Islands, whereas the main sprat catches were from Denmark, Poland, Russia and Latvia. Despite other Bboreal/arctic stocks dealt with, such as summer spawning herring off Iceland, capelin stocks (Greenland/Iceland/Jan Mayen and the Barens Sea), sprat and other southern small pelagic stocks (e.g. anchovy, sardine and horse mackerel) dealt with are far less important in terms of

80   biomass catches, butthey are important for the local economy of local economic importance

81   and have a major ecological role to play in the NEA.

82

83   Sprat (*Sprattus sprattus*) follow regular seasonal migrations and occasionally form huge

84   aggregations (Henderson and Henderson, 2017). Mackerel performs extensive annual

85   migrations with distinct spawning, feeding, overwintering and nursery areas (Boyd et al., 2018;

86   Brunel et al., 2018). Blue whiting performs seasonal and diel vertical migrations (Gonçalves et

87   al., 2017). Herring also undergo extensive migrations between feeding and spawning grounds

88   (Kotterba *et al.*, 2017). Historical collapses and recoveries of pelagic fish stocks have been

89   attributed to a combination of climate and fishing effects and, recruitment success (Planque *et*

90   *al.*, 2010; Fernandes et al., 2010; Fernandes et al., 2015). Besides, overexploitation has been

91   found to decrease fish stock resilience to environmental variability and climate change

92   (Anderson et al., 2008; Bates et al., 2014; Hsieh et al., 2006, Ottersen *et al.*, 2006). Small and

93   intermediate-size pelagic fish species are recognized as key elements in marine food chains

94   (Cury *et al.*, 2011; Kearney *et al.*, 2012) and distribution changes could have significant

95   conservation and management implications (Astthorsson *et al.*, 2012).

96

97   The spawning ability and reproductive cycle of adult herring has often been linked to

98   temperature conditions (Jennings and Beverton, 1991; Winters and Wheeler, 1996; MacKenzie

99   *et al.*, 2007), salinity (Rönkkönen *et al.*, 2004) and food availability (Parmanne *et al.*, 1994;

100 Hufnagl and Peck, 2011). Herring stocks in the NEA tend to have inshore nursery areas (Geffen

101 et al., 2011). Warmer temperature favoured higher sprat abundance (MacKenzie *et al.*, 2007),

102 whereas low salinity reduces its abundance despite the species' tolerance to a wide range of

103 salinity (Peck *et al.*, 2012). Adult mackerel also shows affinity to warm water (Beare and Reid,

104 2002; Jansen *et al.*, 2012; Hughes *et al.*, 2014; Bruge *et al.*, 2016). Predation by mackerel is

105 suggested to affect the distribution of blue whiting (e.g. Payne *et al.*, 2012), and changes in

106 distribution and abundance have also been associated with a strong subpolar gyre (Hátún *et al.*,

107 2009; Payne *et al.*, 2012) and associated oceanographic processes.

108

109 Ecosystem and population models based on size-spectrum theory (Blanchard *et al.*, 2012;

110 Jennings *et al.*, 2008), habitat suitability theory (Kaschner *et al.*, 2006; Phillips *et al.*, 2006) or

111 a combination of both (Fernandes *et al.*, 2013a) are regularly used for projecting future

112 scenarios of widely distributed fish species. These models suggest that temperature and primary

113 production are often the main drivers of change in species' distribution and abundance at global

114 (Jennings *et al.*, 2008; Cheung *et al.*, 2011; Barange *et al.*, 2014; Chust *et al.*, 2014) and

115 regional scales (Fernandes *et al.*, 2017; Fernandes *et al.*, 2016; Speirs et al. 2016). However,

116 despite capturing observed global decadal trends with some success (Fernandes et al., 2013;

117 Jennings and Collingridge, 2016), these models are not precise and realistic enough to inform

118 short-term fisheries management (Dickey-Collas et al., 2014). This is partly because these

119 models do not consider all the drivers and local geographical characteristics that impact specific

120 species or stocks (Planque et al., 2011). Despite the complexity of these models, there are still

121 processes that may be underrepresented, such as top-down effects (Kearney et al., 2012), which

122 are often represented as part of a mortality term. Future projections are not only impacted by

123 uncertainties stemming from unresolved and uncertain processes in the models, but also by

124 natural variability and scenario uncertainty (Payne et al., 2016; Cheung et al., 2016a, Mullon

125 et al., 2016). These points highlight the limitations in predicting the future changes in species

126 biomass and distribution, but it also shows that there is an increased capacity to consider long-

127 term scenarios of change and its implications as well as the short-term forecast improvements

128 (Fernandes et al., 2015; Trifonova et al., 2015).

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131   Considering the importance of widely distributed pelagic species, high mobility, their

132   sensitivity to environmental changes and exploitation patterns and their key role as prey to

133   other resources (Trenkel *et al.*, 2014; Cury *et al.*, 2000), this paper aims to assess the impact of

134   climate change on the long-term potential catches of the most abundant pelagic species and the

135   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 21st century considering multiple mechanisms summarized in Table I and described in 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 multi-decadal 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 abundance with explicit consideration of mechanisms of population dynamics, dispersal (larval and adult) and ecophysiology (see Table I), 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 multi-species version of the model (SS-DBEM) that

160   incorporated species interactions based on size-spectrum (SS) theory and habitat suitability, the

161   SS-DBEM (Fernandes *et al.*, 2013a). Therefore, the model considers predation and food

162   availability through size-spectrum energy transfer from primary producers to consumers of

163   progressively larger body size. Despite yearly outputs, pelagic species have two internal time

164   steps to account for interannual seasonality and both, bottom and surface environmental drivers

165   are considered since these species have pelagic and demersal life stages. For example, herring

166   is a demersal spawner (Lambert and Ware, 1984; Axelsen *et al.*, 2000) whereas mackerel, sprat

167   and blue whiting are pelagic spawners (Coombs et al 1981; Lambert et al., 1984; Fréon et al.,

168   2005).

169

170   Fishing practices are considered in relation to the concept of maximum sustainable yield (MSY;

171   Table I). MSY is defined as the highest average theoretical equilibrium catch that can be

172   continuously taken from a stock under average environmental conditions (Hilborn and Walters,

173   1992). In our application, the fishing mortality under MSY (F_{MSY}) values were obtained from

174   FishBase (www.fishbase.org) assuming that F_{MSY} is approximately equal to 0.4 * M (natural

175   mortality). This equation is used instead of the F_{MSY} from stock assessments because SS-

176   DBEM models the whole species distribution (both inside and outside NEA) instead of

177   modelling individual stocks. Therefore the whole species distribution (both inside and outside

178   NEA) is modelled given that widely distributed species do not follow human management

179   boundaries (Baudron et al., 2020). However, this equation values are in general consistent with

180   values from stock assessments as reviewed below in the discussion section.

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182   The larvae recruitment depends on temperature and currents that are known mechanisms

183   affecting the mortality or success of herring, mackerel, blue whiting and sprat (Alvarez and

184   Chifflet, 2012; Fernandes et al., 2015; Huse. 2016; Martin et al., 2016; Henderson and

185   Henderson, 2017). The model calculates larvae dispersal through ocean currents and diffusion

186   assuming that pelagic larvae disperse passively from surrounding 'source' areas through ocean

187   surface current and diffusion (Cheung et al., 2008; Table I). The magnitude of larval

188   recruitment is dependent on pelagic larval duration (PLD), strength and direction of ocean

189   currents and diffusivity (Table I). PLD, expressed in days, is calculated from an empirical

190   equation established from a meta-analysis of PLD from 72 species of fish and invertebrates

191   (O'Connor et al. 2007). Based on the calculated PLD and ocean current velocity data, the model

192   calculates dispersal of pelagic larvae over time through diffusion and advection. Diffusion and

193   advection of ocean currents are important factors determining dispersal of pelagic larvae of

194   marine organisms (Possingham & Roughgarden 1990; Gaylord & Gaines 2000; Bradbury &

195   Snelgrove 2001; Gaines et al. 2003). The temporal and spatial patterns of pelagic larval

196   dispersal were modelled by a two-dimensional advection-diffusion equation (Table I; Sibert et

197   al. 1999; Gaylord & Gaines 2000; Hundsdorfer & Verwer 2003).

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200

201   Ocean biogeochemical models

202   Ocean environmental conditions from two ocean biogeochemical models of different

203   complexity were used to force the species-based fisheries model SS-DBEM: the NEMO-

204   MEDUSA (NEMO, Nucleus for European Modelling of the Ocean and MEDUSA, Model of

205   Ecosystem Dynamics, nutrient Utilisation, Sequestration and Acidification, Yool *et al.*, 2013)

206   and the GFDL ESM2M (GFDL, Geophysical Fluid Dynamics Laboratory and, ESM2, Earth

207 System Model z coordinate, Dunne *et al.*, 2012; Dunne *et al.*, 2013).;.Both ocean

208 biogeochemical models simulate changes in physical and biogeochemical ocean conditions

209 over the historical period and the 21st century under two different greenhouse gas scenarios

210 (RCP2.6 and RCP8.5). These ocean conditions include seawater temperature, salinity, oxygen,

211 alkalinity, primary production and horizontal and vertical water currents and affect the

212 ecological processes in the SS-DBEM model such as the life-history, habitat, population

213 dynamics and dispersal (Fernandes et al., 2013a; Queirós et al., 2016).

214

215 *NEMO-MEDUSA biogeochemical model*

216 The NEMO-MEDUSA is a global ocean model (Yool et al., 2013) with a half-degree ocean

217 resolution. It simulates the physical environmental conditions and a size-structured ecosystem

218 of small phytoplankton and zooplankton. The NEMO-MEDUSA explicitly includes the

219 biogeochemical cycles of nitrogen, silicon and iron nutrients as well as the cycles of carbon

220 which are not used directly by the fish model but influence the primary production. The

221 NEMO-MEDUSA model was initialized using standard biogeochemical climatological fields

222 (Garcia et al., 2010; Key et al., 2004) and simulated under surface atmospheric forcing derived

223 from the Met Office Unified Model simulations (HadGEM2 configuration).

224

225 *GFDL biogeochemical model*

226 The GFDL ESM2M is a global coupled atmosphere-ocean general circulation model (Dunne

227 et al. 2012, 2013) including a marine biogeochemistry model. The global ocean model has an

228 approximately 1° horizontal resolution and it describes the cycles of carbon, nitrogen,

229 phosphorus, silicon, iron, oxygen, alkalinity and lithogenic material and considers three

230 phytoplankton functional groups.

231 *Greenhouse gas emission scenarios*

232 The two ocean biogeochemical models were run under two greenhouse gas emission scenarios

233 (Moss et al., 2010): a strong mitigation scenario (Representative Concentration Pathways

234 RCP2.6) with an atmospheric CO_2 concentration of 421 ppm by the end of the 21st century, and

235 "the business as usual" high greenhouse gas emissions scenario (RCP8.5) with atmospheric

236 CO_2 concentrations of 936 ppm by the end of this century.

237

238 Validation of the species projections

239 Standardised and long-term fisheries surveys from 1977 to 2007, collated by Simpson et al.

240 (2011), were used to cover the breadth of fisheries for this study. The survey data available for

241 validation included AFBI Irish Sea Q1 and Q3, Cefas Celtic Sea, Cefas North Sea (autumn)

242 and ICES IBTS North Sea (spring) datasets. The raw data are now freely available at the ICES

243 website (www.ices.dk/marine-data/data-portals/Pages/DATRAS.aspx). Data from demersal

244 surveys are widely used to estimate abundance (e.g. ICES, 2014d; Peck et al., 2013) and

245 distribution (e.g. Huse et al., 2008; Jansen et al., 2012) of pelagic species. Given that blue

246 whiting, sprat and herring have all been shown to undertake diel migrations (Cardinale et al.,

247 2003; Post et al., 2019) resulting in greater day trawl catches (Petrakis et al., 2001) and that

248 survey trawl data has been shown to be a reliable indicator of pelagic species abundance (Fig.4,

249 Suppl. Mat.; Montero-Serra et al., 2015; ICES, 2015; Peck et al., 2013) and distribution (Huse

250 et al., 2008; Jansen et al., 2012), the dataset used in this study are considered more reliable

251 when compared to landings (herring) and acoustic (sprat) data (Fig.4, Suppl. Mat.; Montero-

252 Serra et al., 2015). Despite these studies, it is acknowledged that bottom trawl surveys are not

253 designed to target pelagic species, and that additional abundance estimates should be extracted

254 from acoustic surveys (e.g. ICES, 2015). However, acoustic data is not consistently available

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255 over the study area and timeframe despite recent important advances in its harmonization and

256 public availability are promising for its use in ecosystem modelling and validation. For

257 example, the ICES Working Group on Acoustic and Egg Surveys for Sardine and Anchovy in

258 ICES areas 7, 8 and 9 (WGACEGG) is developing a not yet public protocol with the title

259 "Manual for Acoustic Surveys in Ices Areas 6, 7, 8 and 9".

260

261 The total number of individuals for each species was tallied across size classes for each survey

262 haul. An average catchability estimate was applied to similar pelagic species for all size groups

263 using total biomass and catch estimates from Sparholt (1990):

264

265 Corrected abundance= uncorrected abundance * (1 / catchability)

266

267 To control for the differing effort between surveys the swept area for each haul was

268 calculated using estimates of wing-spread for Grand Ouverture Vertical (GOV) trawls from

269 Fraser et al. (2007) multiplied by the distance over which the hauls were undertaken scaled to

270 km^2 following the equation:

271

272 Area swept km^2 = (((6.85 * (log(depth))) + 5.89) * distance) / 1000000

273

274 Depth and distance were measured in meters, being the distance based on haul duration and

275 speed (assumed tow speed 4 knots for duration of haul, except for North Ireland data where the

276 data is defined as the number of individuals/3nm).

277

278 All the survey data is aggregated into a cell grid of 0.5 x 0.5 degrees to match the fish model

279 grid. To compare projected changes with observations from surveys, a time-series per cell for

280 each species have been compared. Besides, time-series aggregating all the spatial cells with

281 survey data for each species is compared to estimate the long-term performance of wider areas.

282 The time-series are normalized between 0 and 1 to consider that the model projects relative

283 change and to have error estimates in an interval that are easier to understand. Since multiple

284 species at multiple cells were considered, we needed to ensure that results are comparable,

285 therefore,time-series of survey data with more than 3 years of missing data were not included.

286 As a result, not enough data to assess the simulated changes in blue whiting was available.

287 Then, time-series from the models were extracted for those years, species and cells where there

288 was commonly available data from the surveys at the 0.5x0.5 degrees and yearly resolution.

289 This restricted the data that could be analysed to the period from 1982 to 2007 (26 years) and

290 the following 3 pelagic species (out of the 4 modelled here): herring, mackerel and sprat. These

291 time-series were produced for both, the survey data and the model projections, and were

292 compared by calculating the average error (AE) between them:

293

294 $$AE = \frac{\sum_j |p_j - s_j|}{j},$$

295

296 where, p is the biomass projected in the SS-DBEM model in a particular year for each species,

297 and s is the biomass from the survey and j is the number of years with data.

Results

<u>Performance of the fish projections</u>

General trends are well simulated by the models when aggregating over big areas (Table II and Fig.2). However, local scale simulations are more complicated (grid cell level). Errors at cells level range between 0.43 and 0.49, whereas errors aggregating all the cells with survey data drops to the range 0.23 to 0.41 (Table II). 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 inter-annual variability. This can be observed in Figure 2 where variability projected by the model and that observed in surveys does 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.

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

321   Herring catches were projected to vary by ±5-10% relative to the present day (2015) averaged

322   over the North-East Atlantic, but the projections diverge between the two gas emission

323   scenarios from 2070 onwards (Fig. 3a). Under the low greenhouse gas emission scenario,

324   herring is projected to increase by up to 15% by the end of this century, while under the high

325   greenhouse gas emission scenario, herring is projected to decrease by up to 35%. On the

326   contrary, mackerel was projected to increase in both future scenarios: up to 5-10% until 2040

327   (Fig. 3b) and much higher increases 15-30% by the end of the century. The model simulates

328   increases in catch trends for mackerel (Fig. 3b) and sprat (Fig. 3c). However, projections of

329   sprat have larger uncertainties in the first decades than other species projections. Sprat would

330   be benefited from changes in environmental conditions under a low gas emission scenario and

331   showing the most significant differences across. Catch of blue whiting is projected to increase

332   by up to 10% in a low emission scenario but it is projected to decrease by up to 40% under a

333   high emission scenario (Fig. 3d). However, there is considerable uncertainty in projections

334   under the high scenario and results need to analyse carefully.

335

336   Catch projections for all the pelagic species show consistent increases under a low emission

337   scenario by the end of the century (Fig. 3). The projections under the high emissions scenarios,

338   however, are more uncertain than low emission scenarios and sometimes do not even agree on

339   the sign of changes. Herring and blue whiting show the lowest uncertainty in the medium term

340   to 2040 (Fig. 3a,d), whereas mackerel and sprat show higher uncertainty in the medium term

341   between 2040 and 2080 (Fig. 3b,c). The uncertainties in potential catches are mainly driven by

342   the high uncertainty in primary production projections (Table III). Despite a general increase

343   in temperature in all ICES areas, northern areas experience generally lower increases than

344   southern areas in the shorter term (2020-2039). Primary production increases in areas II and

345   III and decreases everywhere else despite high uncertainty. Area III, which contains the highest

346   increases, shows the lowest uncertainty in terms of primary production estimates.

347   Figure 4 shows the changes in species distributions for the periods 2020-2039, 2050-2069 and

348   2080-2099 relative to the present baseline period (2000-2019) under the high-emission

349   scenario. Blue whiting, herring and mackerel have a potential for habitat displacement towards

350   the Arctic Ocean, with reductions in the southern area of distribution, especially for herring

351   and blue whiting. For capelin there are no agreement between the models since GFDL projects

352   in general slight increases for both emission scenarios with decadal oscillations, whereas

353   Medusa show strong declines in the high emission scenarios for most of the ICES areas (higher

354   declines in northern areas of NEA). Sprat shows a decline in catch potential in areas IV, VI and

355   VII, but not a corresponding increase in northern regions. There is a consistent pattern showing

356   that northern latitudes will benefit whereas southern areas will be negatively impacted in terms

357   of all species catches. ICES area IIb is the area that shows the highest increases with gains in

358   blue whiting, herring and sprat. Sprat increases are concentrated at the more southerly end of

359   the northern regions IIa and V, areas that show other three species. Herring is expected to show

360   the highest changes by 2020-2039 and 2050-2069, both positive and negative, in terms of the

361   amount of area experiencing changes. Mackerel will be the second species more impacted by

362   2050-2069. blue whiting is projected to increase and sprat to decrease, with a few local

363   exceptions. Herring and mackerel show the highest changes with dramatic northern shifting by

364   the end of the century. Overall, the lower emission scenarios show similar spatial patterns in

365   terms of areas of highest increases and decreases for each species with some differences: (i)

366   slower rates of changes with almost no change by 2020-2039, (ii) changes in 2050-2069 similar

367   to the high emission scenario between 2020-2039 and (iii) changes by the end of the century

368   similar to the mid-century projections under the high emission scenario. The sparse catches

369   have concentrated mostly in the south-western areas. However, the potential catches decrease

370   projected by the model in the future is consistent with the trend observed in the Baltic (ICES

371   area III) catch data. Therefore, the model cannot be trusted to forecast the distribution of species

372   precisely, but it is good for general trends over wide areas (e.g. ICES areas).

373

374

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 III) 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 21st 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

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399 conditions between the two models under the same future scenario may be explained by

400 internal natural variability uncertainty or model uncertainty (Walters et al., 2005; Hawkins and

401 Sutton, 2009; Frölicher et al., 2016; Cheung et al., 2016a; Frölicher et al., 2016).

402 Figure 2 shows that we can trust more the model for herring and sprat because they are more

403 able to reproduce the historical catches (goodness of fit), also considering that the SS-DBEM

404 is not a statistical model where catch data has been used to drive the model (generalization

405 power). However, it seems the model is less reliable for mackerel in comparison with the other

406 species. Figure 2 does not show high uncertainty in the historical projections since there is

407 small difference between the model run trends except for sprat that shows higher differences.

408 Nevertheless, higher uncertainty is shown in the projections (Fig. 3), so that scenario

409 uncertainty is expected to be higher than model internal variability. This study results are in

410 agreement with empirical work which confirms that projected increases of mackerel in the

411 Svalbard Archipelago (Berge et al., 2015). Furthermore, evidence of mackerel distribution

412 changes in association with warmer temperatures across the North Atlantic have been recorded

413 (Overholtz *et al.*, 2011; Hughes *et al.*, 2014; Montero-Serra *et al.*, 2015(Montero-Serra et al.,

414 2015). Sprat abundance and size has been declining since 1980 (Henderson and Henderson,

415 2017) due to changes in temperature and global climate patterns in the Bristol Channel which

416 contains spawning and overwinter areas. A distribution shift of fish species can occur due to

417 changes in habitat suitability, but there are additional processes involved (e.g. geographical

418 attachment, species interactions or demographic structure) which are needed for a shift to

419 happen or which can limit it (Planque *et al.*, 2011). Therefore, a species may be able to move

420 to new areas (Nøttestad *et al.*, 2016) where it has not been previously observed or species can

421 increase their abundance dramatically in areas where they have rarely observed due to density

422 changes (Petitgas *et al.*, 2012; Punzón *et al.*, 2016). Both situations would produce a shift in

423 the centroid of the species distribution. The SS-DBEM model used considers many of these

19

424 processes as detailed in Planque *et al.* (2011). The modelled shift in distribution is determined

425 by changes in predicted carrying capacity of the species in each grid cell, while its shifts are

426 the result of the spatial population dynamics (Cheung *et al.*, 2016b). Firstly, carrying capacity

427 of the population in a cell is positively related to habitat suitability and the energy from primary

428 production that is available for the species with consideration of competition between species

429 in the same cell which can limit shifts speed (Fernandes *et al.*, 2013). Secondly, diffusion of

430 the populations is also related to the gradient of habitat suitability between adjacent cells,

431 resulting in an increase in net diffusion out of the cell if habitat suitability in adjacent cells is

432 higher. Thirdly, population recruitment is determined by the dispersal of larvae which is

433 dependent on surface ocean advection and pelagic larval duration, the latter is a function of

434 temperature. Thus, as ocean conditions change, these three processes result in the change in the

435 distribution of abundance of the species.

436

437 The SS-DBEM model used do not account for the adaptation capacity of the species to changes

438 in environmental conditions, due to the sparse knowledge available. This could slow the rate

439 of changes or limit the spatial occurrence of these changes. Engelhard and Heino (2004)

440 demonstrated substantial changes in phenotypic growth and maturation in herring, but a weak

441 evolutionary response. While some studies investigate differences in spawning herring

442 populations using genetic proxies (Jørgensen et al., 2005; King et al., 1987; McPherson et al.,

443 2003), herring spawning tactics also appear to be influenced by phenotypic variability

444 (Jennings and Beverton, 1991; Winters and Wheeler, 1996). Herring have probably adopted

445 different spawning seasons as one of their survival strategies (Melvin et al., 2009; Sinclair and

446 Tremblay, 1984). Although it is established that in herring the process of first maturation is

447 triggered by a combination of physiological (size condition of fish) and environmental cues

448 such as temperature (Winters and Wheeler, 1996) and photoperiodic cycles (McPherson and

449   Kjesbu, 2012), the relative influence of genetics and environment on herring spawning

450   behaviour (i.e. seasonal strategy) remains unresolved. Similarly, no evidence of climate

451   change-induced genetic selection was found by Heath et al. (2012).

452

453   Pelagic species have lower geographical barriers and higher capacities to change their

454   distribution (Cheung et al., 2008; Trenkel *et al.*, 2014) due to their wide distribution and pelagic

455   habitat but they are also highly vulnerable to environmental variability, including multi-decadal

456   regime shifts (Alheit *et al.*, 2009; Barange *et al.*, 2009; Chavez *et al.*, 2003; Fernandes *et al.*,

457   2010; Hsieh *et al.*, 2009).   This sensitivity to environmental variability is often linked to

458   impacts on the recruitment success of pelagic species (Ibaibarriaga *et al.*, 2007; Hátún *et al.*,

459   2009; MacKenzie *et al.*, 2012; Payne *et al.*, 2013; Bruge *et al.*, 2016; Tsoukali *et al.*, 2016).

460   Atlantic herring, mackerel and blue whiting are migratory species that occupies nursery

461   grounds during the early life stages and then migrates to feeding grounds (Blaxter and Holliday,

462   1963; Corten, 2002; Ruzzante *et al.*, 2006; Volkenandt *et al.*, 2015). The areas of highest

463   negative impact contain many of the key spawning areas of these pelagic species. For example,

464   herring known spawning areas are in the East of Scotland and the Celtic sea (Damme and

465   Bakker, 2014).   Although, mackerel spawning areas from Iberian Peninsula to the West of

466   Ireland and Scotland (Brunel et al., 2018) would be less affected, strong impacts in the southern

467   spawning areas around Iberian Peninsula are projected. Blue whiting spawning areas in the

468   West of Ireland and Scotland (Gonçalves et al., 2017) would be less affected with potential

469   positive impacts on local spawning in the Norwegian coast and the coast of Iceland. Similarly,

470   local herring spawning in Norwegian coast might become more suitable (Berg et al., 2017).

471   These results agree with the increase of northern spawning areas already forecasted decades

472   ago (Johnson, 1977). Geographical attachment of migratory pelagic species to nursery and

473   spawning areas has been observed and discussed (Petitgas et al., 2006; Brunel et al., 2017).

21

474   The life history characteristics of the pelagic species considered in this study render them

475   having low to moderate vulnerability to fishing (Cheung *et al.*, 2005; see also

476   www.fishbase.org). These vulnerabilities to environmental variability and fishing complicates

477   the assessment of the impacts of long-term climate change on these species, both in terms of

478   distribution shifts and mortality (Petitgas et al., 2012; Shephard et al., 2014). One of the main

479   examples is the anchoveta (*Engraulis ringens*), whose catches are highly variable and strongly

480   dependent on the state of the El Nino Southern Oscillation index (FAO, 2016). The intensive

481   fishing impacts in the variations of the stock abundance contributing to the amplification of the

482   change magnitude (Fréon et al., 2008).

483

484   Mackerel increases in northern latitudes have had direct economic consequences for Iceland,

485   Greenland, Norway and Scotland and have raised disputes on catches allocations between these

486   countries (Bazilchuk, 2010; Cendrowicz, 2010; Astthorsson et al. 2012; Jansen et al., 2016;

487   Spijkers and Boonstra, 2017). Given the projected trends in mackerel and that other species

488   may follow similar patterns, our work suggests that further disputes for widely distributed

489   pelagic species quotas may occur, as other recent studies (Baudron et al., 2018). Brexit and

490   politics can add difficulties to these tensions depending on the final agreement between UK

491   and EU (Lubchenco and Grorud-Colvert, 2015; Boyes and Elliott, 2016). This could also have

492   implications for fisheries of higher trophic level species that forage on pelagic species such as

493   cod or haddock, marine mammals and seabirds (Köster et al., 2001; Mullowney and Rose,

494   2014). In addition, these species could follow the pelagic species shift causing further

495   management issues, but also conservation problems with polar species (Renaud et al., 2012).

496

497   Small pelagic fisheries in the Northeast Atlantic are managed partly by North-East Atlantic

498   Fisheries Organization (NEAFC) and partly by the EU Common Fishery Policy, whose

499   objective is to maintain or rebuild fish stocks to levels that can produce their MSY. This policy

500   is already yielding stock improvements on European Atlantic fisheries (Cardinale et al., 2013),

501   particularly in relation to widely distributed pelagic species (Fernandes and Cook, 2013). For

502   example, Atlantic herring seems to have recovered from overexploitation faster than expected

503   (Nash et al., 2009), though later studies have attributed the recovery to natural variability in the

504   climate system (Drinkwater et al., 2014).

505

506   ICES advice works on the stock level for each species giving different values of fishing

507   mortality under MSY for some of the evaluated stocks. Herring historical estimates of

508   sustainable fishing mortality advice (i. e., Fmsy) for most of the stocks is 0.25 except for a

509   couple of stocks with 0.15 and 0.35 values respectively (ICES Stock Summary Database;

510   http://www.ices.dk). ICES summary database used to reports a value of 0.22 for the mackerel

511   stock "mac-nea", the 2014 ICES advice report a value of 0.25 (ICES, 2014b) and the latest

512   2019 report shows a value of 0.23 (ICES, 2019a). For sprat the 2014 ICES advice report

513   suggests the values 0.29 and the ranges 0.26-0.32 (ICES, 2014a) with the latest advice

514   decreasing these values to the range between 0.19 and 0.27 (ICES, 2019b). The value for blue

515   whiting was increased from 0.18 to 0.30 (ICES, 2014c) and further to 0.32 recently (ICES,

516   2016a). The fishing mortality F_{MSY} for each of the species used in this study, (based on natural

517   mortality) for herring, mackerel, blue whiting and sprat, is 0.21, 0.26, 0.18 and 0.49

518   respectively. Most of these F values for the whole species distribution are close to the ones

519   reported based on individual stock assessments above, despite for many stocks there is not yet

520   a stablished Fmsy value in the stock assesment. There are still many mackerel, sprat and blue

521   whiting stocks exploited above MSY levels or of unknown status (ICES, 2016a; ICES, 2019a;

522   ICES, 2019b). Furthermore, research highlights the economic and ecological benefits of fishing

523   levels below MSY (Shephard et al., 2013; Voss et al., 2014; Da-Rocha and Mato-Amboage,

524   2015; Merino et al. 2015).

525

526   Due to the difficulties and lack of some stocks data aforementioned, the European Commission

527   is developing a proposal for a multi-stock multiannual plan for the management of fisheries in

528   the Baltic aiming at cod, herring and sprat (amending Council Regulation (EC) No 2187/2005

529   and repealing Council Regulation (EC) No 1098/2007). The need for multi-species approaches

530   is not only recognized ecologically, but also for an effective ecosystem management approach

531   (Möllmann et al., 2014). The SS-DBEM model considering trophic interactions for 49 species

532   in the North Atlantic has shown a 20% slower latitudinal shift of species than in the single-

533   species approach (Fernandes et al., 2013a). A recent study (Thorpe et al., 2015) highlights that

534   multi-species MSY values can differ by 25-40% compared to the current single-species

535   approaches which confirms previous concerns raised in Walters *et al.* (2015). Recent research

536   suggest that forecast of climate change areas of impact can be used to inform multidisciplinary

537   local spatial planning and stakeholder's actions for of climate-ready management (Fernandes

538   et al., 2017; Queirós et al., 2016; Queirós et al., 2018). These climate-ready approaches use

539   multiple ecosystem models and statistical analysis to identify potential areas where higher

540   changes might occur to be considered by managers and industries in their planning. However,

541   this adds further complexity to the need of transboundary agreements (ICES, 2016b; Krysov et

542   al., 2017) which need to contemplate long-term considerations.

543

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Tables

Table I. 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 (Pauly 2010; Cheung et al., 2011)	$G = HW^a - kW$ $H = g[O_2] * e^{-j1/T}$ $k = h[H^+] * e^{-j2/T}$	H = anabolism coefficient k = catabolism coefficient W = body weight a = anabolism exponent (0.5 to 0.95) W_∞ = asymptotic weight The coefficients g and h were derived from the average W_1, K, and environmental temperature (T) of the species reported in the literature.
Length-Weight	$W = a * L^b$	W = weight L = length
Size-spectrum production (Jennings et al., 2008; Fernandes et al., 2013)	$P = \exp(25.22 - E/kT) * W^{0.76}$	E = activation energy of metabolism k = Boltzmann's constant T = temperature in Kelvin (°C+273)
Intrinsic population growth rate (Hilborn & Walters,1992)	$G = r * A * (1 - (A/KC))$	r = intrinsic rate of population increase A = the relative abundance KC = population carrying capacity
Larval recruitment (O'Connor et al., 2007; Cheung et al., 2008)	$\ln(PLD) = \beta_0 - 1368 \cdot (\ln(T/T_c)) - 0.283_2 \cdot (\ln(T/T_c))^2$ $\beta_0 = 0.739 + 0.739 \cdot (\overline{\ln(T)}) + 0.714 \cdot (DM)$ $\overline{\ln(T)} = \dfrac{\sum_{i=1}^{N} \ln(T_i)}{N}$	PLD = pelagic larvae duration T = surface temperature Tc =15 C DM is the developmental type of larvae (0 lecithotrophic, 1 planktotrophic) N = number of cells where species occur
Larval dispersal (Hundsdorfer & Verwer 2003; Cheung et al., 2008)	$\dfrac{\partial Lav}{\partial t} = \dfrac{\partial}{\partial x}\left(D\dfrac{\partial Lav}{\partial x}\right) + \dfrac{\partial}{\partial y}\left(D\dfrac{\partial Lav}{\partial y}\right) - \dfrac{\partial}{\partial x}(u \cdot Lav) - \dfrac{\partial}{\partial y}(v \cdot N) - \lambda \cdot Lav$	D = diffusion parameter (u, v) = velocity parameters LAV = larvae recruitment
Adult movement	$Cm * h^{-1}$	Cm = centimetre h = hour
Natural mortality (Pauly, 1980; Cheung et al., 2011)	$M = -0.4851 - 0.0824 * \log(Winf) + 0.6757 * \log(K) + 0.4687 * \log(T)$	Winf = asymptotic weight K = von Bertalanffy growth parameter T = average water temperature in the animal's range.
Fishing mortality at MSY	$F_{MSY} = 0.4 * M$	M = Natural mortality

Table II. Estimation of the average error (0-1 range) and standard deviation by comparing time-series of survey data with modelled data at cells level (by cell) and by aggregating all the cells for each species with survey data in the NEA (cell sum).

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 ± 0.79
Sprat	By cell	0.49 ± 0.17	0.47 ± 0.16
	Cell sum	0.31 ± 0.29	0.26 ± 0.22

Table III. Mean and variance of sea surface temperature (SST) and primary production (PP) changes by ICES areas. Current temperature and differences between different futures (2020-2039, 2050-2069, 2080-2099) and present (2000-2019). Temperature is in Celsius degrees and primary production in percentage change. Northern areas are shadowed. Negative average primary production is highlighted in bold.

		2020-2039		2050-2069		2080-2099	
Area	Scenario	SST	PP	SST	PP	SST	PP
II	RCP2.6	+0.3 ±0.1	+1.9 ±0.4	+0.7 ±0.8	+5.0 ±5.8	+0.5 ±0.8	+1.9 ±2.2
	RCP8.5	+0.1 ±0.1	+0.0 ±1.6	+1.4 ±1.1	+9.1 ±6.2	+2.6 ±2.0	+11 ±3.7
III	RCP2.6	+0.9 ±0.1	+4.2 ±3.3	+0.9 ±0.6	+10 ±12	+0.8 ±0.5	+5.2 ± 26
	RCP8.5	+0.8 ±0.1	+5.2 ±6.1	+2.0 ±0.7	+13 ±21	+3.6 ±1.1	+17 ± 30
IV	RCP2.6	+0.5 ±0.7	**-10 ±15**	+0.5 ±0.3	**-12 ±12**	+0.2 ±0.3	**-14 ±15**
	RCP8.5	+0.5 ±0.1	**-6.4 ±9.3**	+1.2 ±0.5	**-11 ±12**	+2.3 ±0.9	**-14 ±10**
V	RCP2.6	+0.2 ±0.0	**-6.0 ±5.7**	-0.1 ±0.3	**-11 ±10**	-0.3 ±0.1	**-12 ±11**
	RCP8.5	-0.1 ±0.5	**-4.3 ±1.5**	0.0 ±0.1	**-15 ±17**	+0.6 ±0.8	**-17 ±16**
VI	RCP2.6	+0.2 ±0.1	**-11 ±12**	+0.2 ±0.1	**-19 ±24**	0.0 ±0.1	**-21 ±26**
	RCP8.5	+0.2 ±0.2	**-8.3 ±9.2**	+0.3 ±0.1	**-23 ±30**	+0.4 ±0.5	**-32 ±28**
VII	RCP2.6	+0.3 ±0.2	**-12 ±13**	+0.3 ±0.2	**-17 ±23**	+0.3 ±0.2	**-19 ±26**
	RCP8.5	+0.4 ±0.2	**-8.4 ±10**	+1.0 ±0.8	**-21 ±26**	+2.0 ±1.6	**-27 ±26**

Figures

Figure 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).

Figure 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.

Figure 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.

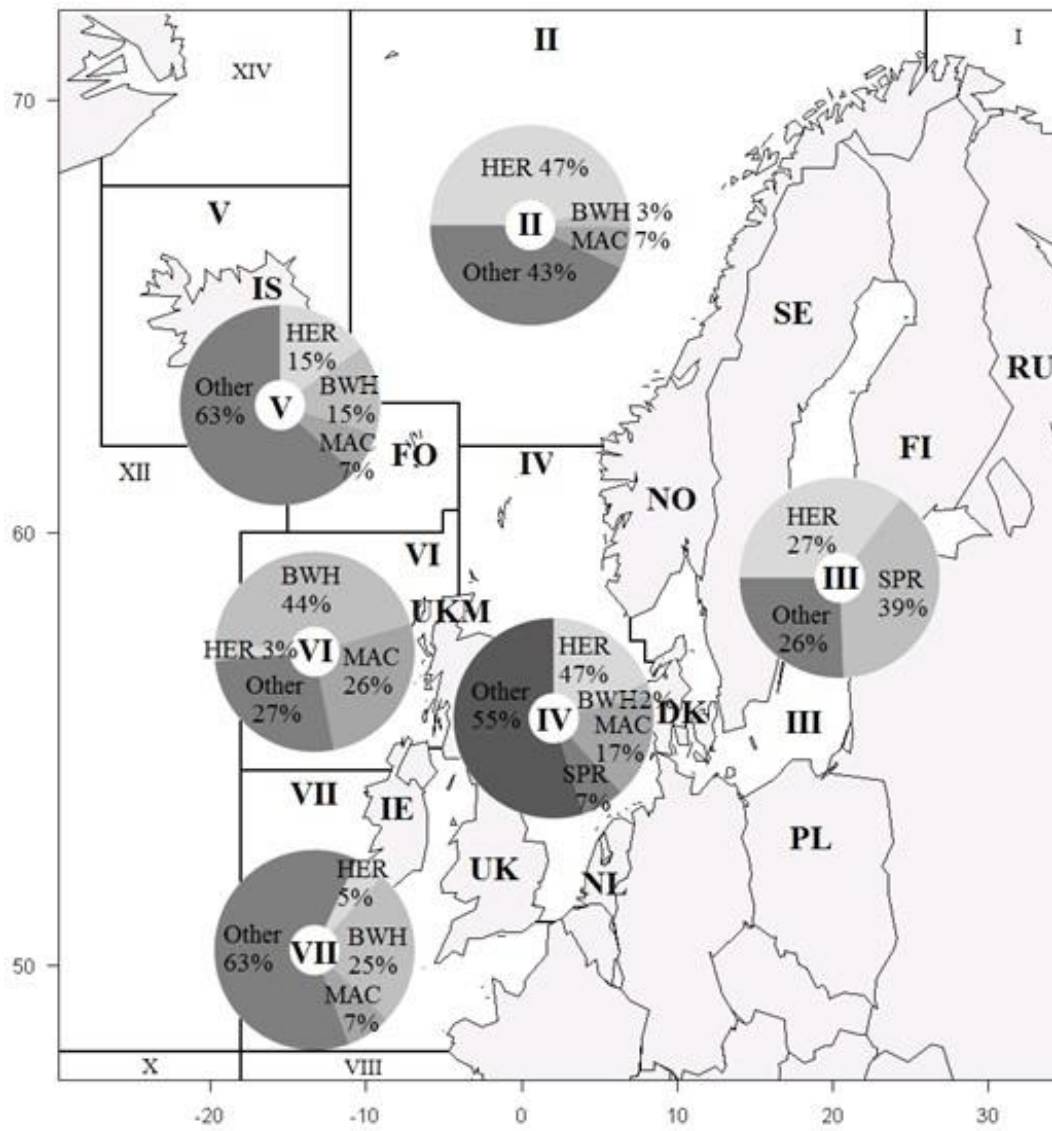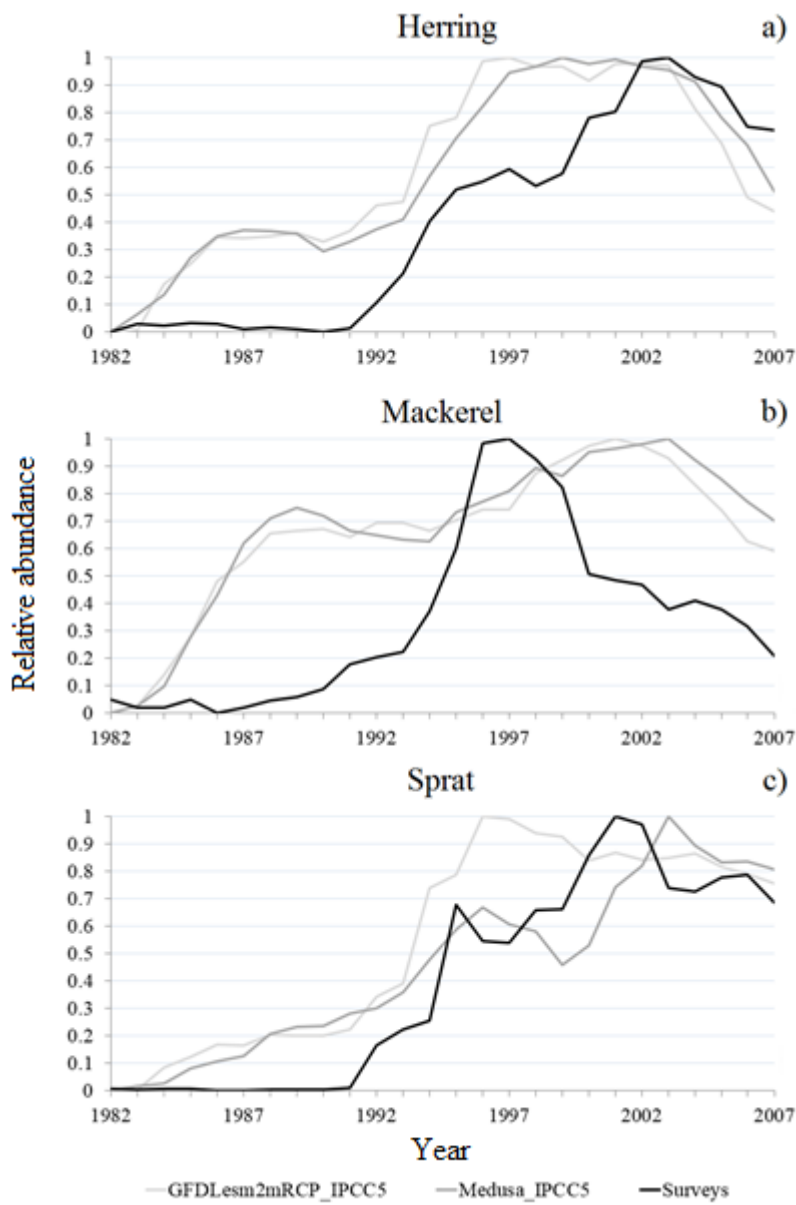Figure 4. Left panel, model projections of current species distribution (20 years average from 2000 to 2019). Right panel, projected relative changes of potential catches (average of two biochemical models) under the high emission scenario (RCP8.5) in ICES areas II to VII. The columns represent differences between different futures (2020-2039, 2050-2069, 2080-2099) and the present (2000-2019).

Fig. 1

1038   Fig. 2

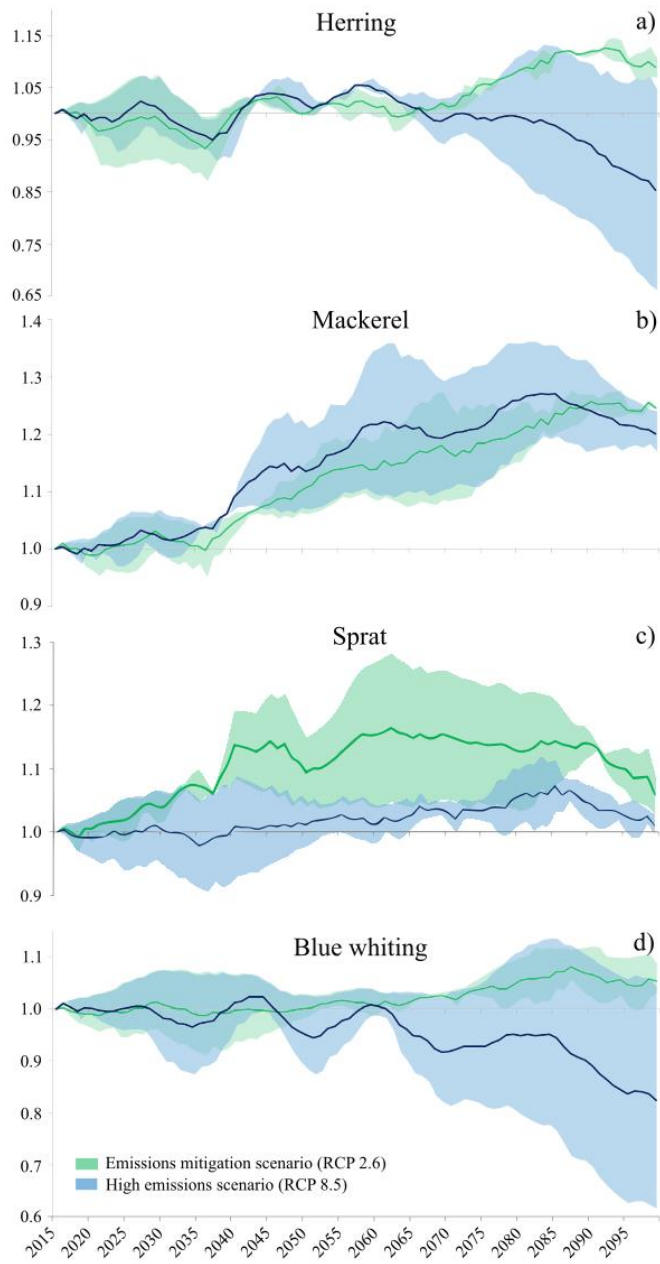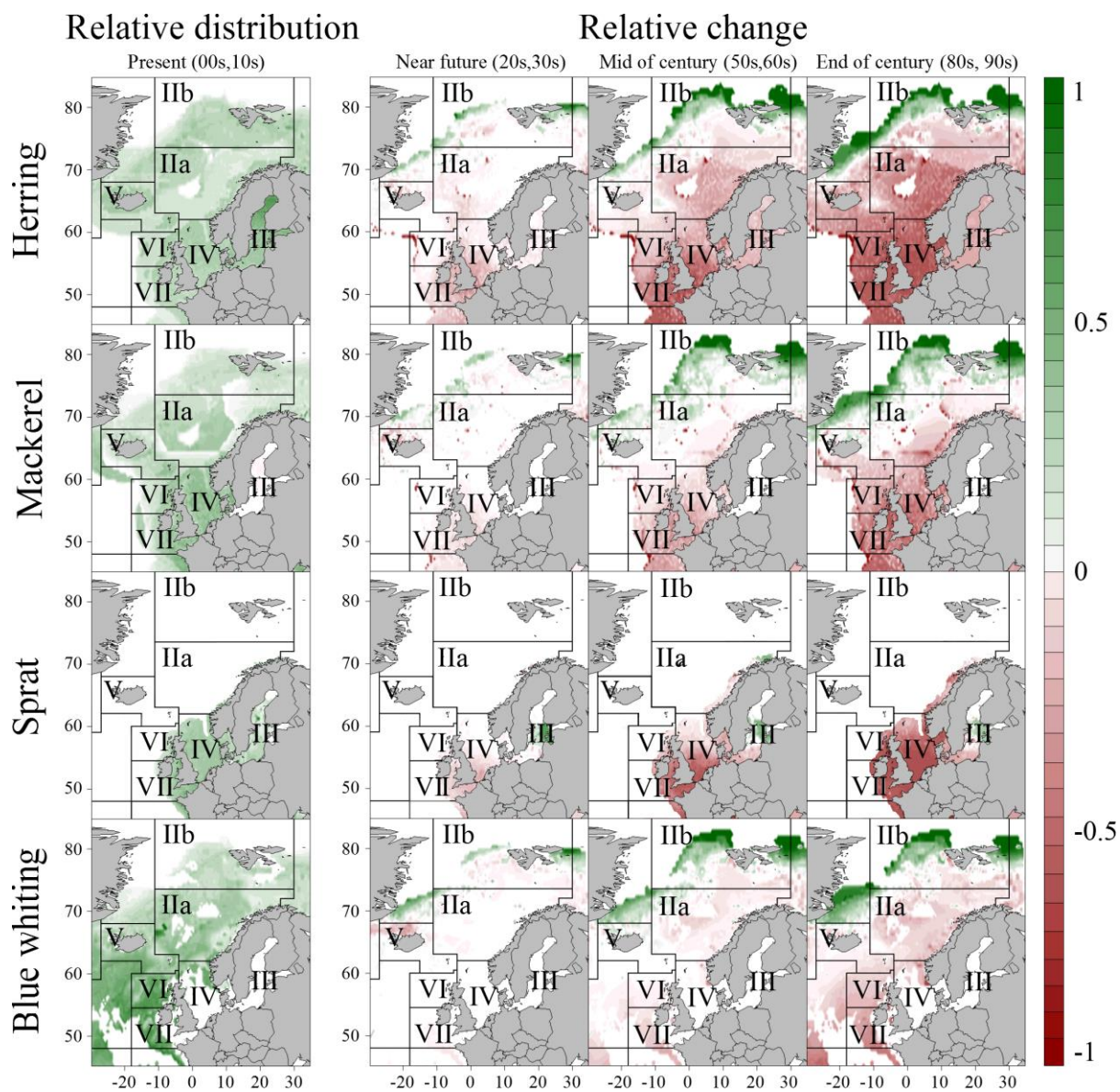Herring a)
Mackerel b)
Sprat c)

Relative abundance

GFDLesm2mRCP_IPCC5   Medusa_IPCC5   Surveys

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1043   Fig. 4-

Relative distribution   Relative change

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