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Estimating the ecological, economic and social impacts of ocean acidification and warming on UK fisheries

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Abstract

Assessments of the combined ecological impacts of ocean acidification and warming (OAW) and their social and economic consequences can help develop adaptive and responsive management strategies in the most sensitive regions. Here, available observational and experimental data, theoretical, and modelling approaches are combined to project and quantify potential effects of OAW on the future fisheries catches and resulting revenues and employment in the UK under different CO_2 emission scenarios. Across all scenarios, based on the limited available experimental results considered, the bivalve species investigated were more affected by OAW than the fish species considered, compared with ocean warming alone. Projected standing stock biomasses decrease between 10 and 60%. These impacts translate into an overall fish and shellfish catch decrease of between 10 and 30% by 2020 across all areas except for the Scotland >10 m fleet. This latter fleet shows average positive impacts until 2050, declining afterwards. The main driver of the projected decreases is temperature rise (0.5–3.3 °C), which exacerbate the impact of decreases in primary production (10–30%) in UK fishing waters. The inclusion of the effect of ocean acidification on the carbon uptake of primary producers had very little impact on the projections of potential fish and shellfish catches (<1%). The <10 m fleet is likely to be the most impacted by-catch decreases in the short term (2020–50), whereas the effects will be experienced more strongly by the >10 m fleet by the end of the century in all countries. Overall, losses in revenue are estimated to range between 1 and 21% in the short term (2020–50) with England and Scotland being the most negatively impacted in absolute terms, and Wales and North Ireland in relative terms. Losses in total employment (fisheries and associated industries) may reach approximately 3–20% during 2020–50 with the >10 m fleet and associated industries bearing the majority of the losses.

Keywords Economic impact, employment, fisheries, ocean acidification and warming, revenue, social impact

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Introduction

Rising atmospheric CO_2 levels are responsible for both ocean warming (OW) and ocean acidification (OA) (Doney *et al.* 2009; Turley 2013). Most model-based climate change impact studies have focused on the effects of temperature change and conclude that OW will affect the distribution and productivity of ecosystems and the resources we extract from them (Cheung *et al.* 2011; Barange *et al.* 2014; Peck *et al.* 2016). OW is consequently expected to have consequences for dependent societies and industries at global and national levels (Allison *et al.* 2009; Lam *et al.* 2012; Merino *et al.* 2012; Barange *et al.* 2014; Jones *et al.* 2014; Groeneveld *et al.* 2016). There is much less understanding of the socioeconomic impacts of OW on communities that are economically reliant on fisheries resources and have low potential for economic diversification. In 2014, the UK had 6383 fishing vessels, of which approximately 80% were <10 m. The majority, found in England, have specialized in low-volume but high-value inshore fisheries such as sole and plaice. A larger portion of the >10-m vessels are found in Scottish waters and capitalize on higher volume, lower value fisheries such as herring and mackerel (MMO 2014). The total number of fishermen directly employed in the UK was 11 845, a decrease of 12% since 2004. The majority, 52%, are based in England & Wales, 40% in Scotland and the remainder in Northern Ireland (MMO 2014). The total value of landings by UK vessels into the UK equated to £615 million with Scotland showing the highest total landings (£402 million) followed England (£167 million) (MMO 2014). The highest value species for England, Wales and Northern Ireland were shellfish, whereas in Scotland it was demersal species. The survival of many fishing-dependent communities is closely linked to the survival of the inshore, smaller (<10 m) vessel fleet, due to its high contribution to their economic prosperity and social and cultural well-being (Symes and Phillipson 2009; Mathis *et al.* 2014).

Ocean acidification (OA), the disturbance of ocean carbonate chemistry from increased CO_2 uptake, with an associated pH decrease, can interfere with the life history of a number of marine species and exacerbate the impacts of OW alone (Hendriks *et al.* 2010; Kroeker *et al.* 2010, 2013; Nagelkerken and Connell 2015). However, our ability to fully predict the biological and socioeconomic effects of rising CO_2 emissions on marine ecosystems is limited (Doney *et al.* 2009; Turley 2013; Hilmi *et al.* 2014; Mathis *et al.* 2014; Voss *et al.* 2015). One of the fundamental problems in the assessment of combined OA and OW impacts (OAW) is the frequent scale mismatch between experimental work and the broadscale projection models needed to estimate large-scale effects (Queirós *et al.* 2015). While models may be adequate for exploring the impacts of a range of changes under particular scenarios (Andonegi *et al.* 2011; Cheung *et al.* 2011; Barange *et al.* 2014; Fernandes *et al.* 2016; Mullon *et al.* 2016), they can also be biased towards historic baselines because parameterization is based on limited empirical data (Fernandes *et al.* 2010, 2015; Rutterford *et al.* 2015), thus missing extreme events and failing to foresee new directions of change (Barnsley 2007; Payne *et al.* 2016). Moreover, the diversity of economic drivers of change, such as changes in costs, technology, trade, substitute goods and demand, can make forecasting problematic (Merino *et al.* 2012, 2014; Mullon *et al.* 2016). Scenarios are useful tools to reduce and synthesize this complexity, enabling stakeholder discussion and management decisions (Pinnegar *et al.* 2006; Mullon *et al.* 2016). It has also been recognized that assessments of socioeconomic impacts of changes to fisheries need to be extended from single to multiple sectors to provide a more holistic view (Isard 1972; Polenske 2004; Kagawa 2011; Kaplan and Leonard 2012). While current research is particularly focused on adequately parameterizing models to make use of the most recent advances in experimental work on the impacts of OAW (Queirós *et al.* 2015), translating these to their associated social and economic consequences is still lacking. The extremely limited understanding of species capacity for further adaptation to ongoing environmental changes (Pedersen *et al.* 2014; Shama *et al.* 2014; Rodríguez-Romero *et al.* 2015; Thor and Dupont 2015) and the lack of understanding of stakeholders adaptive behaviours remain as significant hurdles to the development of models that can project the socioeconomic impacts of OAW (Reed *et al.* 2011; Morgan 2015; Rivera *et al.* 2015).

Here, we use a habitat suitability and mechanistic niche model, SS-DBEM, which considers habitat suitability, species ecophysiology, population dynamics and metabolic theory (Cheung *et al.* 2011; Fernandes *et al.* 2013) to explore the combined effects of OAW on five commercially important demersal fish and mollusc species (Atlantic cod, *Gadus morhua;* European sea bass, *Dicentrarchus labrax*; great Atlantic scallop, *Pecten maximus*; blue mussel, *Mytilus edulis*; and common edible cockle, *Cerastoderma edule*). Additional analysis looks at the potential of pelagic species to fill the gap left by negative impacts on the above species (Atlantic herring, *Clupea harengus*; Atlantic mackerel, *Scomber scombrus*; blue whiting, *Micromesistius poutassou*; and European sprat, *Sprattus sprattus*). Drawing on new empirical and experimental data, the models are coupled with an economic input–output analysis based on published multipliers (Seafish 2007) to estimate the socioeconomic impacts of potential changes in the distribution of fished species due to OAW on the four countries of the UK (England, Scotland, Wales and Northern Ireland). Combining theoretical knowledge, experiments and models, the main objective of this study is to develop an approach that is consistent in considering the biogeochemical, biological, ecological and economic impacts of OAW. This approach includes the following: (i) biogeochemical models that provide projections of the physical–chemical environment and primary production processes driven by specific climate change emission scenarios; (ii) experimental work that investigates the impacts of different OAW scenarios on the life-history traits of key commercial species; (iii) application of a and b to force and parameterize, respectively, the habitat suitability and mechanistic niche model components of the SS-DBEM, incorporating descriptions of changes in life-history traits in response to environmental conditions; and (iv) input–output economic multipliers to estimate the social and economic implications of changes in fisheries and associated industries quantified as number of jobs and revenue at national and subnational levels.

The feedbacks between the ecological and the socioeconomic system are not dynamically modelled, but are fixed through the scenarios, given the associated uncertainties. This is a limitation, but to include dynamic feedbacks would have not only increased complexity but would have not recognized uncertainties in these feedbacks over coming decades. The aim of this work is to build exploratory scenarios of change that describe or visualize possible alternative futures (Borjeson *et al.* 2006; Bishop *et al.* 2007). The usefulness of exploratory scenarios in supporting environmental resource decision-making/management is becoming increasingly recognized internationally (e.g. Millennium Ecosystem Assessment 2011) as well as nationally (e.g. Pinnegar *et al.* 2006; Fernandes *et al.* 2016). In this case, the scenarios consider both the ecological and socioeconomic systems, and the dependency of the socioeconomics upon the ecology.

Methods and materials

Climate change emissions and biogeochemical model projections

Biogeochemical projections from two coupled hydrodynamic-biogeochemical models were used to provide changes in environmental conditions to drive several biological processes (life history, habitat, population dynamics and dispersal) in the habitat suitability and mechanistic niche model. The outputs provided were as follows: surface and near-bottom seawater temperature, salinity, oxygen, pH, primary production and water currents (horizontal and vertical). The biogeochemical simulations were forced using three emission scenarios to model the range of possible futures and to estimate associated environmental conditions. Two IPCC 5th AR scenarios (Rogelj *et al.* 2012), representing the lowest (RCP2.6) and highest (RCP8.5) emission pathways (Moss *et al.* 2010), were used to force the NEMO-MEDUSA 2.0 hydrodynamic-biogeochemical model (Yool *et al.* 2013). The IPCC AR4 A1B ('Business as usual') was also used as an intermediate scenario, to force the hydrodynamic-biogeochemical model POLCOMS-ERSEM (Artioli *et al.* 2014; Butenschön *et al.* 2015). This model provides higher spatial resolution projections and implements a parameterization that simulates the impact of OA on primary productivity: this has been simulated by increasing the carbon uptake by primary producers linearly with atmospheric pCO2, with a consequent unbalance of the C:N and C:P in the phytoplankton (Artioli *et al.* 2014). POLCOMS-ERSEM was run with and without this parameterization to assess the impact of this specific OA feedback on primary production and in turn how this impacts on our species projections. The two biogeochemical models have a different level of complexity in the description of the plankton community; however, the impact of this difference is minimal because the data coming from both models have been aggregated at the same level of complexity (total net primary production) before being used in the high trophic level model. Furthermore, both models have been successfully validated against observed data (Holt *et al.* 2012; Yool *et al.* 2013) demonstrating their ability to reproduce observed patterns despite the different complexity resolution.

Updating life-history parameters using OAW experiments

Data about species responses to OAW generated from experiments undertaken within The Natural Environment Research Council (NERC) UK Ocean Acidification Research Programme (Godbold and Solan 2013; Pope *et al.* 2014) as well as other recently published literature (Melzner *et al.* 2009a; Hendriks *et al.* 2010; Thomsen *et al.* 2010; Frommel *et al.* 2012; Schalkhausser *et al.* 2013) were used to update the habitat suitability and mechanistic niche model parameters (growth, length/weight relationship, larvae as well as adult mortality and dispersal). We limited the use of short-term experiments that often quantify responses to shock (Form and Riebesell 2012) by only considering results from experiments where exposure to OAW lasted 1 month or longer.

Experimental results were available for only a limited number of parameters for fish species, with more information available for bivalves (Table 2):

The parameters for fish (mainly cod and sea bass) for which experimental information was available include the following: larval mortality (LM) from experiments with sea bass (Pope *et al.* 2014); LM estimated from larvae tissue damage (Frommel *et al.* 2012); and length/weight relationship (LW) (Melzner *et al.* 2009b) for cod.

A wider range of parameters from the experimental literature were available for three shellfish species (scallop, mussel and cockle): Adult movement rate (AMR) and adult natural mortality (NM) (Schalkhausser *et al.* 2013), LW and growth (G) (unpublished data, P. Calosi, University of Plymouth) for scallop; for mussel, only NM was found (Berge *et al.* 2006; Hendriks *et al.* 2010); and for the cockle, we found bivalve generic parameters for LW and G (Hendriks *et al.* 2010), and NM from a soft sediment experiment (unpublished data, J.A. Godbold and M. Solan, University of Southampton).

We excluded pelagic fish species from the OAW reparameterization as only one published paper on Atlantic herring OAW experiments was found during the model development stage (Franke and Clemmesen 2011). Further model runs were performed for the three main pelagic species in UK (Atlantic herring, *Clupea harengus*; Atlantic mackerel, *Scomber scombrus*; and blue whiting, *Micromesistius poutassou*) to check the potential of those species to replace demersal and shellfish species that may be negatively affected by OAW. Additionally, European sprat (*Sprattus sprattus*) was considered, despite being a marginal catch in UK, to identify whether it could be a plausible replacement for other pelagic species.

Habitat suitability and mechanistic model for species catches

A habitat suitability and mechanistic niche model, an adaptation of the SS-DBEM species distribution model (Fernandes *et al.* 2013), was used for five commercial fish and bivalve species (Atlantic cod, European sea bass, Atlantic scallop, blue mussel, common cockle). The model used projected environmental outputs from the coupled hydrodynamic-biogeochemical models, past species distribution, the characteristics of habitats, ecology and physiology to parameterize the processes that drive individual species responses to change (Cheung *et al.* 2011; Jones *et al.* 2013). These processes include individual growth rates, larval mortality and recruitment, and dispersal of larvae and adults (Cheung *et al.* 2008, 2011). The model further considers species interactions according to metabolic theory to ensure the carrying capacity of individual geographical cells is not exceeded (Fernandes *et al.* 2013). Metabolic theory allows the estimation of the proportion of fish at each size that a system can support given a level of primary production and environmental conditions (Jennings *et al.* 2008). Here, we allowed the model parameters (growth, length/weight relationship, larvae as well as adult mortality and dispersal) to change dynamically with the temperature and/or pH values projected by the biogeochemical model scenarios (Queirós *et al.* 2015), according to available experimental data and literature.

The model uses, by default, estimated life-history parameters (Table 1) that have been compiled internationally by the Sea Around Us project (www.seaaroundus.org), which are publically available from the FishBase database (www.fishbase.org). The model uses the average of the reported values for a specific species; these are considered to be robust, particularly for commercial species. Available experimental data and literature records that considered multiple levels of temperature and pH change were used to reparameterize the model when appropriate (Berge *et al.* 2006; Melzner *et al.* 2009b; Hendriks *et al.* 2010; Franke and Clemmesen 2011; Frommel *et al.* 2012; Schalkhausser *et al.* 2013; Pope *et al.* 2014). The model equations (Cheung *et al.* 2011; Fernandes *et al.* 2013) were not modified to consider new interactions between OA and OW due to the scarcity of experimental data (e.g. Melzner *et al.* 2009b; Queirós *et al.* 2015). Parameters in the model were only changed to reflect the available experimental data of species responses when the biogeochemical model projection forecast those levels of temperature and pH used in the experiments. This means that the model's equations received different parameters depending on the pH and temperature (Queirós *et al.* 2015; Table 1 and 2). If the values of the experiments were not reached, then the default values from FishBase were used. Therefore, if an experiment used pH and /or temperature extremes that exceeded the biogeochemical model projections, then the reparameterization was not activated and no additional impact included in the results.

Model outputs consisted of yearly changes in the distribution and biomass of each of the five species with a horizontal spatial resolution of $0.5° \times 0.5°$. These were used to investigate both distribution and potential catches averaged by decade (to account for natural variability), between the 'present' (1991–2000) and three 'future' periods (2011–20, 2041–50 and 2090–99). Potential catches for each species were estimated using fishing mortality at maximum sustainable yield (F_{MSY}) levels in all the scenarios and timelines. This

Table 1 Equations and default parameters for habitat suitability and mechanistic model for species catches. The growth parameters are also used for estimating the natural adult mortality based on an empirical equation (Pauly 1980; Cheung *et al.* 2008). Larval mortality depends on temperature and currents using an empirical equation (O'Connor *et al.* 2007) and a survival parameter that was set up to 0.15 in a sensitivity analysis (Cheung *et al.* 2008). For further details on the equations, see Cheung *et al.* (2008).

Equation	Param.	Cod	Sea bass	Scallop	Mussel	Cockle
Growth: $L_t = L_\infty * (1 - e^{-k(t-t_0)})$	L_∞	200	103	12.4	8.1	10
	k	0.18	0.16	0.6	0.4	0.1
	t_0	−0.68	−0.82	0	0	0
Length-Weight: $W = a * L^b$	a	0.01	0.008	24.02	0.0002	0.023
	b	3	3.08	3.03	2.85	2.63
Adult mobility	cm h^{-1}	200	140.2	10	10	10

Table 2 Parameters derived from experiments that replaced default model parameters of Table 1 when environmental conditions projected by the biogeochemical models corresponded to experimental conditions below, as described in the methods. Most experiments consider several CO_2 concentration treatments at ambient temperature, referred to here as the 'ambient' treatments. A common set up is 380, 750 and 1000 ppm CO_2 concentration treatments at ambient temperature. Then, some of those CO_2 concentrations are considered together with a temperature variation of plus 2 or 4 °C, hereafter the 'warm' treatments). Some experiments from the literature use higher than 1000 ppm concentration and temperatures; however, for the model parameterization, we use the observed pH levels in water in each treatment. Therefore, for the model, 5 combinations of reparameterization are often considered. AMR, NM and LM are adult mobility rate cm h^{-1}, adult natural mortality and larval mortality, respectively.

Species	Treatment	pH	L_∞	k	t_0	a	b	AMR	NM	LM
Cod	Ambient	8.08	–	–	–	0.0104	3	–	1	–
	Ambient	7.45	–	–	–	0.0104	3.023	–	1.9259	–
	Ambient	7.08	–	–	–	0.0104	3.162	–	2.296	–
	Warm	8.08	–	–	–	0.0104	3	–	1	–
	Warm	7.45	–	–	–	0.0104	3.023	–	1.9259	–
Sea bass	Ambient	8.03	–	–	–	–	–	–	–	1
	Ambient	7.82	–	–	–	–	–	–	–	0.4775
	Warm	8.028	–	–	–	–	–	–	–	0.3108
	Warm	7.82	–	–	–	–	–	–	–	0.2609
Scallop	Ambient	8.16	12.4	0.6	–	0.0039	3.724	1	1	–
	Ambient	8.08	12.22	0.591	–	0.0153	3.537	1	1	–
	Ambient	7.6	12.03	0.582	–	0.0115	3.349	0.5	1.05	–
	Warm	8.16	12.4	0.6	–	0.0039	3.724	0.7	1.55	–
	Warm	8.08	12.22	0.591	–	0.0153	3.537	0.43	1.55	–
Mussel	Ambient	8.1	–	–	–	–	–	–	1	–
	Ambient	7.5	–	–	–	–	–	–	1.315	–
	Ambient	6.9	–	–	–	–	–	–	1.275	–
	Warm	8.1	–	–	–	–	–	–	1	–
	Warm	7.5	–	–	–	–	–	–	1.315	–
Cockle	Ambient	8.1	10	0.1		0.023	2.632	–	1	–
	Ambient	7.65	10.2	0.102		0.023	1.895	–	1	–
	Ambient	7.38	9.8	0.098		0.023	1.737	–	1	–
	Warm	8.1	10	0.1		0.023	2.632	–	1.063	–
	Warm	7.65	10.05	0.1		0.023	1.895	–	1.188	–

mortality rate was calculated using projected biomass of the species and their intrinsic population growth rate (Pauly 1980; Cheung *et al.* 2008; Fernandes *et al.* 2013). The F_{MSY} values were 0.2928, 0.5547, 0.1335, 0.2441 and 0.2446 for Atlantic cod, European sea bass, Atlantic scallop, blue mussel and common cockle, respectively.

Estimation of the economic implications of projected change in fisheries catches

The species simulations were used to investigate associated potential socioeconomic impacts of OA and OW in the UK without considering industrial adaptation and mitigation measures, which could potentially amplify or minimize impacts. Detailed data on catch by species, region and fleet (<10 m and >10 m) (Dixon *et al.* 2015) were used to calculate the estimated impacts of OAW on catches by area (England, Northern Ireland, Scotland and Wales) and by fishing fleet segments. Vessels <10 m fish close to landing sites and their impact was consequently restricted to the 0.5° × 0.5° cell in which their home ports were located, and their catch averaged by country for simplicity. There was no explicit competition between both fleets. Smaller vessels include only catch potential changes in coastal cells (up to 50 km from the fishing ports), whereas the larger vessels include catch potential changes in the ICES area where each country can fish. Fleet dynamics, where larger vessels could take more fish from coastal areas reducing the amount available for smaller vessels, were not modelled. The projected change in catch for the two demersal and for the three shellfish species were averaged, and used as representative proxies for the whole demersal and shellfish groups, respectively. This allowed consideration of the average potential impacts for the species group, the outputs of which facilitated the economic analyses. The five modelled species account for 25–50% of the demersal and shellfish species landings by weight in the UK, with demersal and shellfish species representing 60–65% of the total value of UK landings (Dixon *et al.* 2015). Projections of the three main pelagic species (herring, mackerel and blue whiting), which account for 90% of the pelagic catches in UK, and sprat as a potential replacement pelagic species, were also considered.

Economy-wide impacts of catch change were estimated using published input–output multipliers from the Seafish 2007 report. The Seafish report provides a variety of multipliers including backward and forward multipliers of Type I and Type II. Backward multipliers estimate the impacts of changes in the demand of an industry's output (e.g. fisheries) on the industry itself and all associated supplying industries (e.g. boat making, fuel refineries). Forward multipliers estimate the impact of a change in the supply of an industry's output (e.g. fisheries) on the industries it supplies (e.g. restaurants, supermarkets) (Leontief 1966, 1986; Miller and Blair 2009; ONS 2014). Both backward and forward multipliers are available for revenue, employment, income and GDP. There is also a distinction between Type I and II backward and forward multipliers. Type I multipliers capture the direct (industry specific) and indirect (associated industries) changes in the demand or supply of an industry's goods. Type II multipliers, in addition to direct and indirect impacts, also capture induced impacts. Induced impacts include changes in other economic sectors (e.g. utilities, clothing, transport) due to changes in expenditure by households containing individuals employed directly and indirectly by the fishing industry. Type II multipliers therefore provide a more holistic assessment of what changes in a single industry mean for the remaining economy. This paper uses the Type II backward multipliers to estimate the economy-wide upstream supply-chain impacts of changes in fish demand (through supply). Even though alternative economic assessments using Type I backward multipliers and Type I and Type II forward multipliers could have also been used, given that the aim of this paper is to provide a detailed analysis of an end-to-end impact assessment under climate change scenarios, the authors focussed only on the Type II backward multiplier.

Each output and employment multiplier used in the estimations for the four regional economies (England, Scotland, Wales and Northern Ireland) and for each group (demersal, shellfish and pelagic) are shown in Table 3. The derivation of these multipliers is described in detail in the Seafish (2007) report using accepted standardized approaches for regional input–output models (Miller and Blair 2009) populated by national statistics. The multipliers were used in this research to estimate the revenue and employment impacts of changes in the fish caught by vessel types and species using the results of the SS-DBEM model. For example, the SS-DBEM model estimated a 1% decrease in demersal fish caught by the <10 m vessel in England by 2020. This was assumed to equate to a 1% decrease in final demand satisfied, which was estimated to be a loss of £0.3 million direct impact based on total value of landings for the different fleets. The multiplier 3.2 is used to estimate total (direct, indirect, induced) economic impact resulting in an estimated £0.9 million loss for England by 2020.

Similarly, the employment multipliers were used to estimate the total employment impacts from changes in catch value. Total catch value, apportioned by fleet type, is equated to a total FTE based

Table 3 Revenue and employment multipliers by country, species, output and employment (Sourced from Seafish 2007). Note that Wales does not have pelagic fleet.

Country	Type of species	Revenue (output)	Employment
England	Demersal	3.21	2.06
	Shellfish	3.70	1.57
	Pelagic	3.24	4.12
Northern Ireland	Demersal	2.07	1.92
	Shellfish	2.35	1.58
	Pelagic	1.55	2.13
Scotland	Demersal	2.08	2.39
	Shellfish	2.30	1.49
	Pelagic	1.61	3.66
Wales	Demersal	1.97	1.26
	Shellfish	2.17	1.52
	Pelagic	0	0

on statistics from the Seafish report and the estimated unit values estimated (Seafish 2007). Following from the example above, a £0.3 million decrease in landings equates to a direct decrease of 8 FTE for the <10 m demersal fleet by 2020. This, multiplied by the employment multiplier 2.06 for <10 m demersal in England, results in an estimated total loss to the English economy of 16 FTE by 2020.

While input–output multipliers can prove useful in estimating economy-wide impacts, caution is required in their interpretation due to the underlying assumptions used to construct the input–output tables from which they are derived. IO tables are created using data describing national economies at a particular point in time and converted into a symmetrical matrix by assuming constant returns to scale, fixed coefficient production technology, constant coefficients of consumption and no supply constraints (Seafish 2007; Miller and Blair 2009; Sumaila and Hannesson 2010; Kaplan and Leonard 2012) . Using these multipliers to estimate impacts over time assumes these economies do not react to changes in technology or prices and that their structure remains the same. These assumptions are made to progress the research; however, future work could consider alternative future economic structures (Duchin 1998).

Results and discussion

Impacts on primary production

After temperature, primary production has been recognized as the main driver of long-term environmental change for fish and fisheries (Jennings *et al.* 2008; Cheung *et al.* 2011; Fernandes *et al.* 2013; Barange *et al.* 2014). The biogeochemical model used in this work projected an increase of between 0.5 and 3.3 °C in temperature and a 10–30% decrease in primary production in UK fishing waters by the end of the century. The inclusion of the effect of ocean acidification on primary production had a negligible impact on the overall fish- and shellfish-projected catches (<1% by the end of the century). Despite this, empirical (Riebesell *et al.* 2007; Bellerby *et al.* 2008; Kroeker *et al.* 2013) and modelling studies (Artioli *et al.* 2014; Kwiatkowski *et al.* 2014) have, however, demonstrated significant changes in lower trophic community composition and shifts in the seasonal dynamics of primary production as a consequence of OA. Changes in our annual mean primary production values are small because these are also dictated by nutrient availability, which is independent of OA.

Impacts on shellfish and fish species biomass

Biomass projections for the demersal fish species considered showed three significant patterns in response to OAW, consistent across the four countries (Fig. 1a, c, e, g). First, impacts of ocean acidification on sea bass biomass were limited compared with the impacts of warming in all the areas and scenarios. Second, while sea bass showed a positive response under lower pCO_2 emissions, this positive response was reversed at medium and high emissions. Third, cod biomass benefited from increasing temperatures in the model projections (Fig. 1a, c, e, g), but inclusion of OA exerted a negative impact across all the scenarios. Impacts on demersal species studied were driven mostly by effects on larval mortality and indications of tissue damage (used as a proxy for impact on natural mortality, see discussion below) observed in experiments under low pH levels and high temperature (Frommel *et al.* 2012; Pope *et al.* 2014). For sea bass, experimental impacts on larval mortality were positive (Table 2). On average, these did not translate into positive increases in biomass due to other mechanisms in the SS-DBEM model, and the projected occurrence of the positive OAW conditions in simulations from the biogeochemical models. Experiments on cod showed tissue damage under OAW conditions. To consider this negative impact, we assumed that natural mortality (larval, adult and predation) would be proportional to tissue damage. This decision is not without controversy, but it highlights one of the many challenges when incorporating experimental work into models. Most experiments do not yet provide all the parameters needed for models such as the SS-DBEM, even for commercial species for which more experiments have been undertaken (see Table 2). Most experiments focus on individual, specific traits (Maneja *et al.* 2013) of particular species, and lack the more systematic measurement and analysis of multiple traits required by models, such as those performed by Melzner *et al.* (2009b) or Queirós *et al.* (2015). In addition, experiments are often carried out with small sample sizes (Frommel *et al.* 2012; Maneja

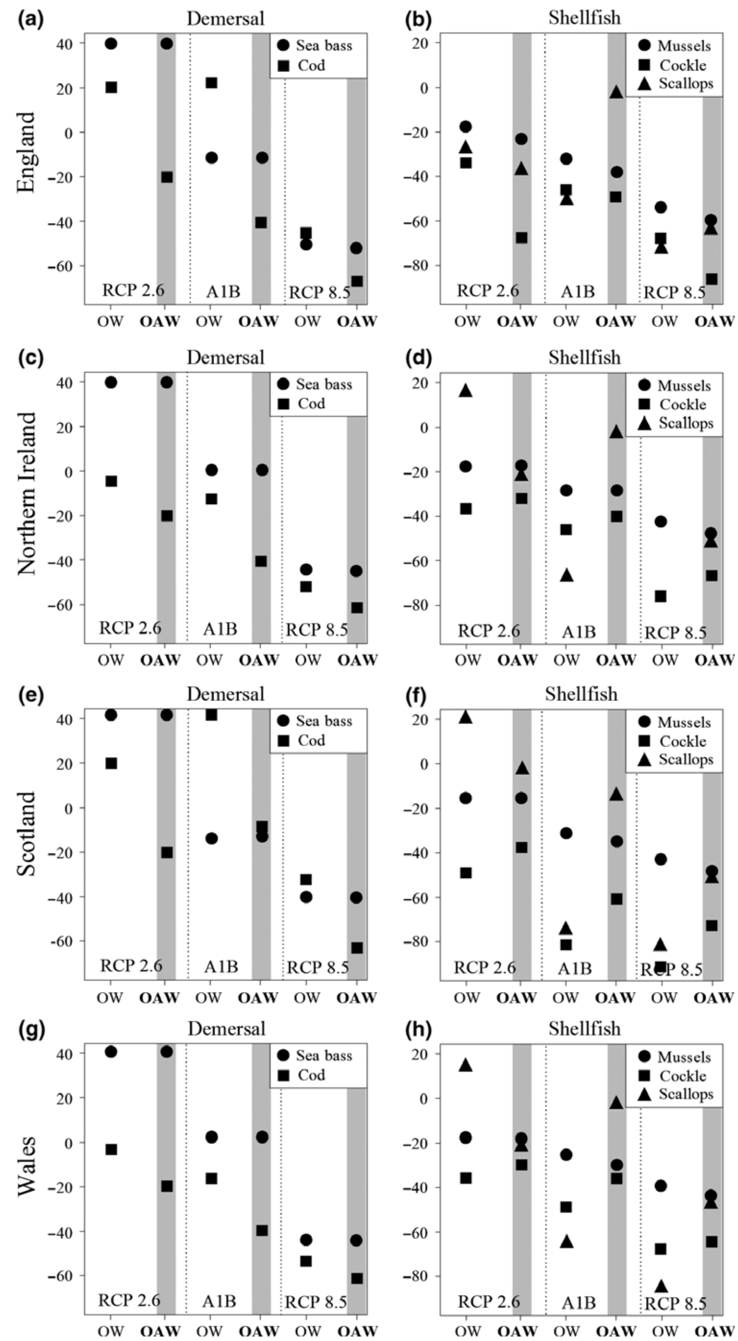

Figure 1 Percentage decadal changes in biomass for the five species shellfish and demersal species in 2090–99 vs. 1990–2000 in the relevant ICES subareas and country where catches take place considering the species are harvested at MSY levels. The scenarios are ordered from lower (RCP 2.6) to higher emissions (RCP 8.5) and with increased parameterization: OW, only warming impacts considered, OAW, considering both warming and acidification impacts. A1B refers to an intermediate climate scenario. Pelagic species are excluded due to lack of combined OAW experiments.

et al. 2013), which can lead to biases or low statistical power to detect impacts of the stressors analysed. However, recent studies are using this information to demonstrate the potential impacts of OAW (Cooley and Doney 2009; Cooley *et al.* 2015a,b; Díaz-Gil *et al.* 2015). As such, it was deemed appropriate here to use these other measured responses as proxies, provided they are transparent in their application. Overall, our results suggest that the real impacts of OAW, and other environmental changes (positive or negative) that could be exacerbated in the future, may have previously been underestimated in global and regional distribution models. However, bioeconomic models are starting to integrate this knowledge in their projections (Punt *et al.* 2014a,b; Cooley *et al.* 2015a,b; Seijo *et al.* 2016). Some of the data availability issues are derived from the difficulties and high cost of performing experiments, but the research community is starting to produce more appropriate outcomes to parameterize this type of models (Maneja *et al.* 2014, 2015; Pope *et al.* 2014; Queirós *et al.* 2015).

Under high-emission scenarios, the biomass decrease for the two demersal fish species exceeded 40% by the end of the century compared with the 'present' baseline (Fig. 1a, d, g, j). The bivalve species considered showed more complex and variable results (Fig. 1b, d, f, h). These were consistent across the four countries, except for scallop, which showed a more variable pattern across regions. There was a smaller biomass reduction for mussels under OAW than OW, but mussels were still linearly sensitive to increased emissions. Cockles appeared to be very sensitive to OAW at all emission levels, while scallops responded less negatively to OAW at medium- and high-emission levels. In general, these findings suggest that the combined biomass potential for the bivalves is expected to decline approximately 30 and 60% at low- and high-emission levels, respectively, by the end of the century.

The combination of multiple independent experiments led to broadly consistent response patterns across the five species. This is particularly the case for molluscs, for which there is a greater range of parameters available from OAW experiments, partially because these experiments are comparatively cheaper and less challenging to run than those on vertebrates. We take these results here as a proxy for expected responses of complete demersal fish and mollusc communities to OAW, acknowledging the limitations imposed on this study by experimental data sparsity.

Implication for UK fish catches

Some local positive impacts were observed in potential catch of pelagic and total fisheries for Scotland in the short term (2020) and, to a lesser extent, in the medium term (2050) particularly under a high-emission scenario. For England, the impact on potential pelagic and demersal catch is negligible and a similar outcome is found for Welsh demersal fisheries. Small positive impacts on demersal species catch last until the end of the century, but only under lower emissions. The high-emission scenario showed negative impacts on demersal fisheries of both fleets, with decreases in potential catches ranging from 15% (England) to 20% (Scotland and Wales) by the end of the century. Catches of demersal species in Northern Ireland showed little variation across time, scenarios and fleets, partly driven by the limited dependence of Northern Ireland fisheries on the species considered (Fig. 2). Shellfish catches (Fig. 3b, f, j, n) showed only negative impacts for all the four countries, with similar trends across countries and fleets in the short term and medium term. The exception was Northern Ireland, where the smaller vessel fleet (<10 m) would be more affected than the larger vessel fleet (>10 m). There were positive impacts on pelagic species catches until 2020 in

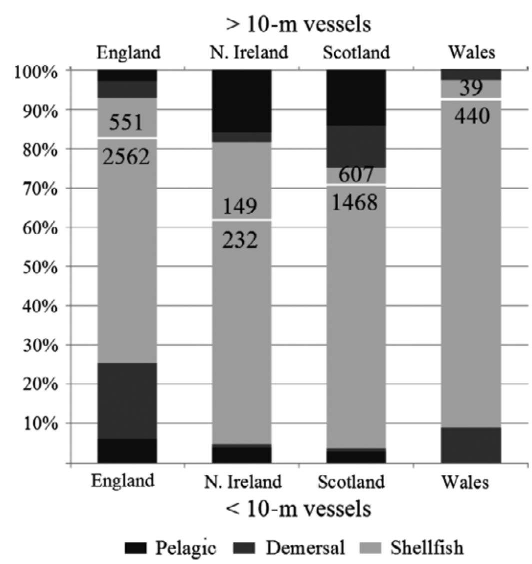

Figure 2 Structure of the fishing fleet by country as a proportion (%) of vessels by their length and target species. Lower bars (below the white spacer) represent the proportions for under 10 m vessels, whereas upper bars (above the white spacer) show proportions for vessels over 10 m length. The grey scale shows the proportion of catches for each species group represented within each of the two fleets. Numbers inside the graph show the absolute number of vessels in each fleet and country.

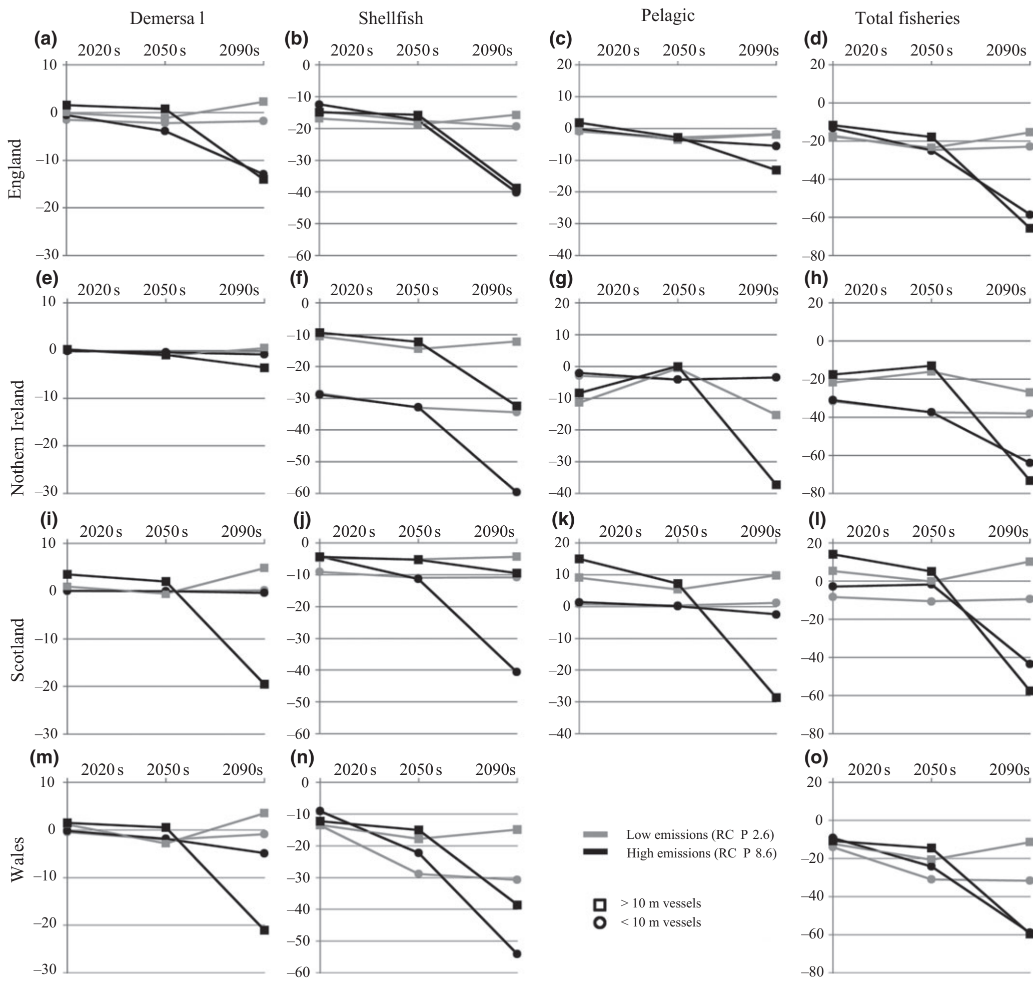

Figure 3 Percentage change in average catch potential during three 'future' decadal periods (2020s: 2011–20, 2050s: 2041–50 and 2090s: 2090–99) relative to the 'present' (1991–2000). The 'total fisheries' represents the effect on total catches that would occur with the projected changes in demersal and shellfish species. These results are presented for lower (RCP 2.6) and higher (RCP 8.5) emission scenarios using OAW reparameterization, and for both >10 m and <10 m fleet.

England and Scotland (Fig. 3c, k), but catches decreased thereafter. Scotland was the only country that showed increases in pelagic species catch potential under a higher emission scenario up to 2050, but this effect was reversed thereafter. Positive impacts were observed in Scotland under low-emission scenarios (Fig. 3k). Northern Ireland showed decreases in pelagic catches by 2020 which were followed by an increase by 2050, but with further decreases thereafter.

The patterns of catch potential for pelagic fish were largely driven by significant latitudinal distributional shifts in the model projections (Fig. 4), compared with demersal and shellfish patterns. Total catches of smaller vessels (<10 m) were the most impacted by 2020, with countries showing decreases of up to 20%, driven mainly by decreases in shellfish catch but also, to a lesser extent, decreases in catches of pelagic fish (Fig. 3). Longer-term impacts by 2100 relative to current projected yields under MSY showed that a lower emission scenario would retain decreases up to 30%, similar to shorter term projections, whereas a higher emission scenario could drive decreases of up to 60% in England, Scotland and Wales and potentially 80% in Northern Ireland (Fig. 3d, h, l, o). By the end of the century, the greatest changes in total catch were decreases projected in Northern Ireland as a result of their high dependency on shellfish (Fig. 3f), which are the impacted most by

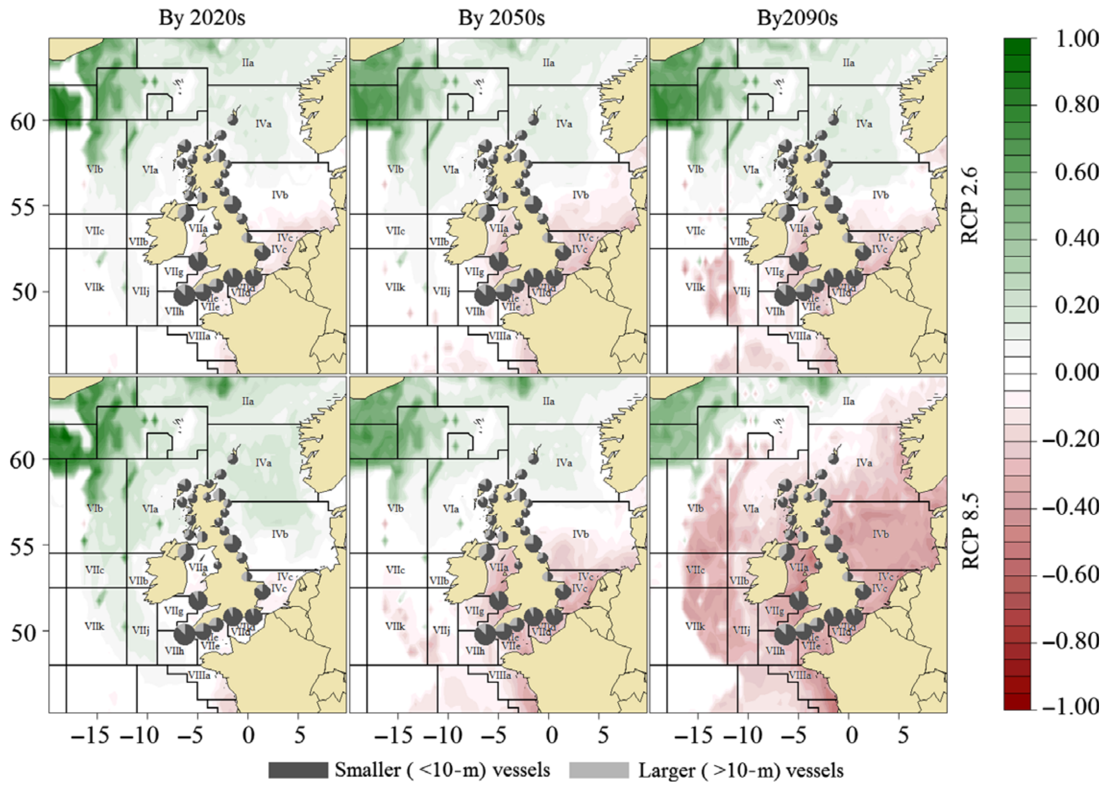

Figure 4 Areas of highest change (positive and negative) of average catch potential for pelagic species during three 'future' decadal periods (2020s: 2011–20, 2050s: 2041–50 and 2090s: 2090–99) relative to the 'present' (1991–2000). ICES areas where UK countries fish are labelled in the map. Pie charts show main landing ports showing proportion of smaller and larger vessels. The size of the chart is proportional to the number of vessels.

OAW (Fig. 1). Projected total catch in Scotland for large vessels under the low-emission scenario was the only case where long-term increases (Fig. 3i) were projected.

Positive impacts on demersal or pelagic species catch projected here are not sufficient to mitigate the negative impacts on shellfish total catches under the current national fisheries footprints. The modelled impacts of OW (alone) on pelagic species, or in combination with OA (OAW) on demersal fish and shellfish, are mostly negative and would exacerbate the OW impacts from 2050 (Fig. 3). Positive effects on catches of pelagic species were concentrated in Scotland due to a northward latitudinal shift of pelagic species (Fig. 3). However, it appears that the total catch of the small vessel fleet would not benefit sufficiently from those potential positive feedbacks in order to mitigate negative impacts in other species. This is because small vessel fleet targets mostly shellfish (MMO 2014; Fig. 2). Northern Ireland should also see benefits from pelagic species shifts in distribution, but only for a short period around 2050.

The above changes were mainly driven by a projected increase in temperature (0.5–3.3 °C), which exacerbated the impact of a decrease in primary production (10–30%) projected by the biogeochemical models in UK fishing waters. Further decreases in shellfish species could be driven by a projected decrease in pH (0.04–0.11 and 0.05–0.37 on average in high- and low-emission scenarios, respectively). The average changes in pH in the high-emission scenarios were in the range of variation considered in experiments (Table 2), but similar pH changes were observed in specific areas and years even in the lower emission scenario. These findings contrast with previous work based on modelling biogeochemical conditions and fish size-spectrum models, which projected the total potential biomass productivity of the sea to increase on average by 3.4% globally (Barange *et al.* 2014), although this study did not consider

OA impacts or the impacts of projected environmental change on aggregated physiological processes (Cooley and Doney 2009; Turley 2013).

Our results do not consider the combined impacts of OAW on pelagic fish species due to the absence of experimental data of sufficient quality. However, potential increases in pelagic species catches (MMO 2014; Fig. 2) are unlikely to compensate for the economic losses in the demersal and shellfish fisheries, whose combined value is twice that of pelagic species (Dixon *et al.* 2015). Despite price increases in previous years, pelagic fish prices dropped in 2013 (MMO, 2014) and fishing costs have increased (Watson *et al.* 2013). This will affect the profitability of the fleet, and even if prices go up in the future, they are unlikely to compensate for demersal and shellfish catch losses that are more targeted by smaller vessels fleet at the country level. In the future, higher prices for pelagic species, if any, are more likely to be driven by resource competition with the aquaculture industry and the price the consumer is willing to pay for wild fish over other sources of protein like cultured fish (Merino *et al.* 2012).

To a certain extent, the projected decreases in pelagic species could be explained simply by the combination of reduced primary production and an increase in temperature. However, this mechanism does not account for all of the observed differences. Latitudinal species shifts, driven by climate change (Cheung *et al.* 2011; Fernandes *et al.* 2013), also play a significant role, which particularly benefits Northern Ireland and Scotland (Fig. 3). It is likely other species would replace those that were displaced. Sprat projections in the Baltic, where this is a prevalent species, show a potential increase in catch under OW (Voss *et al.* 2011) that is consistent with projections using our fisheries model in the Baltic (Mullon *et al.* 2016). However, the same sprat trend was not observed in UK fishing areas. A latitudinal shift or population increase in southern species, such as sardine and anchovy, may be more likely (Petitgas *et al.* 2012). Alternatively, the available ecological space could be filled by invasive species, for example lionfish (Cure *et al.* 2014).

Finally, these negative impacts on modelled species have to be considered with judicious caution. In the current modelling work, we are unable, due to lack of sufficient data, to constrain the significance of further mechanisms, which could increase the resilience of species (in particular pelagic species) to OW and OAW. Such mechanisms include the following: (i) the existing levels of phenotypic variation found among different populations living along environmental/latitudinal gradients (Gaston *et al.* 2009; Bozinovic *et al.* 2011); (ii) the within- and trans-generational phenotypic plasticity shown by many pelagic and demersal fish and shellfish species (Jennings and Beverton 1991; Winters and Wheeler 1996; Engelhard and Heino 2004; Pedersen *et al.* 2014; Shama *et al.* 2014; Thor and Dupont 2015); (iii) the potential for metazoan adaptive responses (Kelly and Hofmann 2013, Munday *et al.* 2013) considering the large number of eggs some species can produce (and thus the potential population size and their potential for adaptation, see e.g. Sunday *et al.* 2014) and their general reproductive strategy (Melvin *et al.*, 2009); and (iv) the presence of potential geographical and ecological barriers affecting connectivity among populations and effectively preventing species from shifting their geographical ranges.

Economic implications

The estimated total net economic loss to the whole of the UK from the projected changes in demersal, shellfish and pelagic catch by 2050 (Table 4) amounts to £87 million (m) on average across all the emission scenarios. England bears the greatest loss in absolute terms with £ 65 m, followed by Scotland and Northern Ireland with losses of £8 m each and Wales with £6 m of losses (all in 2011 prices; Table 4). These losses represent only a minor proportion of total UK GDP (£1.37 trillion in 2013; ONS 2014). However, if taken as a percentage of the total output of the fishing industry and its associated sectors, the potential impact of these losses becomes apparent. Total revenue attributable to each region's fishing and associated sectors in 2011 was £551 m for England, £634 m for Scotland, £50 m for Northern Ireland and £33 m for Wales. The estimated losses as a proportion of total regional revenues ranged between 1 and 18%, dependent on region, by 2050. Northern Ireland and Wales see the greatest loss in relative terms across all their fleets and associated economic sectors, with decreases in revenue of 16 and 18%, respectively (Fig. 5) due to their higher dependence on shellfish species (Fig. 2).

Except for short-term positive impacts for Scotland and England, the model projections under

Table 4 Total economic impact (direct, indirect and induced) of OAW on fisheries activities by 2020s, 2050s and 2090s in terms of output revenue (£ m) and full-time equivalent (FTE) employment disaggregated between small and large fleets in the four nations and the two emission scenarios in relation to present (1991–2000).

				2020s		2050s		2090s	
Country				£ m	FTE	£ m	FTE	£ m	FTE
England	RCP2.6	<10 m	Demersal	−0.9	−17	−2.6	−49	−1.1	−21
			Shellfish	−14.3	−479	−17.2	−578	−19.0	−636
			Pelagic	−0.1	−1	−0.2	−2	−0.1	−1
			Total	−15.3	−497	−20.0	−628	−20.2	−658
		>10 m	Demersal	−0.2	−3	−2.0	−36	3.9	72
			Shellfish	−43.6	−1461	−48.5	−1624	−40.7	−1363
			Pelagic	−0.1	−1	−0.6	−7	−0.3	−4
			Total	−43.8	−1465	−51.0	−1668	−37.1	−1295
	RCP8.5	<10 m	Demersal	−0.4	−6	−2.4	−44	−7.9	−147
			Shellfish	−12.2	−409	−17.1	−573	−39.3	−1318
			Pelagic	0.0	0	−0.2	−3	−0.4	−4
			Total	−12.6	−415	−19.7	−620	−47.6	−1470
		>10 m	Demersal	2.6	49	1.2	23	−23.5	−437
			Shellfish	−38.6	−1292	−40.7	−1364	−100.0	−3351
			Pelagic	0.3	4	−0.4	−6	−2.1	−26
			Total	−35.7	−1240	−39.9	−1346	−125.5	−3814
Northern Ireland	RCP2.6	<10 m	Demersal	0.0	0	0.0	0	0.0	0
			Shellfish	−2.2	−53	−2.5	−62	−2.6	−64
			Pelagic	0.0	0	−0.0	0	0.0	0
			Total	−2.2	−53	−2.5	−62	−2.6	−64
		>10 m	Demersal	0.0	0	0.0	−1	0.0	0
			Shellfish	−4.5	−111	−6.2	−153	−5.2	−128
			Pelagic	−1.2	−13	0.0	−1	−1.6	−18
			Total	−5.7	−124	−6.2	−155	−6.8	−146
	RCP8.5	<10 m	Demersal	0.0	0	0.0	0	0.0	0
			Shellfish	−2.2	−54	−2.5	−61	−4.5	−111
			Pelagic	0.0	0	0.0	0	0.0	0
			Total	−2.2	−54	−2.5	−61	−4.5	−111
		>10 m	Demersal	0.0	0	0.0	−1	−0.1	−2
			Shellfish	−4.0	−99	−5.2	−128	−13.7	−341
			Pelagic	−0.9	−10	0.0	0	−9.7	−44
			Total	−4.9	−109	−5.2	−129	−4.0	−387
Scotland	RCP2.6	<10 m	Demersal	0.0	0	0.0	0	0.0	0
			Shellfish	−7.8	−197	−9.4	−238	−9.3	−234
			Pelagic	0.0	0	0.0	0	0.0	0
			Total	−7.8	−197	−9.4	−238	−9.3	−234
		>10 m	Demersal	2.6	33	−1.7	−21	13.9	175
			Shellfish	−12.5	−315	−13.3	−337	−11.4	−288
			Pelagic	16.9	115	10.9	69	18.2	124
			Total	7.1	−166	−5.0	−289	20.7	11
	RCP8.5	<10 m	Demersal	0.0	0	0.0	0	0.0	0
			Shellfish	−3.7	−92	−9.7	−244	−34.9	−880
			Pelagic	0.0	0	0.0	0	0.0	0
			Total	−3.7	−92	−9.7	−244	−34.9	−880
		>10 m	Demersal	10.0	127	5.9	75	−56.4	−712
			Shellfish	−11.2	−284	−11.2	−284	−24.9	−630
			Pelagic	27.9	190	13.6	92	−53.3	−363
			Total	26.7	33	8.2	117	−134.6	−1705

Table 4 Continued.

Country				2020s		2050s		2090s	
				£ m	FTE	£ m	FTE	£ m	FTE
Wales	RCP2.6	<10 m	Demersal	0.0	0	0.0	−1	0.0	0
			Shellfish	−1.6	−37	−3.4	−78	−3.6	−83
			Pelagic	0.0	0	0.0	0	0.0	0
			Total	−1.6	−37	−3.4	−79	−3.6	−83
		>10 m	Demersal	0.4	14	−1.0	−34	1.2	43
			Shellfish	−1.9	−45	−2.6	−59	−2.1	−49
			Pelagic	0.0	0	0.0	0	0.0	0
			Total	−1.5	−31	−3.6	−94	−0.9	−6
	RCP8.5	<10 m	Demersal	0.0	0	0.0	−1	−0.1	−2
			Shellfish	−3.7	−25	−2.6	−60	−6.3	−146
			Pelagic	0.0	0	0.0	0	0.0	0
			Total	−3.7	−25	−2.6	−61	−6.4	−148
		>10 m	Demersal	10.0	18	0.2	6	−7.4	−258
			Shellfish	−11.2	−41	−2.2	−50	−5.6	−129
			Pelagic	27.9	0	0.0	0	0.0	0
			Total	−26.7	−23	−2.0	−56	−12.9	−387

low- and high-emission scenarios show negative OAW impacts on catch for the main UK shellfish fisheries. When considering the impacts of OW on pelagic species and OAW on demersal species to 2020, positive effects on these species mitigate the losses in shellfish harvest, but only in Scotland for the >10 m fleet. For the >10 m sector total revenues would increase in Scotland by £26.7 m by 2020 equivalent to an associated overall increase of 33 full-time equivalent (FTE) jobs in the higher emission scenario (Table 4). There would be little mitigation effect for the <10 m fleet in any scenario, and no positive impact for any of the Scottish fleet under the lower emission scenario. Under the high-emission scenario, a small gain for the Scottish >10 m fleet could also observed to 2050 (with a total increase in revenue of £8.2 m and an increase in 117 FTEs) in relation to present. Positive impacts are projected for Scotland by 2100, but only under a lower emission scenario (+ £20.7 m revenue and + 11 FTE). In the high-emission scenario, impacts were negative (−£134.6 m revenue and −1705 FTE).

For England and Wales, some positive impacts on demersal and pelagic species catch can be seen by 2020 in the >10 m fleet under high-emission scenarios, but not enough to mitigate losses to the shellfish harvest. There would be no mitigation effect in the English <10 m fleet. No other positive impacts on pelagic species catch were observed by 2050 and 2100 for England. Changes in pelagic species catch have no impact on Welsh revenue and employment due to their low dependency on these species (Fig. 2). For Northern Ireland, changes in pelagic species catch exacerbate the impacts of declining shellfish catch, resulting in a decrease of 16% in revenue by 2050. By 2100, like in Scotland, the Northern Irish >10 m vessels would be more negatively affected than the <10 m vessels (Fig. 5). However, Northern Ireland's >10 m vessels would not see the positive gains likely to be experienced by the Scottish >10 m fleet. Northern Ireland would also see the largest relative losses in employment (13% by 2020 and 15% by 2100) within the fisheries industry and associated sectors. Overall, the absolute loss in revenue will impact the >10 m vessels more than <10 m vessels in England and Northern Ireland until 2050 (Fig. 5). This trend continues for Northern Ireland, but for England the <10 m would be more highly impacted post-2050. In Wales, there would be little distinction between vessels. In all countries, the >10 m are facing the greatest loss in employment along with their associated sectors across all scenarios and time periods.

Where the revenue or employment losses, or gains, enter into the economy differs with country

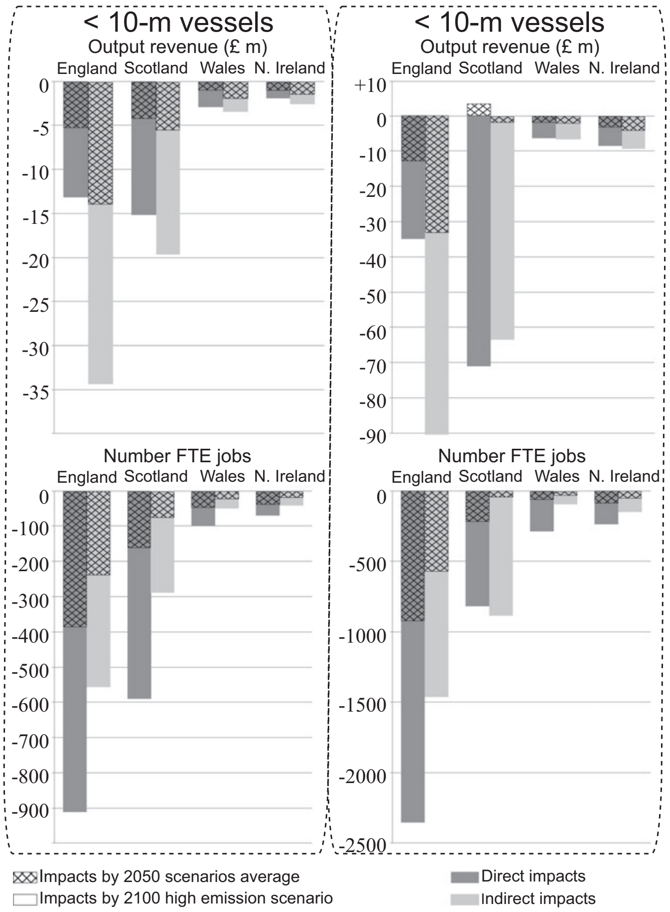

Figure 5 Economic impact of OAW on direct (dark) and indirect and induced (light) fisheries activities in 2050 and 2100, based on multipliers of output revenue and potential losses in full-time equivalent (FTE) employment disaggregated between small and large fleets in the four nations. Impacts of the average of all 2050 scenarios and of the 2100 low-emission scenario were similar, and therefore, only the former are shown, in addition to the impacts of high emissions by 2100.

due to the structure of their supply chain. As shown in Figure 5, revenue losses across time periods and emission scenarios would be highest as indirect and induced impacts. Losses will be greatest for the associated industries (e.g. boat building, fuel), as opposed to being directly lost by the fishing industry itself. This reflects the dependence of the fishing industry on goods and services from a range of industries each with varying operating and pricing structures and each adding different levels of value compared with the fishing industry. In contrast, employment could be more greatly affected directly in the fishing sector compared with associated upstream sectors. Of the total jobs lost by 2050 for the low-emission scenarios, England would lose 1446 FTE directly in the fisheries sector (63% of total estimated job losses), Scotland would lose 376 FTE and Wales 119 FTE (each representing approx. 70% of total estimated job losses), and Northern Ireland would lose 137 FTE (63% of total estimated job losses). These projections indicate that these losses will occur predominately among the <10 m vessels for Wales and Scotland and the >10 m vessels for England and Northern Ireland.

Impacts on society

Under the current economic structure for each of the four countries, losses to the fishing industry and wider economy would be likely for both the <10 m and >10 m fleets. The exception is Scotland's >10 m fleet where gains were projected for the period up to 2050 (high scenario) and 2100 (low scenario). The impacts on the small vessel fleets are expected to be higher than for the >10 m fleet in Wales and Scotland and to occur more quickly (Table 4). Smaller vessels rely on higher value demersal and shellfish species that are more sensitive to OAW in the short term. The <10 m fleet is also more numerous and supports a higher proportion of fisher employment (MMO, 2014; Fig. 2). In contrast, >10 m fleet would have higher relative and absolute losses in the long term. The economic impact of change to the >10 m fleet will be greater because it supports a higher level of employment in associated sectors compared with the smaller vessels fleet (Table 3), although not in terms of revenue. These findings suggest that management adaptations would need to be tailored to specific components of the fishing industry.

The sensitivity and adaptability of regions containing fishery-dependent households will affect the ability of such communities to withstand any changes that may occur as a consequence of OWA. Levels of deprivation have been used to assess the adaptive capacity of fishing communities. For example, Morrissey (2014) concluded that fishing and seafood processing are located in areas with higher than average deprivation in UK. This may make it harder for fishery-dependent household to adapt to change. Deprivation indices are, however, relative and thus offer only an indication of where to look to identify vulnerabilities. The level of fishery dependence in communities may also affect household and community adaptive capacity (Allison *et al.* 2009). The percentage of the working population employed in the fisheries sector is particularly low in UK port communities, except in Scotland, and lower than the global average of 5–10% (Lindkvist 2000; Symes 2000). This may suggest that the majority of UK fishing communities are robust to changes resulting from OWA. However, defining dependence so narrowly disguises a number of important factors. While the proportion of fishermen in some communities is small, the absolute number in some communities is sizable (e.g. 112 in Milford Haven, Wales; 87 in Portree, Scotland; 58 in Dartmouth, England and 57 in Looe, England). Losses to these numbers could destabilize parts of these communities, especially when upstream and downstream supply-chain effects are taken into consideration. The contribution of fisheries to local gross value added (GVA) may also be a relevant indicator of dependence. In the Shetland Islands, for example, the fisheries sector (including aquaculture and processing) accounts for over a third of the value of the total economy (SIC, 2008), making it potentially vulnerable to changing fish stocks.

Management measures in response to OAW need to consider the cascading socioeconomic and cultural consequences for local communities (Urquhart *et al.* 2011; Shaw *et al.* 2015). Moreover when latitudinal shift of species distributions such as mackerel have already raised disputes on catches allocations between Iceland, Norway and Scotland (Bazilchuk 2010; Cendrowicz 2010; Astthorsson *et al.* 2012;). Fisheries dependence cannot be limited to economic issues; it has a sociocultural dimension involving traditions, community bonds, shared values and knowledge (Brookfield *et al.* 2005), much more difficult to quantify in models. Indeed, some fisherman will resist diversification altogether due to economic and sociocultural constraints (Morgan 2015) limiting their autonomous adaptive behaviours. However, some would diversify to adapt, for example, targeting new species resulting from latitudinal shifts of species distributions. However, this may result in disputes over catch allocations, as occurred between Iceland, Norway and Scotland in the case of mackerel (Bazilchuk 2010; Cendrowicz 2010; Astthorsson *et al.* 2012). Another adaptive behaviour strategy observed in UK is that older fishers are not reinvesting in their boats before retirement due to the dearth of young people entering the industry (Reed *et al.* 2011). Understanding the impacts of catch, revenue and employment change on these wider dimensions, and how future generations of fishers may adapt, must be a central component of future studies into the socioeconomic impacts of OAW.

Implications for policy and management

One of the key challenges in climate change research is linking biophysical changes to quantified impacts on relevant social and economic factors (Barange *et al.* 2010). This work contributes to closing the gap between natural impacts and social consequences, identifying and quantifying who will be most negatively impacted, when, and why, in the socioecological system of UK fisheries. These findings contribute to the process of developing adaptive management options that can help prevent those changes, mitigate the impacts or support the adaptation of those most impacted or with limited opportunities for adaptation. By identifying how and what fish resources may respond to OWA (Russell *et al.*, 2015; Rodríguez-Romero *et al.* 2015), our analysis has also the potential to contribute to the design of adaptive monitoring strategies, and to assist the fishing industry in their own responses to changes in stock distribution and productivity.

Although this work indicates some short-term gains from OWA for specific components of the UK fisheries sector, the overall impact is largely negative for all fleets. This highlights the need for long-term prevention and mitigation approaches. The UK fishing industry has already suffered significant job losses over the last 35 years, from 22 134 fishers in 1976 to 12 405 in 2011, with associated year on year reduction in the number of vessels and effort (MMO, 2014). These reductions have been driven by new multi- and international agreements (e.g. the extension of the Exclusive Economic Zones, the establishment of the Common Fisheries Policy in Europe and ship decommissioning policies) and have allowed staged exits from the fishing industry (Kerby 2013). However, OWA impacts may take place without timely and proactive policy adjustments leading to vessel entry–exit to the fishery over time as a result of profits (Smith 1969). OWA effects on marine species, despite being pervasive (Nagelkerken and Connell 2015), will be more gradual and may be more difficult to detect by associated industries than previous reductions driven by overfishing. Volatile fuel prices, the precarious state of some stocks and excess capacity in the industry (Abernethy *et al.* 2010), combined with spatial competition with other industries, demand a level of proactive planning not currently observed to ensure the viability of the industry (Queirós *et al.* 2016). Future research into the changes in the economy, cost structures due to resource rents and break-even points for fishers (which dictate entry–exit strategies) should be undertaken to enable more tailored policy and management interventions.

Proactive planning should include adequate responses cognizant of the cascade of effects that may result from changes in the fishing sector (Metcalf *et al.* 2015). This may include modifications to existing fisheries management programmes with tighter controls on inputs and effort coupled with technical changes (as suggested by e.g. Rossiter and Stead 2003) and/or gradual exit schemes, which support fishermen in the transition to alternative livelihoods. Long-term planning and a consensus building approach to policy development is also required, in contrast to the short time frame and partisan approach that dominates decision-making (Giddens 2009; Trifonova *et al.* 2015; Queirós *et al.* 2016).

Needless to say, 'climate-proofing' fisheries policy and management does not remove the need to develop CO_2 and other greenhouse gas emission reduction strategies (Cooley *et al.* 2015a). The outcomes of the UNFCC COP 21 process may open the door to such strategies, which should be implemented in parallel to the sector-specific responses to OAW (Queirós *et al.* 2016).

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