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Spatial ecosystem modelling of marine renewable energy installations: Gauging the utility of Ecospace

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ABSTRACT

The deployment of offshore structures for renewable energy generation (wind/wave/tidal) will lead to the alteration of access to the area of installation for several users of the sea including: shipping, fishing, tourism and recreational users. Arguably, the largest impact will be upon the fishing industry where access loss may lead to displacement and reduced catch per unit effort in turn leading to conflict. To prevent conflict, it is important to understand mitigating factors. Marine renewable energy devices (MREDs) and associated infrastructure will be placed on the seabed, affecting benthic infauna and epifauna, important sources of food for many species including those of commercial importance, potentially providing benefits to the fishing industry and mitigating the causes of conflict. Two key plausible benefits of MREDs are the 'artificial reef effect' and the 'exclusion zone effect'. This study investigated the utility of the Ecopath with Ecosim and Ecospace modelling software to address the implications of these 'effects'. Two case study models were developed, one at the whole west coast of Scotland shelf scale and one at a smaller single installation scale. Our results suggested that the Ecospace model could potentially predict the effects of MRED installations, but revealed that there are a number of considerations which should be taken into account before attempting to do this. Key considerations include data availability (an issue in all modelling), spatial scale and resolution. Other limitations to this particular study such as the ability to make changes over time are currently being addressed by ongoing developments of the software. Despite the considerations and limitations, these case studies reveal the usefulness of spatial ecosystem modelling, particularly Ecospace, to investigate this issue.

1. Introduction

As the anthropogenic drivers of climate change become more apparent, the use of renewable energy to move towards a global low carbon economy is gathering momentum. In the past decade, the focus has been on technologies such as solar photo-voltaic and biomass technologies, ocean thermal energy conversion, wind, wave and tidal (Gross et al., 2003). Indeed, energy extraction from the marine environment is currently an area of growth owing to a vast potential source of energy resources. This is due to constantly developing technology (Pelc and Fujita, 2002) and suggestions that moving renewable energy generation offshore reduces issues involved with siting onshore such as visual impact (Gill, 2005; Ladenburg, 2008), planning control and regulation, and limited available onshore sites (Haggett, 2008). It is predicted that 7% of the world's electricity production will come from the ocean by 2050 (Esteban and Leary, 2012). Therefore, it is important that environmental, social and economic impacts of marine renewable energy device (MRED) installations are identified and measured to ensure that decisions regarding offshore energy are sustainable and equitable.

There are several potential negative impacts of MRED deployment. Birds, mammals and fish may collide with MREDs; human-induced noise and electromagnetic fields may affect some marine species; and MREDs may constitute suitable habitats for non-indigenous species, thus facilitating their spread (Gill, 2005). Furthermore, the placement of devices and their associated infrastructure on the sea floor may lead to the displacement of organisms in the local area as well as modifying habitat resulting in changes to local food-web dynamics. It will also lead to changes in access to the area of installation for users of the sea including shipping, fishing, tourism and recreation (Jay, 2010; Nobre

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et al., 2009; Punt et al., 2009). Arguably, the largest impact will be upon the fishing industry where access loss may lead to displacement and a reduction in catch per unit effort (Alexander et al., 2013). Less understood are any potential positive effects of MREDs upon the ecosystem. Fishers have suggested that artificial reefs and exclusion zones may provide potential mitigation for loss of access (Alexander et al., 2013) and it is this concept that we address here.

MREDs and associated infrastructure placed on the seabed will likely act as de-facto 'artificial reefs'. This has received some attention with a focus upon offshore wind farms (Petersen and Malm, 2006; Wilhelmsson et al., 2006; Wilson and Elliott, 2009) and wave power devices (Langhamer and Wilhelmsson, 2007; Langhamer et al., 2009). Although there has not yet been any research into tidal devices and their potential to act as artificial reefs (ARs), the similarities with wind and wave devices in seabed moorings and associated infra-structure means that this will likely occur. Diver observations, fish surveys, photography, habitat plates and settlement panels, has led to proposals that ARs increase local biodiversity, species abundance and biomass, especially of mobile species by providing additional habitat and refuge (Beaumont, 2006; Hueckel et al., 1989; Martin et al., 2005). However, some studies found that diversity and abundance were lower on an AR than control sites (Davis et al., 1982; Fabi et al., 2002), and that some species are depressed while others are not (Barros et al., 2001; Fukunaga and Bailey-Brock, 2008). In a study of ARs installed to enhance artisanal fisheries in Portugal, ARs were suggested to act as recruitment areas and an extension of natural mating/spawning grounds (Leitao et al., 2009). If ARs do increase species abundance, this could be beneficial for commercial fishers by increasing the catch potential in the local area.

Exclusion zones (EZs) are likely to be placed around MRED installations, as has happened at the Cornwall Wave Hub (Campbell et al., 2014), closing the areas to certain fishing gear types or creating no-take zones. Limited research has focused upon the potential EZ effects of offshore development (Shields and Payne, 2014 and references therein) however, it is possible to draw parallels with marine protected areas (MPAs). MPAs were developed primarily as a conservation tool due to the decline of fish stocks and deterioration of habitats worldwide; they are also suggested to be a fishing management tool used to control the spatial distribution of fishing pressure (Halpern, 2003; Hilborn et al., 2004). Some argue that when fishing is stopped, species become more abundant and diverse, as well as larger and more fecund, and that the protection of spawning stock biomass can increase recruitment and re-stock fished areas (Roberts and Polunin, 1993). In addition, connectivity (a consequence of which can be the exchange of populations through larval dispersal) can enhance fish production outside of MPAs leading to a 'spill-over effect' and enhanced catches in adjacent areas (Sale et al., 2005). Roberts et al. (2001) found that over five years, St Lucian marine reserves led to an improvement in neighbouring fisheries catches despite a 35% decrease in fishing ground area. Spill-over from EZs was a significant although variable factor in the dynamics of the fishery in Mombasa Marine Park, Kenya, although spill-over also interacted with fisheries gear, morphology and tidal patterns (McClanahan and Mangi, 2000). In another study, approximately 7% of spiny lobster emigrated annually from the Mediterranean Columbretes Islands Marine Reserve to an adjacent fishery (Goni et al., 2010). If EZs enhance catches in adjacent areas, this will also be of benefit to local fishers.

Empirical exploration of the potential benefits of MREDs caused by the AR and EZ effects is prohibitively expensive. Alternatively, computational models can be used to represent a marine ecosystem and provide indications of how the ecosystem is likely to change in response to these effects as well as how the fishing industry will subsequently be affected. Most ecosystems are complex and creating a suitable model is challenging. Should a credible model be developed, model parameters can then be changed to explore a range of scenarios that can be tested empirically for verification. There are several ecosystem models in use (e.g. Baretta et al., 1995; Fulton et al., 2004; Shin and Cury, 2001), however the most used and tested ecosystem modelling tool for investigating how ecosystems respond to changes in fishing (and other pressures) is Ecopath with Ecosim (EwE) (Christensen, 2009). EwE is a dynamic food-web modelling suite which describes ecosystem resources and their interactions (Christensen and Walters, 2004).

To investigate AR and EZ effects, spatial models are required. Ecospace, a spatial modelling algorithm for EwE, was developed to investigate the effects of marine protected areas (e.g. Beattie et al., 2002; Chen et al., 2009; Salomon et al., 2002). Colléter et al. (2014) used Ecospace to investigate the potential spillover effect from MPAs, showing that potential exports from small scale MPAs are limited and thus may only benefit local fishing activities. Ecospace has also been used to investigate the effects of ARs on marine ecosystems and fisheries: Pitcher et al. (2002) found that small protected areas with human-made reefs would achieve little to avert a collapse of the fisheries in the area or a shift towards lower value species, however larger protected areas may do much to restore valuable fisheries in the South China Sea. This would suggest that Ecospace is an appropriate tool to investigate the EZ and AR effects of MREDs.

Our aim was to guage the utility of Ecospace to address the question of whether MREDs can benefit, and thus mitigate a potential loss of access for the fishing industry by providing: (a) habitat through the 'reef-effect' and (b) protection through the 'exclusion zone effect'. To do this we developed two model 'case studies' representing the west coast of Scotland, an area of key interest in the UK for offshore renewable energy extraction-building upon the Ecopath with Ecosim (EwE) model of the west coast of Scotland (Alexander et al., 2015).

2. Materials and methods

2.1. Area of study

The first case study model represents the west coast of Scotland ecosystem (wcoS), which covers the continental shelf, defined as all sea area above the 200 m contour within ICES Division VIa (Fig. 1(a)). The wcoS area covers approximately 110,000 km^2, and includes the waters around the Outer Hebrides, Skye, the Small Isles, Mull, Islay, and the Firth of Lorn and Firth of Clyde island groups. The second case study model (Great Race) (Fig. 1(b)) occurs within the first case study area and covers approximately 6.25 km^2 of water and coastline off the west coast of the island Jura, within the Firth of Lorn. This is an area of potential interest to tidal energy producers due to the strong tidal stream within the site. Bordering the Great Race to the east and north are the Gulf of Corryvreckan and the Garvellachs respectively. These features are part of the Firth of Lorn SAC, where tidal developments would not be permitted inside and within 1 km of the site; therefore an area of outside of the SAC was chosen. The purpose of the second model was to test a fine-scale alternative to the coarser-scale wcoS site study.

Commercial fisheries operating in the wcoS area include demersal trawls, pelagic trawls, dredges, gillnets, longlines, creels and scallop fishing by hand with 1799 fishers operating 950 vessels as of 2013; providing 34 per cent of the total value of all Scottish landings (Scottish Government, 2014). The majority of fishers on the wcoS occupy the '10 m and under' section of the Scottish fleet, and focus upon demersal (mainly cod, haddock and whiting) and shellfish (mainly *Nephrops* and scallops) species, although mackerel is also of importance (Scottish Government, 2011).

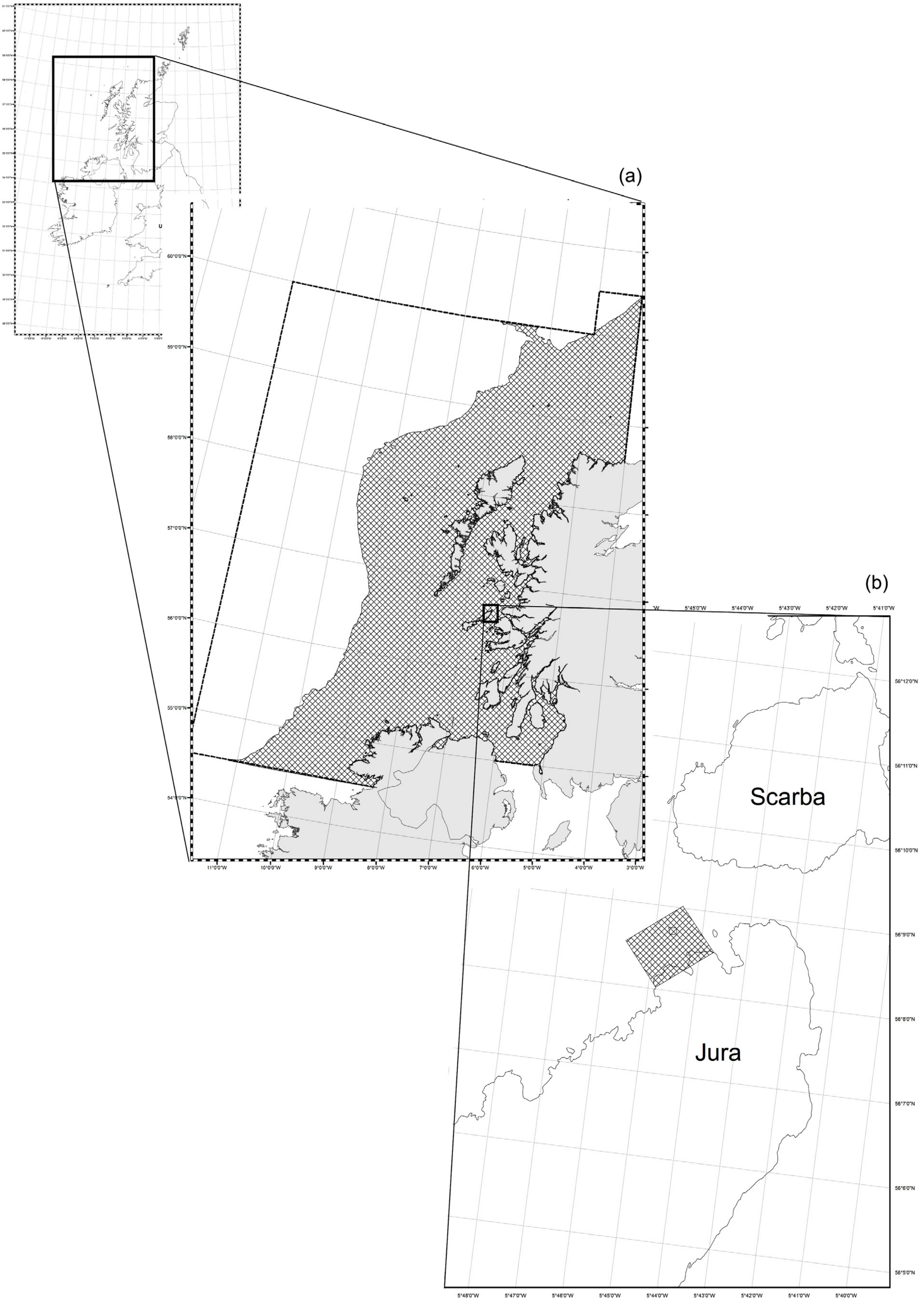

Fig. 1. Map of Scotland showing the case study areas (cross-hatched areas). (a) The study area for the 'wcoS' site. (b) The study area for the 'Great Race' site. The dashed outline on (a) represents the ICES VIa fishing area.

2.2. Spatial ecosystem model

Spatial simulations were based on the Ecopath with Ecosim (EwE) model of the west coast of Scotland (Alexander et al., 2015). The model comprises 41 functional groups including marine mammals (3), seabirds (1), fish (23, six of which were composed of adult and juvenile stages for cod, haddock and whiting), invertebrates (5), cephalopods (1), zooplankton (2), benthos (3), primary producers (2) and detritus (1). The model includes five fishing fleetswhich encompases all the fishing that occurs in the area: demersal trawl, *Nephrops* trawl, other trawl, pelagic trawls, and potting and diving.

The mass-balanced Ecopath model and time-series-fitted Ecosim model developed in Alexander et al. (2015) was used as the starting point for the simulations in Ecospace, the spatial component of EwE for both Ecospace models. Ecospace (Walters et al., 1999) dynamically allocates biomass across a raster grid map which is divided into a number of habitats to which functional groups and fishing fleets are assigned. Movement is represented using a 'Eularian' approach, where movement is treated as flows of organisms among fixed spatial reference points or cells without retaining information on where these organisms have been prior to that point. Ecospace represents spatial distribution of fishing mortality by gear type by using a relatively simple "gravity model" wherein the proportion of total effort allocated to each cell is assumed to be proportional to the relative profitability of fishing in that cell (Walters et al., 1999).

2.2.1. WcoS Ecospace parameters

The Ecospace habitat grid map was initially created in ArcGIS by combining seabed data (from the Mapping European Seabed Habitats project, http://www.searchmesh.net/) and depth data (from Seazone/Edina, http://edina.ac.uk/).

The data was combined, converted to comma-separated value (CSV) format and imported directly into Ecospace. As the grid is rectangular in shape, and the model developed by Alexander et al. (2015) did not include the area below the 200 m contour line, the cells to the north west of Scotland were designated as 'Off-shelf' (>200 m depth). The study area was divided into a grid of 43×70 cells; each cell represented an area of $\sim 8.5\ \mathrm{km}^2$. This resolution was selected as a compromise between showing detail and maintaining a high computing speed.

Functional groups were assigned to habitats based on a combination of data from the Marine Life Information Network (MARLIN) and the literature (Table 1). The functional group in question will have a higher feeding- and hence growth rate in the assigned habitat over other habitats; will have a higher survival rate in that habitat; and will have a higher movement rate outside of that habitat. Detritus and phytoplankton were assumed to be found everywhere, as were seabirds. The off-shelf area was not assigned as a preferred habitat for any functional group, which caused problems which will be discussed later.

In Ecospace, a fraction of the biomass of each cell is always moving. The amount of movement is known as the base dispersal rate (M) where M is the dispersal rate in km/year. M is the rate the organisms of given ecosystem would disperse as a result of random movements and is estimated from fish swimming speed (Martell et al., 2005) as:

$$M = \frac{\mathrm{Si}}{\pi L} \tag{1}$$

where Si is the swimming speed of different groups and L is the grid length (8.5 km). Dispersal rates were estimated based on published fish movement rates (Table 2). Where more than one published reference for fish movement rate exist, an average value was used. Within the 'dispersal' settings in Ecospace it is also possible to set a vulnerability to predation in 'bad (non-assigned) habitat' parameters (range from 1, not vulnerable – 100, very vulnerable, default of 2), relative feeding rates in bad habitat parameters (range from 0 to 1, default of 0.5), and relative dispersal in bad habitat (range from 1 to 10, default value of 2). Values for these parameters were estimated based on logic. Vulnerability to predation in bad habitats was set at 10 for lobsters, edible crabs, velvet crabs, other crustaceans, epifauna and infauna, who may be unable to escape easily in these habitats. Relative feeding rates for these species was also reduced to 0.1 due to the danger associated with foraging.

The distribution of fishing fleet activity is determined by assigning each fleet to a habitat(s). It was not possible to access spatial data for the distribution of fleets on the wcoS. It was assumed that given similarities in gear types, the North Sea and west coast of Scotland fleets were likely to operate in similar environments; therefore habitat allocations were based upon the North Sea model (Mackinson and Daskalov, 2007) (Table 3). Fixed and variable costs of fishing were taken from North Sea model estimates described in Heymans et al. (2011) and references therein. Effort and sailing costs were based upon ratios from Curtis et al. (2009). Costs of fishing can be calculated by Ecospace by combining fishing effort and sailing costs. Effort was held constant during the simulations as it is unknown how effort would change under the different scenarios. Sailing costs are calculated based upon the 'distance from port'. Ports were entered onto the Ecospace basemap.

2.2.2. Great Race Ecospace parameters

For the second 'case study' model, it was assumed that the species which exist within the wcoS model and their ecosystem interactions also occur within the Great Race ecosystem. Therefore, the same base Ecopath and Ecosim model was used as a starting point. Similar to the wcoS Ecospace scenario, a basemap of habitat types was developed in ArcGIS before translation into Ecospace and again the habitat classifications were dependant on depth and sediment data. High resolution (2 m) multibeam bathymetry data was sourced from the INIS Hydro project, accessible online via the United Kingdom Hydrographic Office (UKHO) INSPIRE portal (www.ukho.gov.uk/inspire/). Sediment data was sourced from the Mapping European Seabed Habitat (MESH) project (www.searchmesh.net/webGIS). The study area was divided into a grid that measured 100 by 100 cells, with each cell 25 m in length. Habitat preferences, fisheries preferences and dispersal rates were the same as those used for the wcoS ecosystem scenario. There were no commercial ports within the study area, therefore sailing costs were not included in the model and the cost of fishing not calculated.

2.3. Ecospace simulations

The wcoS study was a multiple MRED installation study, investigating effects of a number of energy extraction developments upon the wcoS ecosystem, as suggested by the Crown Estate Scottish offshore wind farm sites, and wave and tidal agreements for lease (Fig. 2). A description of the offshore wind farm, tidal and wave energy extraction sites included in this study can be found in Table 4. Four simulations were run in Ecospace for a period of 25 years (15 years after equilibrium was achieved):

1. A 'baseline' simulation with no changes to the ecosystem;
2. An 'artificial reef' simulation in which the installation areas were assigned as artificial reef habitats and mobile gears excluded;
3. An 'exclusions' simulation in which the installation areas were assigned as MPAs with no fishing allowed within the area;
4. A 'combined' simulation where installation sites were assigned as artificial reefs and were closed to all fishing.

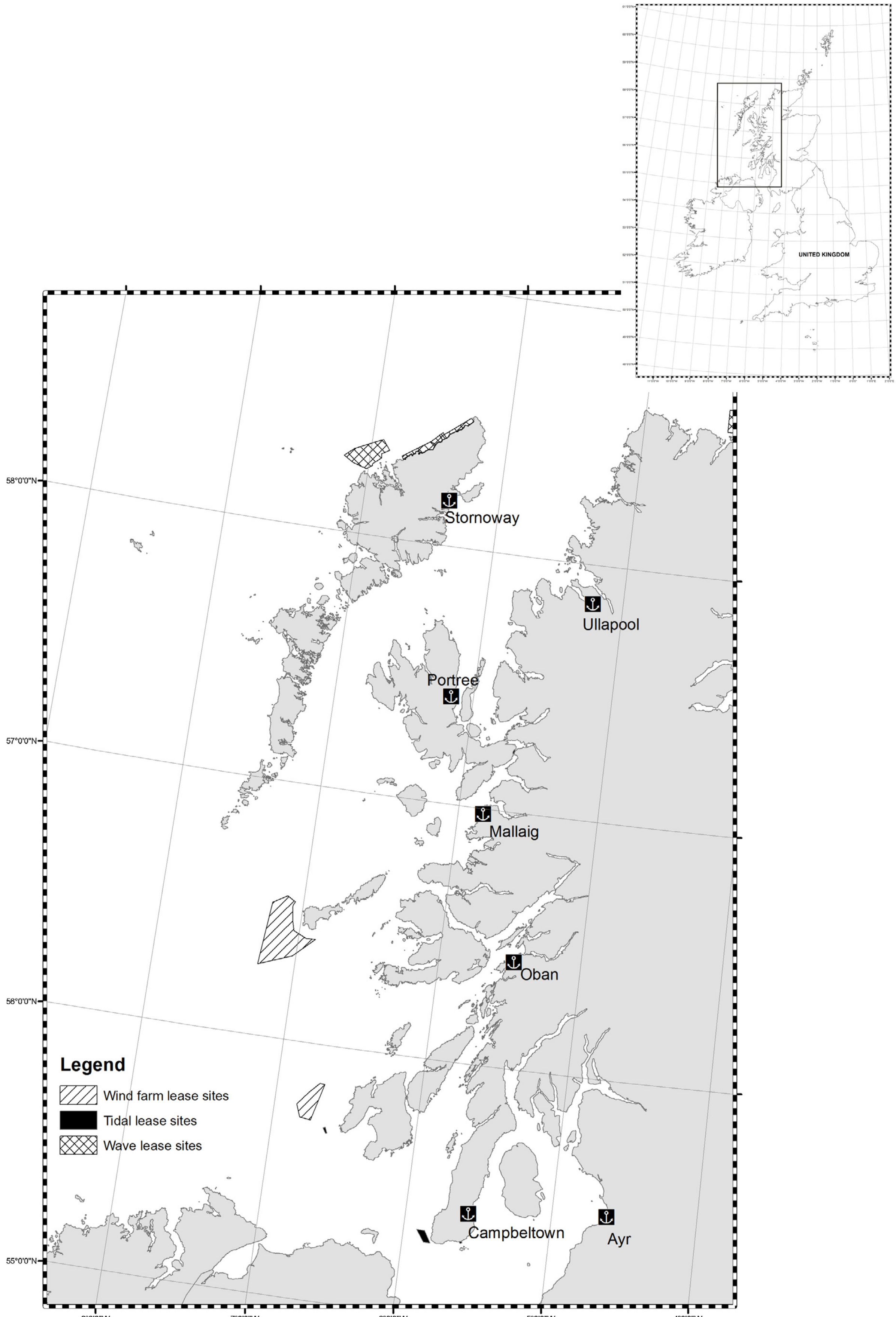

Fig. 2. Locations of sites proposed by the Crown Estate for MRED installations.

Table 1
Habitat assignations for all species. Assignations portrayed by + symbol.

Group/habitat	Bay	Mud 50–200	Rock 0–50	Rock 50–200	Sand 0–50	Sand 50–200	Sealoch	Sediment 0–50	Sediment 50–200	Off-shelf deep	Artificial reefs
Grey seals	+	+	+	+	+	+	+	+	+		+
Harbour seals	+	+	+		+	+	+	+	+		+
Cetaceans		+		+	+	+			+		
Seabirds	+	+	+	+	+	+	+	+	+		+
Cod mature	+	+	+	+	+	+	+	+	+		+
Cod immature	+	+	+	+	+	+	+	+	+		+
Haddock mature				+		+			+		
Haddock immature	+			+		+					
Whiting mature		+		+		+	+	+	+		
Whiting immature	+		+		+		+	+	+		+
Pollock	+	+	+	+		+	+	+			+
Gurnards	+		+	+	+	+		+	+		+
Monkfish	+	+	+	+	+	+	+	+	+		+
Flatfish	+	+	+	+	+	+	+	+	+		+
Rays	+	+			+	+		+	+		
Sharks	+	+			+	+	+	+	+		
Large demersals	+	+	+	+	+	+	+	+	+		+
Other small fish		+		+		+			+		
Mackerel	+	+	+	+	+	+	+	+	+		+
Horse Mackerel		+		+		+			+		
Blue Whiting		+		+		+			+		
Other pelagics	+	+	+	+	+	+	+	+	+		+
Herring	+	+	+	+	+	+		+	+		+
Norway pout		+		+		+			+		
Poor cod	+	+	+	+	+	+					+
Sandeel	+				+	+					
Sprat	+	+	+	+	+	+		+	+		+
Nephrops	+	+					+	+	+		
Lobster	+		+	+			+	+	+		+
Edible crab	+		+		+		+	+			+
Velvet crab	+		+		+		+	+			+
Crustaceans	+		+		+		+	+			+
Cephalopod	+	+	+	+	+	+	+	+	+		+
Large zooplankton	+	+	+	+	+	+	+	+	+		+
Small zooplankton	+	+	+	+	+	+	+	+	+		+
Infauna	+	+	+	+	+	+	+	+	+		+
Scallops	+				+	+	+	+	+		
Epifauna	+	+	+	+	+	+	+	+	+		+
Algae	+		+		+		+	+			+
Phytoplankton	+	+	+	+	+	+	+	+	+		+
Detritus	+	+	+	+	+	+	+	+	+		+

The Great Race study was a single installation study (composed of six turbines). Six simulations were run in Ecospace for a period of 25 years:

1. A 'baseline' simulation with no changes to the ecosystem;
2. A 25 m artificial reef simulation;
3. A 50 m artificial reef simulation;
4. A 500 m exclusion zone simulation;
5. A 25 m artificial reef and 500 m exclusion zone simulation;
6. A 50 m artificial reef and 500 m exclusion zone simulation.

3. Results

3.1. Multiple installation case study (wcoS model)

Under all scenarios, for the majority of species, biomass changes were <1 per cent (compared to the baseline run). Only the species of commercial importance with a biomass change of >1% will be discussed in this section.

3.1.1. Artificial reef and exclusion zone effects on food-web

Five commercial fish species experienced biomass changes of >1% in the AR and combined scenarios (Fig. 3(a)) with the largest positive change seen for monkfish (*Lophius piscatorius* and *Lophius budegassa*) which increased by 5% in both scenarios, and largest negative changes for pollock (*Pollachius pollachius* and *Pollachius virens*) by 1% and 2% and whiting (*Merlangius merlangus*) by 1% in both scenarios. However, none of the 5 species saw changes in biomass of >1% in the exclusion zone (EZ) scenarios. Four commercial shellfish species experienced changes in biomass of >1% (Fig. 3(b)). The largest changes were seen for Edible crab (*Cancer pagurus*) which was positively affected by the AR effect, increasing by 5%, and *Nephrops* (*Nephrops norvegicus*) and scallops (*Pecten maximus and Aequipecten opercularis*) which were negatively affected by ARs, by 3% in both. In general, species biomass in the wcoS study was more likely to be affected by ARs than EZs, with lobster benefiting most from the combined scenario.

3.1.2. Inside vs. outside MRED installations

The results showed that whilst, across all scenarios, there was a substantial change in biomass inside MRED installation areas, some species increased whilst others decreased (Fig. 4(a)–(c)). For commercial shellfish species, the wcoS model predicted that the ARs have a clear positive effect on biomass overall. For example, in the AR scenarios: *Nephrops* saw an increase of 98%, lobster saw an increase of 40%, edible crab saw an increase of 31% and scallops saw an increase of 44%. However, this may be an artefact of the basemap resolution issues mentioned earlier. In the EZ scenarios, only lobster saw a large increase in biomass of 54%. On the whole, most of the biomass changes occurred within the MRED installation sites.

Table 2
Dispersal rates for all species.

Group name	Base dispersal rate (km/year)	Dispersal in bad habitat	Vulnerability to predation in bad habitat	Feeding rate in bad habitat
Grey seals	1772.63	5	2	0.5
Harbour seals	2499.4	5	2	0.5
Cetaceans	3072.55	5	2	0.5
Seabirds	5761.03	5	2	0.5
Cod mature	330.89	5	2	0.5
Cod immature	416.57	5	2	0.5
Haddock mature	709.05	5	2	0.5
Haddock immature	416.57	5	2	0.5
Whiting mature	271.8	5	2	0.5
Whiting immature	416.57	5	2	0.5
Pollock	1240.84	5	2	0.5
Gurnards	555.42	5	2	0.5
Monkfish	401.8	5	2	0.5
Flatfish	401.8	5	2	0.5
Rays	212.72	5	2	0.5
Sharks	342.98	5	2	0.5
Large demersals	933.58	5	2	0.5
Other small fish	649.96	5	2	0.5
Mackerel	1039.94	5	2	0.5
Horse Mackerel	685.42	5	2	0.5
Blue Whiting	271.8	5	2	0.5
Other pelagics	189.08	5	2	0.5
Herring	206.81	5	2	0.5
Norway pout	153.63	5	2	0.5
Poor cod	271.8	5	2	0.5
Sandeel	180.81	5	2	0.5
Sprat	744.5	5	2	0.5
Nephrops	543.61	1	10	0.1
Lobster	14.18	1	10	0.1
Edible crab	49.63	5	10	0.1
Velvet crab	49.63	5	10	0.1
Crustaceans	49.63	5	10	0.1
Cephalopod	189.08	5	2	0.5
Large zooplankton	300	5	2	0.5
Small zooplankton	300	5	2	0.5
Infauna	13.78	1	10	0.1
Scallops	274.76	1	10	0.1
Epifauna	274.76	1	10	0.1
Algae	300	10	2	0.5
Phytoplankton	300	5	2	0.5
Detritus	10	5	2	0.5

Table 3
Habitat assignations for all fleets.

Fleet	Bay	Mud 50–200	Rock 0–50	Rock 50–200	Sand 0–50	Sand 50–200	Sealoch	Sediment 0–50	Sediment 50–200	Off-shelf deep	Artificial reefs
Demersal trawl	+	+	+	+	+	+		+	+		
Nephrops trawl	+	+							+		
Other trawl	+		+		+		+	+			
Potting/diving	+	+	+	+	+	+	+	+	+		+
Pelagic trawl		+		+		+			+		+

3.1.3. Artificial reef and exclusion zone effects on fisheries

In the wcoS study, ARs were most likely to negatively impact fishing fleets: *Nephrops* trawl (−3% AR, −2% AREZ), other trawls (−8% AR, −8% AREZ) and potters and divers (−2% AR, −0.5% AREZ). All trawlers were predicted to be negatively impacted by EZs, whereas potters and divers were predicted to experience an 0.5% increase in catch value. The changes in catch value occur due to changes in catch composition (e.g. Fig. 5). For example, under the wcoS AR scenarios, the demersal fleet was predicted to reduce its catch of cod by 37% and cod was a species which composed a significant component of the catch for that fleet as well as being of high value for that fleet (£1741 per tonne). This was also the

Table 4
Offshore wind, tidal and wave energy development sites included in this study.

Developer	Development	Type of development	Size
Scottish power renewables	Argyll Array	Wind	375 turbines with 1800 MW capacity
Scottish and Southern Energy	Islay	Wind	138 turbines with 680 MW capacity
Nautricity	Mull of Kintyre	Tidal	3 MW potential capacity
DP Energy	West Islay	Tidal	30 MW potential capacity
Lewis Wave Power Ltd.	North West Lewis	Wave	30 MW potential capacity
Pelamis Wave Power	Bernera Wave Farm	Wave	10 MW potential capacity

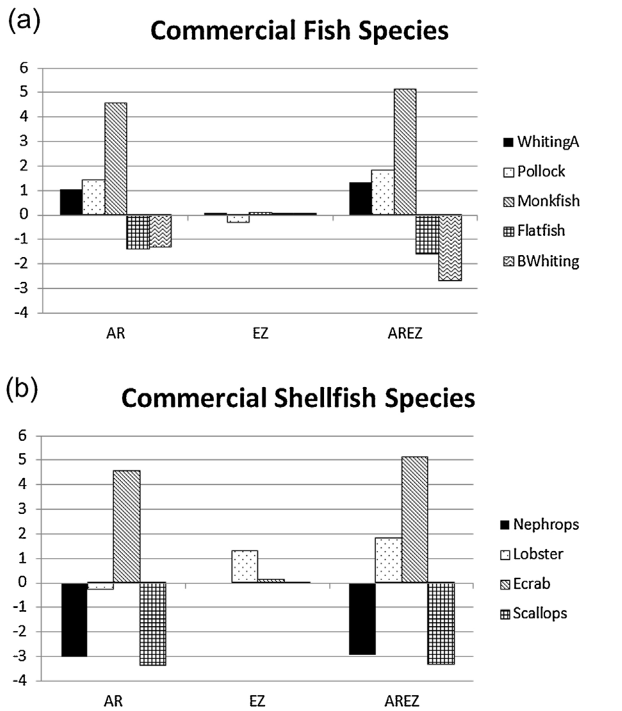

Fig. 3. Percentage biomass change from baseline for key commercial fish (a) and shellfish (b) species for the artificial reef effect (AR), exclusion zone effect (EZ) and combined effect (AREZ).

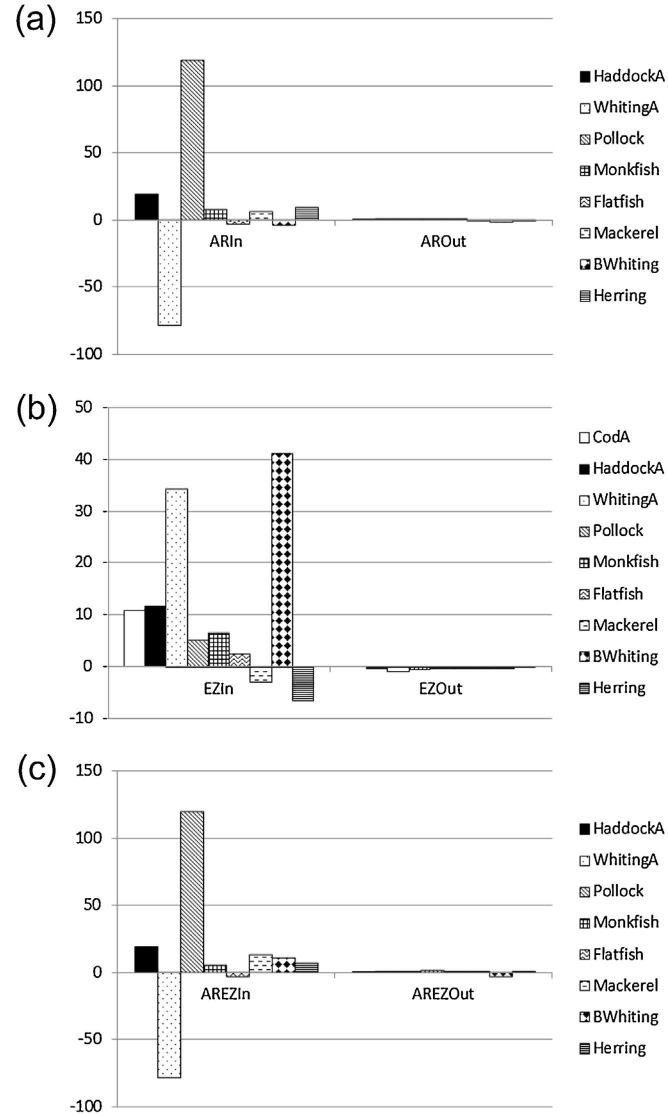

Fig. 4. Percentage biomass change from baseline for key commercial fish species, inside and outside of the MRED sites, for the artificial reef effect (a), exclusion zone effect (b) and combined effect (c).

case for lobster (23% reduction in catch, value of £8538 per tonne). Although there was an increase of species such as monkfish (+23%) and *Nephrops* (+46%), these were not of as high value for that fleet. Under the same scenario, the other trawls fleet experienced a large reduction of its lobster catch (−45%), also a species of high value to that fleet (£9284 per tone year^{-1}) and its scallop catch (−52%, £1850 per tonne). However, increases in species such as Norway pout (+83%) and sandeel (66%) did not make up for this loss due to their low value (£96 and £238 per tonne respectively) and even the increase in nephrop catch (69%, £7065 per tonne) could not mitigate the reduced catch value.

3.2. Single installation case study (Great Race model)

As with the multiple installation scale study, under all scenarios, for the majority of species, biomass changes were <1 per cent (compared to the baseline run). Only the species of commercial importance with a biomass change of >1% will be discussed in this section.

3.2.1. Artificial reef and exclusion zone effects on food-web

Nine commercial species saw biomass changes of >1% across the range of scenarios (Fig. 6(a)–(c)). The species with the largest positive change in the AR scenario was velvet crab (+16% 25AR, +25% 50AR). Six species, however, were affected negatively across the AR scenarios, mostly at the 25 m scale, particularly other crustaceans (−13%). Some species were also affected by the EZ effect, particularly Haddock (*Melanogrammus aeglefinus*) which saw a 22% increase in biomass in the EZ alone and with the 25 m and 50 m artificial reefs included, saw increases of 21% and 21% respectively. Similar to the wcoS study, lobster biomass increased in the EZ scenarios (11% across the EZ and combined scenarios). Overall, a larger number of species were affected in this case study with AR scenarios largely leading to negative changes to biomass and EZ scenarios leading to positive changes.

3.2.2. Inside vs. outside MRED installations

Similar to the wcoS study, the main changes to biomass for commercial species were seen within the MRED installation areas. In the 25AR and 50AR scenarios whiting saw an increase in biomass within the MRED areas (4in both), whereas cod and haddock saw decreases in biomass within the sites (between −2% and −9%). All species saw a very small decrease outside the MRED site (between −0.3% and −2%). In the EZ scenarios, only cod and haddock saw biomass changes of >1%. Haddock was predicted to increase substantially within (24%) and outside (20%) of the MRED sites. Cod was predicted to decrease within the site (−2%) and to increase outside of the site (9%). The results from the 25AREZ and 50AREZ scenarios showed similar changes to the EZ scenarios.

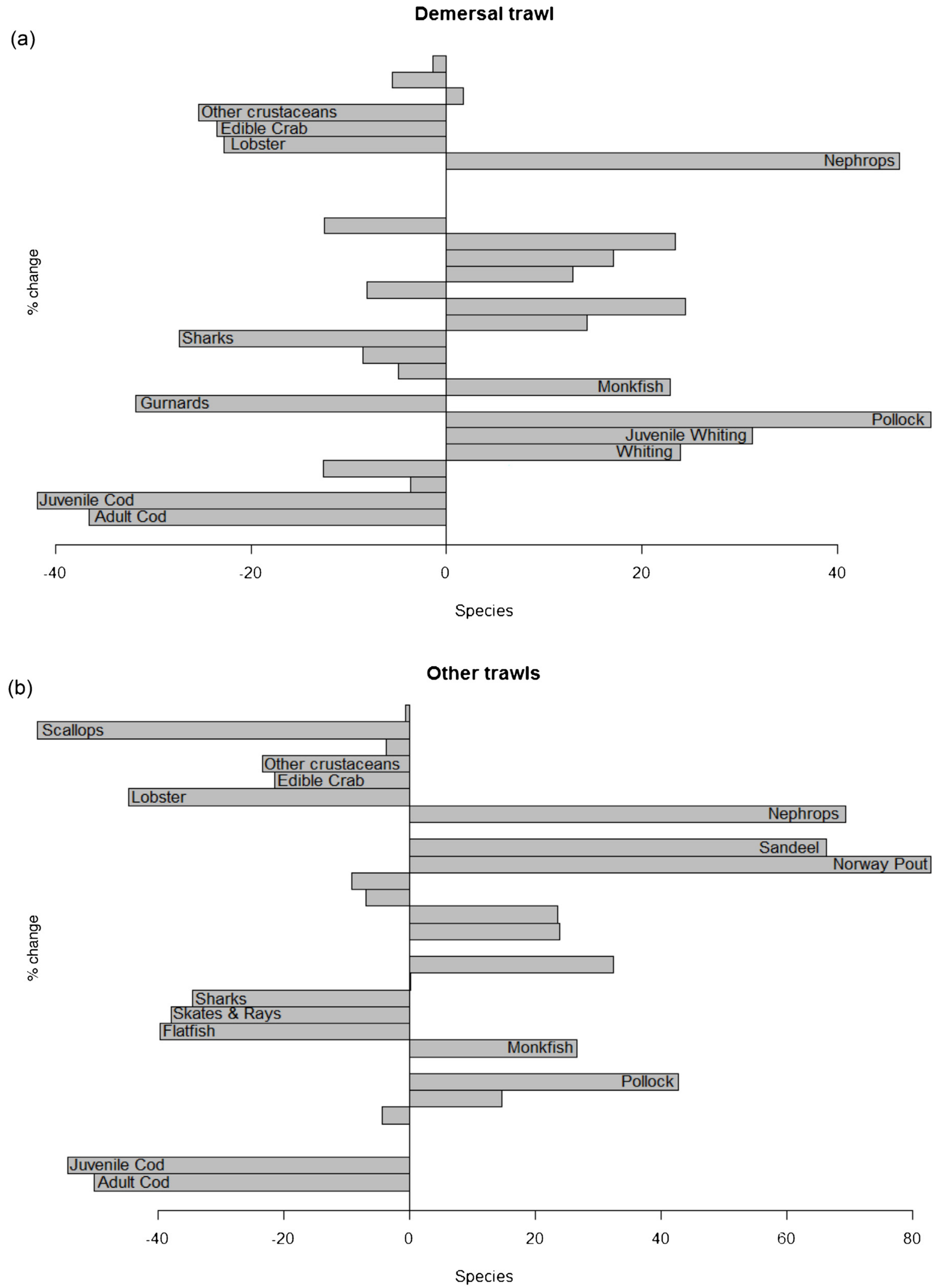

Fig. 5. Example of percentage change in catch composition for the demersal and other trawls. Only those species which are of commercial interest have been labelled.

3.2.3. *Artificial reef and exclusion zone effects on fisheries*

The Great Race model predicted that only the *Nephrops* trawl catch would be negatively impacted (−0.3% 25AR and −0.5% 50AR) and the values were small. The demersal trawl was predicted to receive the largest benefit from the AR effect (0.2% 25AR 0.4% 50AR), although again the values were small. Two fleets were predicted to receive less catch under the EZ scenarios: other trawls (-16% across the three scenarios – EZ, 25AREZ, 50AREZ), potters and divers (−2% across the three scenarios). Overall, similar to the wcoS study, this case study saw a decline in catch for the *Nephrops* trawlers in the

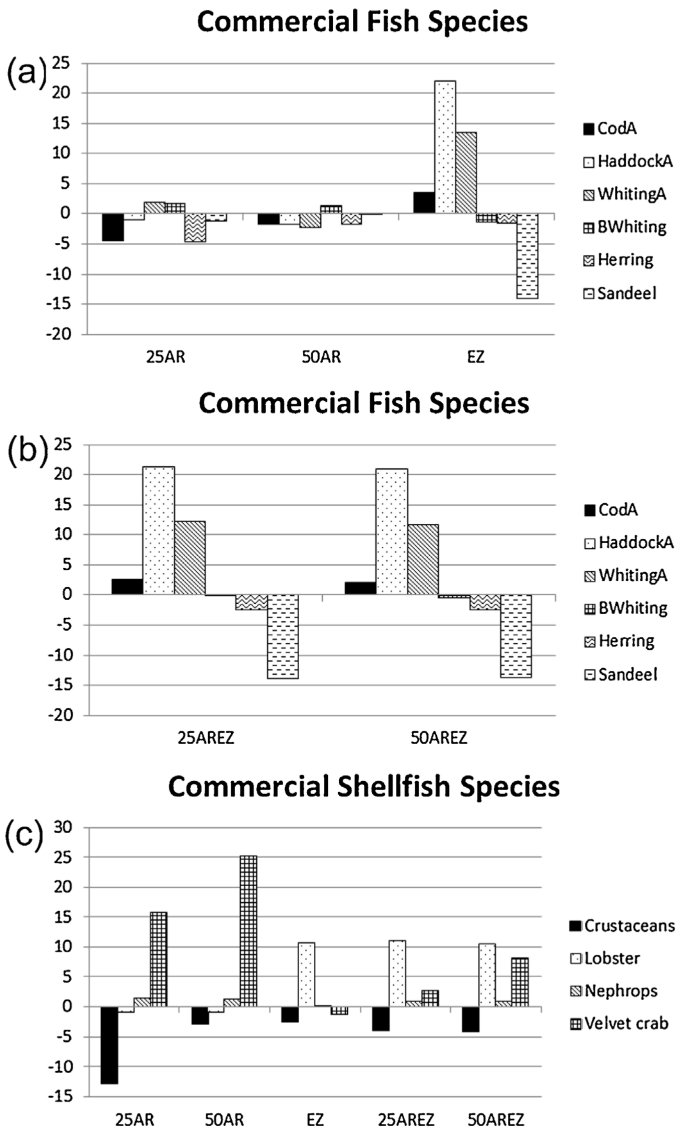

Fig. 6. Percentage biomass change from baseline for key commercial fish (a) and (b) and shellfish (c) species for the 20 m and 50 m artificial reef effect (25AR and 50AR), exclusion zone effect (EZ) and combined effects (25AREZ and 50AREZ).

AR simulations and for the *Nephrops* and other trawls in the EZ simulations.

4. Discussion

This is the first attempt to use a spatial food-web model to investigate the effects of MRED installations. The case study models used showed interesting findings with biomass changes mostly occurring within the MRED installation areas rather than outside; and predicted declines in catch value for the *Nephrops* trawlers in the AR scenarios and for the *Nephrops* and other trawls in the EZ scenarios. However, the findings of this study should be understood within the context of its being a first attempt.

4.1. Outputs from case study models

4.1.1. Artificial reef and exclusion zone effects on species of commercial interest

At the whole ecosystem scale, species biomass was more likely to be affected by ARs whereas at the single installation scale, species biomass was more likely to be affected by EZs. In both case studies, biomass changes were predicted to occur within the MRED installation areas rather than outside, suggesting that there is an effect of MRED installations. In the wcoS study, the biomass of species including monkfish, lobster and edible crab were predicted to increase within the AR simulations, both inside (larger increase) and outside (smaller increase) of the MRED installation sites. In the Great Race study, haddock and lobster were also predicted to increase inside and outside of the exclusion zone.

The area occupied by device foundations (creating ARs) will be small and given that the AR areas in the ecosystem scale case study covered the same extent as the EZs, the model biomass results are likely to be overestimated and unreliable. Despite this, it is worth discussing the modelled AR outputs further from a theoretical standpoint. It is also important to remember that the AR effect is based on combined habitat alteration and restriction of mobile fishing gears, such as trawlers and dredgers, due to potential gear damage/loss.

In both studies, monkfish saw an increase due to inclusion of ARs in the models, although the increase was much larger in the wcoS model study. It would be reasonable to assume, given that Ecospace is a food-web model, that an decrease in the predators of monkfish led to less predation and hence the increase in biomass. However, the modelled diet composition for monkfish does not include many predators and so changes to predator biomass was unlikely the reason for this increase. Monkfish do however have a large and varied diet in the models and many of their prey species increased under the AR scenarios, particularly herring which comprises nearly 11% of the monkfish diet. Thus, the increase of available food may have contributed towards the increase in monkfish biomass.

In the AR simulations the biomass of some crustacean species increased, particularly edible crab. Again, it might be reasonable to assume that an decrease in the predators of edible crabs led to less predation and an increase in biomass and in this case, this is the reason. A further explanation may relate to the increased habitat availability caused by inclusion of further rocky substrate which is a key habitat for crustaceans. In this case, the model prediction would be realistic given that crustaceans demonstrate a significant dependence on habitat availability and have been found to dominate some AR sites (Bortone et al., 1998; Jensen et al., 1994; Wilding and Sayer, 2002) including offshore wave power foundations (Langhamer et al., 2009).

Our wcoS case study model predicted small decreases in lobster and *Nephrops* biomass in the AR scenarios. In this case, predator biomass increases were not the reason as all predators of *Nephrops* (with the exception of poor cod) and of lobsters (with the exception of whiting) also experienced a reduction in biomass. Whereas most of the species discussed above (monkfish, edible crab and lobster) live in or above the sediment and rocky substrate, for *Nephrops*, an alternative explanation may be that they live in muddy substrate and the ARs modelled in the wcoS study reduced the availability of this type of habitat by changing (in most instances) sediment to rocky substrate and this may have led to the reduction in biomass. For lobster, the explanation may relate to the predicted large increase in velvet crab biomass which comprise 10% of their diet. The Great Race model alternatively predicted that *Nephrops* biomass would increase, albeit by a small amount, and this was also found in a study by Sayer et al. (2005) who used an Ecospace to model artificial reefs in Loch Linnhe on the west coast of Scotland. In this scenario, predator biomass also reduced; however, in both studies this may have been an artefact of using small spatial models and issues related to this will be discussed in the next section. The predicted lobster biomass decreases may be realistic as was also found in an empirical study in eastern Canada (Scarratt, 1968) where after two years lobster biomass on a rocky reef was lower than in productive lobster grounds nearby.

In the EZ scenarios, a larger number of species were affected at the single installation scale. In the Great Race case study model, haddock and lobster saw large increases of biomass in the EZ scenarios whereas in the wcoS study, lobster was the only species to experience a biomass increase. It would be realistic to assume that species biomass in the EZ scenarios was affected by changes in fisheries; however, whilst there were small reductions to lobster catch from trawl fleets in the EZ scenarios, there were also increases in catches by potting fleets (which already caught the larger share of that species), thus it is unlikely that changing access had any affect within the model scenarios. The same is true for haddock. As in the AR scenarios described above, lobster predators largely reduced however the explanation may relate to a predicted large increase in velvet crab biomass, a key prey component. For haddock, similarly to monkfish described above, there are few predators (with little change in biomass) and a wide variety of prey, some of which increased substantially such as sandeels and other small fish. Stocks of haddock were found to increase after fishing closures in e.g. Georges Bank (Gell and Roberts, 2003), thus this prediction, at least, may be reasonable.

Few direct efforts have been made to evaluate how reserve size itself affects the impact of no-take areas, although this has been assessed using Ecospace (Walters, 2000a). In the real world, reserves and closed areas have been found to work well across a range of sizes. Reserves of <1 km^2 have worked for sedentary animals living on coral reefs in the Philippines (Russ and Alcala, 1996). Similarly, closures totalling 17,000 km^2 on Georges Bank have turned around a long term decline of several exploited groundfish species as well as scallops (Murawski et al., 2000). Nonetheless, a comparison of 58 datasets from 19 European marine reserves (Claudet et al., 2008) indicated that size does matter: for every tenfold increase in the size of a no-take zone, a 35% increase in the density of commercial fishes was found. While our results appear to suggest the opposite, it is likely that it is simply easier to identify localised biomass changes at the small EZ scale, whereas at the whole ecosystem scale the EZ effect, and any related spill-over effect is less noticeable due to the small proportion of area the EZs occupy.

4.1.2. Artificial reef and exclusion zone effects on fisheries

Few other studies have undertaken an assessment of the impacts of offshore development upon the fishing industry. Those which have, found minimal impacts on the average incomes and financial profits of fishermen, although this was found to increase or decrease depending on the location of MPAs (Scholz et al., 2011). The case studies described here predicted differing effects on catch across scenarios, however, both studies predicted small declines in catch for the *Nephrops* trawls in the AR scenarios and for the *Nephrops* and other trawls in the EZ scenarios. Catch declines for N *Nephrops* trawls are likely due: firstly, trawling for *Nephrops* on AR sites would not be viable due to the gears used thus automatically making the area a no-take zone causing similar effects to the EZ scenarios; secondly, the wcoS study predicted a decline in *Nephrops* biomass in AR sites, likely due to habitat alteration, indicating that the fleet has less available biomass to catch. Although the Great Race model predicted an increase in *Nephrops* biomass under EZ scenarios, this increase was small and unlikely to lead to a spillover effect. Thus the loss of access to *Nephrops* fishing grounds would not be mitigated. Similarly, other trawls cannot operate on AR sites. Furthermore, scallop biomass decreased in the wcoS scenarios and saw little change in the Great Race scenarios again indicating that loss of access to scallop fishing areas for scallop dredgers, a key component of the other trawls fleet, would not be mitigated by the AR or EZ effects. This highlights the importance of including fishermen in the planning process (e.g. using novel participatory spatial planning methods such as that described in Alexander et al., 2012) when locating offshore extraction devices in order to minimise impacts upon the fishing industry.

In addition to a predicted lack of mitigation from ARs and EZs, changes in catch composition (wcoS study) meant that some fleets were predicted to no longer be able to catch high value species, also contributing to declining catch values. Under the majority of scenarios, fleets caught less high value species and more lower value species. This could lead to a situation whereby fishers cannot operate in traditionally fished areas or catch traditionally caught species, have to travel further to alternative sites in turn leading to increased steaming, increasing spend, longer fishing to break-even and a downward spiral of losing time and money (Alexander et al., 2013).

4.2. Utility of Ecospace to model MRED impacts/benefits

Given that Ecospace has previously been used to undertake studies relating to artificial reefs and marine protected areas independently (e.g. Christensen et al., 2009; Pitcher et al., 2002; Salomon et al., 2002), we hypothesised that it would be a suitable tool to use to investigate the spatial effects of MREDs. However, a number of difficulties were experienced during the modelling process and this, combined with the results, revealed a number of areas that should be improved upon in future studies.

We identified potential issues with the Ecospace software itself. That the spatial map must be rectangular led to problems in modelling an area which was not rectangular. A number of methods were used to address this: initially the non-used cells were allocated as land cells, but this meant that the model did not correctly predict the spatial distributions of species; then the cells were left as water cells but with no species allocated to live in them, however, this led to an 'edge effect' (where the biomass of some species concentrated around the boundary of the off-shelf area). One way in which this could be addressed would be the development of a variable rather than fixed spatial grid within the Ecospace module, where cells could be of different sizes to account for the actual geographical space being modelled (such as the non-overlapping, unstructured triangular grids used in models such as FVCOM, Chen et al., 2006). Additionally, at the time of undertaking this study, the software did not allow for simulations to be stopped mid-run, for the spatial map to then be altered, and for the altered model to run into the future. This meant that the scenarios had to be run over the period of the Ecosim model (1985–2008) which is of course unrealistic. However, has been addressed with the development of a spatial temporal model (Steenbeek et al., 2013). Finally, species were either assigned or not assigned to habitat types and preference could not be given to any one habitat over another, although this is being addressed through development of the habitat capacity model (Christensen et al., 2014) which should be implemented in future studies to improve results.

Regarding the models themselves, we experienced issues relating to data availability. It was not possible to obtain data by which to assign wcoS fleets to habitats, and therefore data was used from the North Sea model. Despite similarities in gear types between the North Sea and wcoS fleets, this is likely to be a poor substitute for wcoS fisheries data and may have had consequences for the simulations in terms of fleet distributions. Given that the focus is on the response of fishing fleets to MREDs, it is imperative that the fisheries data is reliable. Thus, in future studies, spatial data on the distribution of fleets on the wcoS should be collected and/or at locally relevant expert knowledge used. Furthermore, both base maps were created using data provided by the MESH project, which itself uses a large amount of modelled data, and in reality the seabed may be more complex and varied than the data suggests. This would mean that changes from one type of habitat to another caused by the AR aspect of MREDs within the modelled system is not reflected accurately. Perhaps most importantly, the lack of appropriate data also meant that it was not possible to assess the validity of the study findings as the models could not be cross-checked with existing spatial data.

The spatial scale of the models caused a number of problems. In the larger wcoS model, the coarse spatial scale (each cell = ~8.5 km^2) meant that smaller installations could not be considered (e.g. Scottish Power Renewables' Sound of Islay tidal energy development: http://www.scottishpowerrenewables.com/pages/sound_of_islay.asp) and the size of some installations were over-estimated; this subsequently overemphasised the scale of changes, particularly in the artificial reef scenarios. The coarse spatial scale also overemphasised the amount of habitat available for species such as algae and velvet crab which occupy a narrow niche (the photic zone), therefore the models estimated unrealistic biomass changes (e.g. 695% during the baseline run for velvet crab). This had implications for the scenarios as shown in the results where a number of species who predated upon velvet crab experienced large biomass increases. Even in the fine-scale model, there was a large increase in velvet crab biomass. In the finer-scale model, there were a number of issues relating to the lack of changing Ecopath parameters to reflect differences at that scale; essentially the finer scale model was the wcoS model with a different Ecospace basemap. However, as noted by Walters (2000b), dispersal rates, trophic responses and spatial distribution of fishing effort can have an effect on modelling small areas in Ecospace – and only habitat, species allocation and fishing effort was amended at the finer spatial scale. In the creation of a small scale and highly detailed basemap, the model was used on a much finer scale that it was conceptually designed for raising issues relating to flow across boundaries which were not taken into account. It may be possible to create a model at this scale for modelling e.g. benthos, but in this case, far ranging species were included in the food web and so therefore flows in and out of the spatial model were not properly represented. It is a fine balancing-act to get the resolution of the Ecospace basemap correct. This is something which should be considered carefully as it can have large implications for modelled scenarios. The area of an Ecospace model should really encompass most of the area of the important species being modelled.

Other spatial models have been created such as Marxan (Ball et al., 2009) (a spatial decision support system used in conservation planning, most notably the Great Barrier Reef), the Cumulative Impacts Model (Halpern et al., 2008) (used to compare least and most impacted regions as part of ecosystem based management) and InVEST (Kareiva et al., 2011) (a family of tools, run in ArcGIS, to map and value ecosystem goods and services). A number of other ecosystem models also exist, e.g. Atlantis (Fulton et al., 2011) and ERSEM (Baretta et al., 1995). To address questions relating to the AR and EZ effects of MREDs in areas such as the wcoS and their mitigation potential, models with the following characteristics would be needed: the ability to model data poor ecosystems; the ability to model human-induced spatial changes to habitat at specific points in time; and the ability to identify how these changes affect species of commercial significance. None of these other models are food web models, and therefore changing interactions between species as a result of changes to the ecosystem cannot be accounted for using these algorithms. It may be that combining these tools may be useful in generating a general indication of commercial species biomass and fishing trends in response to changes in the environment (such as the deployment of marine renewable energy installations). For example Marxan and Ecopath with Ecosim has been used to investigate potential trade-offs associated with MPA management strategies (Metcalfe et al., 2015). Nonetheless, any combination of these models at this time would not be able to address a human-induced temporal change to habitat at a point in the future, an aspect of Ecospace which has been addressed through the temporal-spatial module (Steenbeek et al., 2013). Furthermore, the models will only be as good as the data used to create them and due to the limited data availability on the wcoS, this would still be a problem, particularly for data-intensive ecosystem models such as Atlantis. EwE remains the most suitable ecosystem modelling routine for characterising data-poor systems (Ainsworth and Pitcher, 2005). Given these issues, it may be that Ecospace is currently the most useful tool for investigating the impacts of MREDs on the wcoS, if the issues raised above can be addressed.

5. Conclusions

This research represents the first study undertaken to assess the potential consequences of MRED deployment for the fishing industry, particularly in terms of understanding changes to the ecosystem upon which the industry relies. Using two case study models, we investigated whether MREDs can positively benefit the food-web, and the fishing industry which relies upon it by providing: (a) habitat through the 'reef-effect' and (b) protection through the 'exclusion zone effect', thus mitigating a potential loss of access issue for the fishing industry. The findings of this study suggest that it is currently not possible to definitively state whether the effects of MREDs upon the ecosystem would mitigate any loss of access to fishing areas caused by MREDs, particularly given the number of problems identified during the modelling process.

The results of the two case studies could be made more reliable through further observational studies of which species are found to live on MRED sites, the inclusion of data on the actual AR footprints of MREDs and the spatial distribution of wcoS fisheries. Further research should also be taken to understand the spatial distribution of species biomass on the wcoS in order to validate the baseline models and to update the habitat use of the species in the model using the habitat capacity module of Ecospace. Despite the limitations experienced during this study, we argue that Ecospace, given current ongoing developments, could be an appropriate tool to address questions relating to the spatial effects of MREDs.

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