

2021-07-22

Predicting sea pen (Pennatulacea) distribution on the UK continental shelf: evidence of range modification by benthic trawling

Downie, AL

<https://pearl.plymouth.ac.uk/handle/10026.1/21251>

10.3354/meps13744

Marine Ecology Progress Series

Inter-Research Science Center

All content in PEARL is protected by copyright law. Author manuscripts are made available in accordance with publisher policies. Please cite only the published version using the details provided on the item record or document. In the absence of an open licence (e.g. Creative Commons), permissions for further reuse of content should be sought from the publisher or author.

1 **Predicting sea pen (Pennatulacea) distribution on the UK continental shelf: evidence of**
2 **range modification by benthic trawling**

3
4 Anna-Leena Downie^{1*}, Tamsyn Noble-James¹,

5 Ana Chaverra^{1,2} and Kerry L. Howell²

6
7 ¹*Centre for Environment, Fisheries and Aquaculture Science,*
8 *Pakefield Rd, Lowestoft, NR33 0HT, UK*

9 ²*Marine Biology & Ecology Research Centre, Marine Institute, Plymouth University,*
10 *Plymouth, PL4 8AA, UK.*

11
12 * *Corresponding Author:* anna.downie@cefas.co.uk

13
14 **Abstract**

15 Sea pen communities are Vulnerable Marine Ecosystems (VME) which occur worldwide in soft
16 bottom sediments where trawling often occurs. The ability of marine managers to assess,
17 monitor and mitigate impacts to sea pens at national scales has, however, been constrained
18 by a limited understanding of environmental requirements, geographical distribution, and
19 responses to trawling. In this study we use Random Forest species distribution modelling
20 (SDM) to predict the distribution of suitable habitat for the tall, slender, and phosphorescent
21 sea pens (*Funiculina quadrangularis*, *Virgularia mirabilis* and *Pennatula phosphorea*) on the
22 UK continental shelf, exploring the results relative to the distribution of fishing activity.
23 Occurrence of all three species corresponded to areas of low current and wave velocity,
24 where suspended matter in the water column was also low. However, for *F. quadrangularis*,
25 the largest species, the models indicated substantially different drivers of distribution

1 between the Greater North Sea and Celtic Seas ICES Ecoregions. This disparity appears to
2 reflect modification to the range and realised niche of this species in the Greater North Sea,
3 due to trawling impacts. *P. phosphorea* and *V. mirabilis* appear to be more resilient to
4 trawling, with no clear negative relationships observed. Our findings illustrate the value of
5 broadscale qualitative comparisons between SDMs and human activity data for insights on
6 pressure-state relationships. When combined with robust distribution maps, this improved
7 understanding of vulnerability will enable marine managers to make ecologically sound,
8 defensible decisions and deliver tangible conservation outcomes for sea pen communities.

9

1 1 Introduction

2 Sea pens (Pennatulacea) are globally distributed colonial octocorals that are adapted to
3 inhabit soft muddy or sandy sediments. Standing erect from the seabed, often in dense
4 aggregations or 'fields', they provide structural complexity in otherwise featureless
5 sediments, creating microhabitats, shelter and attachment substrata for sessile and motile
6 fauna (Buhl-Mortensen & Buhl-Mortensen 2014, De Clippele et al. 2015). Such aggregations
7 also function as nursery grounds for commercially important fish (e.g. the redfish genus
8 *Sebastes*; Baillon, 2012) and serve as a food source for a range of invertebrates (García-
9 Matucheski & Muniain 2011, Gale et al. 2013). Despite their wide distribution, high
10 abundance and functional value, sea pens remain poorly studied in comparison to hard corals
11 (Scleractinia).

12

13 Sea pens, as relatively slow growing and long-lived organisms, are vulnerable to damage,
14 displacement or removal by demersal fishing activities (Hixon & Tissot 2007, Malecha & Stone
15 2009, Lauria et al. 2017). As such, their conservation status is acknowledged through various
16 international legislative and policy instruments, with sea pen communities being recognised
17 as Vulnerable Marine Ecosystems (VME) by the United Nations General Assembly (Rogers &
18 Gianni 2010). Conservation measures for VMEs have advanced globally in recent decades (as
19 summarised by Aguilar et al. 2017), with a variety of spatial management measures adopted
20 at national and international scales, including the establishment of Marine Protected Areas
21 (MPAs). Although legislative provisions have improved, functional implementation of spatial
22 measures has been slow; partly due to a poor understanding of environmental requirements
23 and geographical distributions of VMEs (Aguilar et al. 2017). This is particularly relevant at the

1 scale of individual nations, given the fundamental role of national frameworks in driving
2 spatial conservation measures (Grip 2017). As sea pens are vulnerable to trawling impacts, a
3 clear understanding of their environmental requirements and geographical ranges (both
4 observed and theoretical) is essential to underpin spatial management of fishing activity
5 within and beyond MPAs.

6

7 The United Kingdom continental shelf (UKCS) is a good example of a marine region that is
8 extensively and chronically impacted by mobile demersal fishing, resulting in modification of
9 biological assemblages and biogeochemical functions (Tillin et al. 2006, Thurstan et al. 2010,
10 Kröger et al. 2018, Rijnsdorp et al. 2018). Sea pens in this region inhabit mud-rich, depositional
11 habitats (Hill & Wilson 2000, Greathead et al. 2007, 2015), resulting in particular vulnerability
12 to the commercially important *Nephrops norvegicus* (or '*Nephrops*') fishery, which deploys
13 otter trawls on muddy sediments, abrading the seabed surface and penetrating the top few
14 centimetres (Ungfors et al. 2013, Eigaard et al. 2016). UK *Nephrops* landings have risen
15 dramatically over recent decades in comparison to other European countries (Ungfors et al.
16 2013, Marine Management Organisation 2019), and the targeted muds are reported to be
17 more significantly impacted by trawling, and to take longer to recover, than sand or gravel
18 habitats (Hiddink et al. 2006, Rijnsdorp et al. 2016). Despite the potential threats to sea pen
19 communities from demersal trawling, no studies have yet explored these impacts at the scale
20 of the UKCS.

21

22 Sea pen communities on the UKCS comprise the tall sea pen (*Funiculina quadrangularis*),
23 slender sea pen (*Virgularia mirabilis*) and phosphorescent sea pen (*Pennatula phosphorea*),
24 the ecology and ranges of which have been explored in territorial Scottish waters by

1 Howson et al. (1994) and Greathead et al. (2007, 2015). Due to the relative paucity of research
2 on the responses of these species to trawling abrasion, sensitivity assessments and
3 evaluations of population viability in fished areas largely depend on physiological traits
4 information and studies of analogous species (Hill & Wilson 2000, Ager 2003, Jones 2008),
5 supported by a small number of local-scale trawling impact studies (Howson & Davies 1991,
6 Tuck et al. 1998, Greathead et al. 2005, Murray et al. 2015). The available evidence suggests
7 that the sensitivity of these species to trawling impacts may vary depending on physiology
8 (i.e. flexibility and recovery potential) and retraction ability (Hill et al. 2020). *F.*
9 *quadrangularis*, the largest and least often recorded of the three species (exceeding 200 cm
10 in length) is considered the most sensitive to trawling impacts given its inability to retract its
11 brittle axial rod beneath the sediment (Greathead et al. 2007) and the slow growth rate of
12 analogous species (Wilson et al. 2002). The smaller and more often encountered *V. mirabilis*
13 and *P. phosphorea* are able to retract into burrows and are generally considered less
14 vulnerable than *F. quadrangularis* (Greathead et al. 2007, Ambroso et al. 2013). There is some
15 evidence to support the lower sensitivity of these two species, although direct impact studies
16 have thus far been confined to the north-western Scottish inshore (Greathead et al., 2005;
17 Tuck et al., 1998).

18

19 Species distribution models (SDM) are increasingly used in management of marine habitats,
20 where determining actual species distribution can be logistically and financially challenging.
21 SDMs provide geographically broad spatial predictions of environmental suitability for
22 specified fauna, based on sample data and environmental data layers. The increasing
23 accessibility of remotely sensed environmental data products now allows extrapolation from
24 a limited amount of sample data to much larger areas of seabed (Tyberghein et al. 2012, He

1 et al. 2015). A number of regional scale studies have applied the SDM approach to predict sea
2 pen distribution (Krigsman et al. 2012, Knudby et al. 2013, Kenchington et al. 2014, Beazley
3 et al. 2016, Murillo et al. 2016, Lauria et al. 2017, Bastari et al. 2018, Kinlan et al. 2020),
4 including a study of the Scottish West Coast, lochs and islands by Greathead et al. (2015). In
5 addition to analysing species environmental requirements and extrapolating their
6 distributions into unsampled areas, SDMs can also be used to shed light on external factors
7 affecting natural distributions. The predicted distributions reflect the input data, and models
8 using data from impacted populations will predict impacted distributions, allowing
9 comparison to natural distributions. These predicted distributions can then be evaluated
10 relative to the known distribution of anthropogenic pressures, to support marine spatial
11 planning and management (e.g. Marshall et al. 2014, Reiss et al. 2015). This linkage of species
12 observations, environmental parameters and anthropogenic pressures enhances
13 understanding of 'pressure-state' relationships, enabling defensible, evidence-based
14 management decisions, for example; exclusion of human activities from areas of high species
15 vulnerability and establishment (and modification) of ecologically coherent, well-connected
16 MPA networks (e.g. Smith et al., 2009; Sundblad et al., 2011). To this point the majority of
17 SDM studies on sea pens have focused on spatial predictions of environmental suitability and
18 have not linked predicted species distribution to anthropogenic impacts. A synergistic
19 approach combining spatial predictions by SDM and comparison to anthropogenic activity
20 data would improve the ability of marine managers to assess, mitigate and monitor impacts
21 to sea pen communities. Thus far, to our knowledge, this connection has not yet been made
22 at a national scale.

23

1 In this study we present the first application of SDM to sea pens at the scale of the UKCS,
2 using Random Forest classification models (Breiman 2001) to investigate environmental
3 drivers of *F. quadrangularis*, *V. mirabilis* and *P. phosphorea* distribution and predict their
4 occurrence across the UKCS. Here we use modelled environmental parameters to predict
5 suitable habitat for the three species, training and validating the models using a large legacy
6 trawl and imagery dataset. The predicted distributions are mapped and evaluated in the
7 context of demersal fishing activity data. Finally, we use these results to explore whether
8 anthropogenic pressures may have shaped the observed distribution of sea pens on the UKCS
9 and discuss the implications for sea pen management and conservation.

10

11 **2 Methods**

12 The study area (Figure 1) covers most of the continental shelf inside the UK Exclusive
13 Economic Zone (EEZ). The spatial extent of the study was determined by the extent of the
14 sediment composition layers (Mitchell et al. 2019a, b) used as predictor variables in the
15 models. Consequently, the northernmost tip of the shelf EEZ is excluded due to lack of
16 environmental data coverage. The full study area is hereafter referred to as the UK
17 continental shelf (UKCS). Based on preliminary results indicating differences in modelled sea
18 pen distribution patterns and environmental responses between sea areas, the study area
19 was further divided into two separate oceanographic 'Ecoregions', Celtic Seas (CELS) and
20 Greater North Sea (GRNS), as delineated by the International Council for the Exploration of
21 the Sea (ICES) (ICES 2020). All environmental data and species observations were aligned to
22 a raster grid of the study area with a cell size of 0.002 degrees (~200 metres).

1 2.1 Species presence / absence data

2 Presence / absence sample data for the tall sea pen (*Funiculina quadrangularis*), slender sea
3 pen (*Virgularia mirabilis*) and phosphorescent sea pen (*Pennatula phosphorea*) were collated
4 from multiple legacy sources, comprising data collected using underwater imagery, dive
5 surveys, benthic trawls and grabs over a period spanning 1961 to 2019. The majority of
6 observations were derived from the publicly available UK Marine Recorder database (Public
7 snapshot v20170825, available from <https://incc.gov.uk/our-work/marine-recorder>,
8 downloaded on 01/11/2017). These were supplemented by additional underwater imagery
9 and grab samples collected as part of the UK Marine Protected Areas Programme, as well as
10 Cefas benthic trawl surveys and grab sampling conducted under various research and
11 monitoring programmes.

12
13 Presence / absence observations for each of the three sea pen species were reduced to one
14 observation per 0.002 degree (~200 m) raster cell. For trawls and video tows, where co-
15 ordinates were recorded at the beginning and end, the positional midpoint was used as the
16 observation location. The precision of positioning for the observations differed based on both
17 their vintage and sampling method, but this was mitigated by the size of the raster cells and
18 consequently the spatial scale of prediction. Each raster cell with at least one intersecting
19 presence observation was classified as a presence. Absence observations from grab samples
20 were excluded as they are considered to under-sample large, sparsely distributed epifauna
21 such as sea pens. To limit the effects of spatial autocorrelation and redundancy, the data in
22 raster cells were further spatially subsampled to exclude neighbouring points closer than 250
23 metres. Datasets with very low prevalence lack the ability to adequately represent the nature

1 of species dependence on environmental conditions (Santika 2011). Hence, a final
2 subsampling step was conducted individually for each species and region to randomly down-
3 sample absence records, achieving a 10% prevalence for each dataset used in models. Table
4 1 shows the total number of raster cells with a presence / absence record and the number of
5 cells included for modelling in each region.

6

7 **2.2 Environmental predictor layers**

8 The environmental predictor layers included in model development, their units, sources and
9 spatial resolution of the source data are listed in Table 2. All layers were resampled to the
10 native resolution of the bathymetry layer (0.002 degrees, or ~200 m). Where necessary,
11 values were interpolated using the Empirical Bayesian Kriging function in ArcGIS10.5
12 Geostatistical Analyst (with default settings).

13

14 SAGA GIS tools for QGIS (v. 3.2; Conrad et al. 2015) were used to calculate a set of regional
15 terrain variables that, whilst terrestrial in origin, can be used to represent aspects of seabed
16 topography. The regional terrain variables calculated include Channel Network Base Level,
17 Channel Network Distance, Valley Depth, Closed Depressions, Relative Slope Position and
18 Standardised Height (Shaw et al. 2004). The concept of the channel network base level is used
19 to distinguish topographic highs and lows, by using the Digital Elevation Modelling (DEM) to
20 create a channel network. The channel network base level is an interpolated elevation surface
21 connecting the channel elevations. The channel network distance is calculated as the vertical
22 distance between the DEM elevation and the channel network base level elevation. Valley
23 depth is calculated as the vertical distance to the lowest elevation of source flow. Closed

1 depressions represent sinks in the topography, surrounded on all sides by higher ground, with
2 values increasing with the height difference to the surrounding higher ground. The relative
3 slope position ranges from 0-1 as an index of location along the entire length of a slope.
4 Standardised height is the product of normalised height (value ranging from 0-1 from the
5 lowest to the highest position within a respective reference) multiplied by absolute height,
6 relating a location to its wider surrounding topology (Shaw et al. 2004).

7

8 Layers used for current and wave velocity at the seafloor, winter suspended particulate
9 matter, sand, mud and gravel fractions, as well as sand to gravel log ratio and mud to gravel
10 log ratio are those produced by Mitchell et al. (2019c), available for download through the
11 Cefas Data Hub (Mitchell et al. 2019a,b; Table 2). Mean, maximum and minimum bottom
12 temperatures, along with the mean and maximum annual temperature ranges for 2017-2019,
13 were calculated from daily-mean seabed temperatures (C°), obtained from a high resolution
14 North-West European Shelf forecasting ocean assimilation model, downloaded from
15 <http://marine.copernicus.eu/> (Table 2). Seasonal averages for spring (March, April, May),
16 summer (June, July, August) and autumn (September, October, November) of chlorophyll-a
17 concentrations (mg m^{-3}) and net primary productivity of carbon ($\text{mg m}^{-3} \text{ day}^{-1}$) between 2017-
18 2019 were computed from monthly composites of ESA Ocean Colour CCI Remote Sensing
19 Reflectance data and the UK Met Office Operational Suite Atlantic Margin Model FOAM
20 output downloaded from <http://marine.copernicus.eu/> (Table 2).

21

22 Vector layers of ICES Ecoregions (ICES, 2015) and ICES Statistical Areas (ICES, 2005) were
23 rasterised and included as factor variables acting as proxies for biogeographical attributes and
24 oceanographic conditions in the various sea areas surrounding the UK.

1

2 **2.3 Model training and validation**

3 Originating from the field of machine learning, the Random Forest method is increasingly used
4 by ecologists due to its high classification accuracy and ability to characterize complex
5 interactions between variables (Prasad et al. 2006, Cutler et al. 2007). Three Random Forest
6 distribution models (Breiman 2001) were built for each sea pen species; 1) a full model trained
7 using data from the entire study area (hereafter referred to as the UKCS model), 2) a model
8 trained using only data from the Greater North Sea (hereafter referred to as the GRNS model)
9 and 3) a model trained using only data from the Celtic Seas (hereafter referred to as the CELS
10 model). Each model was used to predict the distribution onto the entire study area (UKCS).
11 Transferability of models between the two Ecoregions was tested by calculating model
12 performance statistics on test data separately for each. In the case of the UKCS model,
13 accuracy was tested for the full UKCS test dataset and for each Ecoregion.

14 All analyses were run using the R statistical computing software (vs. 3.5.1, R Core Team 2018).
15 Pre-selection of predictor variables was conducted using the *Boruta* algorithm in the 'Boruta'
16 R package (Kursa & Rudnicki 2010), to reduce the number of variables included in the model
17 for ease of interpretation and to allow comparison between species. The algorithm compares
18 the importance of a variable calculated by a Random Forest model to the importance of a
19 random permutation of the same variables over several iterations. Variables with higher than
20 random importance for a least one species were considered for inclusion in the final predictor
21 variable set. Of the correlated predictor variables (correlation coefficient > 0.6), the variable
22 with the highest average importance across all three species (calculated by *Boruta*) was
23 selected to be included in the final set of variables. The UKCS model also included Ecoregion

1 as a factor variable. The final set of predictor variables included in models is indicated in
2 Table 2.

3 As our dataset was compiled from multiple sources, consideration was given to retaining one
4 or more of the datasets for independent validation. However, combinations of all the
5 component datasets were required to achieve full geographical coverage of the study area
6 for both training and testing the models. Consequently, a multi-run cross-validation on the
7 full dataset was determined to be the more appropriate validation strategy, affording the
8 most comprehensive training and test datasets, together with an approximation of model
9 stability over multiple subsets of data. The 'biomod2' package (vs. 3.4.6, Thuiller et al. 2009,
10 2020) was used to train and test 10 cross-validation (CV) runs of each model to control for
11 any artefacts resulting from a single split of the data into training and test data sets by random
12 selection. The 10 training and test data splits were created by assigning two five-fold cross-
13 validation sets with equal numbers of presences included in each for balance, resulting in an
14 80/20 split into training/testing data.

15

16 Random Forest models were built using 500 trees and five variables randomly sampled as
17 candidates at each split. Variable importance statistics and partial response curves were
18 extracted from each cross-validation run. Predictions from each CV run were dichotomized
19 into presence / absence using a threshold that optimizes the True Skill Statistic (TSS; Allouche
20 et al. 2006). TSS was selected based on its insensitivity to prevalence and its equal weighting
21 of sensitivity and specificity, avoiding pitfalls of both the Kappa statistic and 'area under the
22 receiver operating characteristic curve' (AUC) highlighted in numerous studies (including
23 Manel et al. 2001, Lobo et al. 2008, Jiménez-Valverde 2012). The final predicted distribution

1 layer from each model combined the dichotomized outputs from all 10 CV runs, classified into
2 three categories; 1) absence, 2) low likelihood of presence (1-5 CV runs predict presence) and
3 high likelihood of presence (> 5 CV runs predict presence).

4 The accuracy of predicted presences / absences were examined using the 'PresenceAbsence'
5 package (Freeman & Moisen 2008). Specificity, sensitivity (Fielding & Bell 1997), the Kappa
6 statistic and TSS were calculated for the corresponding test data in each cross-validation run.
7 For the UKCS model separate statistics were calculated by filtering the test data by each
8 Ecoregion in turn. For testing the transferability of the model trained on data from one
9 Ecoregion to the other, the test data from the corresponding cross-validation run from the
10 other Ecoregion were used.

11 **2.4 Overlap with demersal fisheries**

12 The potential contribution of bottom contact fishing impacts to the observed difference in
13 environmental preferences of *F. quadrangularis* between the two Ecoregions was
14 investigated further. Spatial fishing intensity data were overlain with presence as predicted
15 by the CELS model, and the ratio of observed absences in impacted to unimpacted sample
16 locations was compared in areas where presence was predicted.

17 The spatial distribution of potential impacts from fishing activity was determined by
18 combining information on bottom contact fishing from two sources. The spatial distribution
19 of offshore physical disturbance by mobile bottom contacting fishing gears was derived from
20 gridded Vessel Monitoring Systems data (VMS) (ICES 2019), acquired from vessels ≥ 12 m in
21 length. The gridded VMS layer was created by calculating the mean value of the average
22 swept area ratios (SAR) for 2009-2017 from annual 5 km resolution 'OSPAR Bottom Fishing
23 Intensity – Surface' data layers, downloaded from the OSPAR Data and Information System

1 (ODIMS, https://odims.ospar.org/odims_data_files/). For inshore waters an additional layer
2 was obtained, representing the number of small (<15 m) fishing vessels using bottom
3 contacting gears operating in coastal areas (Kafas et al. 2014). This layer was produced by
4 adding the number of vessels for scallop dredgers, *Nephrops* trawlers and other bottom
5 trawlers, as reported by the ScotMap Inshore Fisheries Mapping Project (2007-2011;
6 downloaded from the Marine Scotland Data portal, <https://dx.doi.org/10.7489/1554-1>).

7 The fishing intensity layers ('Mean SAR' / 'no. vessels') were intersected with sample points
8 for *F. quadrangularis*. Histograms of presence observations across increasing fishing intensity
9 (both layers) were used to determine cut-off values for fishing intensity where
10 *F. quadrangularis* was less numerous, inferring a negative impact on this species. The cut-off
11 values were applied to each fishing intensity layer and both were combined into one layer
12 delineating the areas where fishing is likely to impact *F. quadrangularis*.

13 The number of absence observations from areas with a predicted high likelihood of presence
14 (>5 CV runs predict presence), low likelihood of presence (1-5 CV runs predict a presence) and
15 absence were examined to compare the fraction of absences in 'impacted' vs. 'unimpacted'
16 areas in both Ecoregions.

17 **3 Results**

18 **3.1 Model performance and transferability**

19 The UKCS models for all three sea pen species showed good performance (Table 3).
20 *F. quadrangularis* was modelled with the highest overall accuracy (TSS = 0.90), followed by
21 *P. phosphorea* (TSS = 0.85), then *V. mirabilis* (TSS = 0.77). Inclusion of Ecoregion as a predictor
22 variable in the UKCS models resulted in accurate predictions in both the CELS and GRNS areas.

1 For *P. phosphorea* and *V. mirabilis* the UKCS model showed fractionally higher sensitivity and
2 overall performance in the Greater North Sea (GRNS) than the Celtic Sea (CELS) (Table 3).
3 *F. quadrangularis* models trained on data from one Ecoregion did not transfer well to the
4 other. The transferred models severely underpredicted presence, with very low sensitivity
5 (0.04 and 0.12 GRNS to CELS and CELS to GRNS, respectively) and consequently poor overall
6 performance in the opposite region (GRNS to CELS TSS = 0.02, CELS to GRNS TSS = 0.04). For
7 *P. phosphorea* and *V. mirabilis* transferability was much better from the CELS model to the
8 Greater North Sea region than from the GRNS model to the Celtic Seas, with an equal or better
9 ability to predict presences and only a marginally poorer specificity (Table 3). The poorer
10 sensitivity and specificity in the *P. phosphorea* and *V. mirabilis* GRNS models transferred to
11 the Celtic Seas, indicating that presence was underpredicted in some areas of this Ecoregion
12 and overpredicted in others, suggesting poor specification of the species niche by the models
13 (Table 3).

14 **3.2 Factors driving sea pen distributions**

15 *F. quadrangularis*, *V. mirabilis* and *P. phosphorea* all display broadly similar environmental
16 requirements when modelled and tested using the full dataset (including both ICES
17 Ecoregions). The most important predictor variables for all three species include mean
18 bottom temperature, wave and current velocities, and the concentration of winter suspended
19 particulate matter in the water column. Primary productivity is also a notable contributor to
20 the *P. phosphorea* and *V. mirabilis* models. The topographic and sediment variables
21 contribute to a lesser degree, but all increase model performance when included. However,
22 the order of importance differs between the Ecoregions. The GRNS models are all strongly
23 driven by temperature, whilst the CELS models on the other hand are more influenced by

1 suspended matter and current and / or wave velocities. Sand and mud content are also more
2 influential in the CELS models than the GRNS models (Figure 2).

3 All species in both Ecoregions are more likely to occur in low current and wave velocities,
4 where concentration of suspended matter in the water column is low. *F. quadrangularis* and
5 *P. phosphorea* show a preference for current and wave velocities up to 0.3 m s^{-1} and 0.5 m s^{-1} ,
6 respectively, and suspended matter at less than 2 g m^{-3} , with 95% of all presences occurring
7 in these conditions. *V. mirabilis* shows a somewhat higher tolerance of wave conditions and
8 turbidity, up to 1.1 m s^{-1} for wave velocity and 5.5 g m^{-3} for suspended matter. The
9 relationship with primary productivity is more complicated. *P. phosphorea* is more likely to
10 occur in areas with low primary productivity, whereas *V. mirabilis* shows a positive association
11 in the CELS model but a negative one in the GRNS model. Responses to bottom temperature
12 and sediment, and in the case of *F. quadrangularis* to bottom topography, differ between the
13 Ecoregions. All species temperature ranges are similar ($7.8\text{-}12.9 \text{ }^{\circ}\text{C}$) but extend into much
14 warmer mean temperatures in the Celtic Seas than in the Greater North Sea. The clearest
15 difference is seen in *F. quadrangularis*, which in the North Sea is most likely to occur between
16 $8\text{-}9^{\circ}\text{C}$, whilst in the Celtic Seas this extends to $\sim 10.5 \text{ }^{\circ}\text{C}$ (Figure 3).

17 *P. phosphorea* and *V. mirabilis* consistently occur in areas with sheltered concave topography
18 with increased 'enclosedness', represented by higher values of valley depth and closed
19 depressions along with low standardised height. *F. quadrangularis* follows these trends in the
20 Celtic Seas, but in the Greater North Sea it occurs almost exclusively offshore, and on flat or
21 slightly elevated ground (Figure 3).

22 Although the effect of mud and sand content is much lower in the GRNS models, the presence
23 probability for all species, and in both Ecoregions, is increased by higher mud content. The

1 CELS models show higher probability of presence for all species with lower sand content,
2 whilst in the GRNS models a slight increase is seen with high sand content. The mud to gravel
3 ratio also shows an opposite trend of influence between the Ecoregions. Whilst presences are
4 more likely with a high mud to gravel ratio in the GRNS, the opposite is true of CELS. The
5 differences are most pronounced for *F. quadrangularis* (Figure 3).

6 The predicted distributions of *F. quadrangularis* across the UKCS by the CELS and GRNS
7 models are almost entirely different (Figure 4). The GRNS model does not predict the
8 presences observed in the warmer inshore bottom temperatures of the CELS, instead
9 predicting presences further offshore in cooler areas with higher sand content. The CELS
10 model, on the other hand, predicts into muddy basins in a wide temperature range, but fails
11 to predict most of the presences in the North Sea that occur in patches of sandier sediments.

12 **3.3 Predicted distributions of sea pens on the UK continental shelf**

13 The UKCS models indicate a largely overlapping distribution, with the main habitat for each
14 species occurring in North Atlantic waters off the coast of Scotland and in the northern half
15 of the North Sea. In terms of spatial extent, the area of predicted distribution was greatest
16 for *P. phosphorea*, followed by *V. mirabilis* and *F. quadrangularis*. *F. quadrangularis* has the
17 most restricted distribution, being confined to the northern waters and, in the south, the shelf
18 edge where canyon heads cut into the shelf break. A small suitable patch is also predicted in
19 the Western English Channel. *P. phosphorea* is also predicted to occur in pockets of suitable
20 habitat in the Western English Channel and southern Celtic Sea towards the shelf edge.
21 *V. mirabilis* habitat has the widest geographical distribution, with predicted patches in all sea
22 areas excluding the Eastern English Channel and Southern North Sea (Figure 5).

23

1 **3.4 The role of bottom contact fishing impacts in the observed distribution of *Funiculina***
2 ***quadrangularis***

3 Comparison of the CELS and GRNS modelled presence distributions and the fishing impact
4 layers revealed spatial trends which related to the Ecoregion model disparities observed for
5 *F. quadrangularis*. *F. quadrangularis* is notably absent from the area predicted as suitable
6 habitat by the CELS model (Figure 6a). The CELS model, therefore, fails to accurately describe
7 the observed distribution of *F. quadrangularis* in the Greater North Sea, despite
8 encompassing the range of environmental conditions existing within it. A large proportion
9 (71%) of the muddier seabed in the Greater North Sea, predicted as suitable *F. quadrangularis*
10 habitat by the CELS model, is impacted by benthic trawling by vessels ≥ 12 m in length, in
11 comparison with only 39% in the Celtic Seas (Table 4). An additional 5% of the Greater North
12 Sea and 15% of the Celtic Seas Ecoregions, is covered by grid cells where inshore fisheries
13 vessels (<15 m in length) are known to operate. Where presences are observed in areas
14 predicted as suitable by the CELS model, they occur outside of the mapped extent of fishing
15 impact (Figure 6a(i)). *F. quadrangularis* are also observed on sandier ground adjacent to the
16 edges of the mud basins predicted as suitable habitat by the CELS model. This effect is
17 exemplified in Figure 6a(ii), which shows the presence of *F. quadrangularis* in patches of
18 sandier and mixed sediments in one of the large mud basins in the Fladen Ground. In contrast,
19 *P. phosphorea*, which has a very similar predicted distribution, does occur widely across the
20 areas predicted as suitable for *F. quadrangularis*, where *F. quadrangularis* is absent
21 (Figure 6b). An example of *F. quadrangularis* on the sandy mixed ground patches in
22 Figure 6a(ii) is shown in Figure 7.

1 Figure 8 shows presences of *F. quadrangularis* across the whole study area are highly skewed
2 towards areas of low fishing impact. Presences are overwhelmingly observed where no fishing
3 occurs according to SAR from VMS data (Figure 8a). Similarly, in the inshore region, the
4 majority of presences are observed in areas where less than four inshore vessels (<15 m) are
5 known to regularly fish using bottom contacting gear (Figure 8b). The majority of absences of
6 *F. quadrangularis* observed where the CELS model indicates a high likelihood of presence are
7 from areas impacted by bottom contact fisheries (70% and 60% in the North Sea and Celtic
8 Seas, although these values are likely to be inflated to some degree due to a greater area of
9 predicted suitable habitat being trawled in the North Sea than the Celtic Seas). The
10 comparable values where the model predicts absence are only 24% and 20% (Figure 8c).

11 **4 Discussion**

12 The Random Forest models were highly effective in predicting occurrences of *Funiculina*
13 *quadrangularis*, *Virgularia mirabilis* and *Pennatula phosphorea* on the UKCS.
14 *F. quadrangularis* was predicted with the highest accuracy, followed by *P. phosphorea* and
15 *V. mirabilis*. Preliminary investigations, however, indicated disparities in the responses of sea
16 pens to environmental parameters between the Celtic Seas and Greater North Sea ICES
17 Ecoregions, particularly for *F. quadrangularis* (the species most sensitive to physical
18 disturbance). This was corroborated by models trained on data from one ICES Ecoregion only,
19 which showed different partial responses to environmental variables in each Ecoregion. A
20 formal assessment testing the accuracy of a model trained on data from one ICES Ecoregion
21 using the independent data from the other Ecoregion confirmed the poor transferability of
22 models between Ecoregions. Further qualitative examination of the predicted and observed
23 distributions, in conjunction with fishing data, revealed that these differences are likely to

1 reflect the modification of sea pen communities by demersal trawling. *F. quadrangularis*
2 appeared to be largely absent from its expected range in the Greater North Sea, whilst *P.*
3 *phosphorea* and *V. mirabilis* appeared less vulnerable to trawling impacts.

4
5 The broadscale geographical distributions predicted by the full UKCS model (using data from
6 both Ecoregions) are largely consistent with sea pen observation data held by the Ocean
7 Biogeographic Information System (OBIS 2020) and the UK National Biodiversity Network
8 (NBN) Atlas (NBN 2020). This is unsurprising given the inclusion of NBN data in the model,
9 however there were some inconsistencies in predicted and observed distributions. All three
10 sea pens (particularly *F. quadrangularis*) are predicted to occur at the shelf break at the
11 extreme south-western tip of the UK EEZ, but are not represented by direct observations in
12 this particular dataset. Fields of *F. quadrangularis* were observed on the shelf break in this
13 region during the Flanders Research Foundation funded MINIMOUNDS project (2013-2016;
14 Howell, unpublished). This provides an example of the efficacy of the model, even where
15 direct observations are absent. The model also predicts all three sea pen species at limited
16 patches in the Western Channel. *P. phosphorea* and *V. mirabilis* are both recorded in this area
17 from NBN and Cefas data, however *F. quadrangularis* are not (also being absent from the OBIS
18 data). It remains unclear whether this absence is due to overprediction by the model (which
19 in this area has been assigned low confidence), a sparse distribution leading to a low
20 probability of encounter, or absence due to human impacts. The different time periods
21 covered by the sea pen observation data (1961 to 2019) and environmental data, such as
22 temperature and primary productivity (2017-2019), add a potential source of error where the
23 conditions may have been different at the time the samples were collected.

24

1 The sea pen distributions observed for all three species are strongly influenced by lower
2 seabed temperatures, low wave and current velocities, and low levels of suspended
3 particulate matter, confirming a requirement for deep, depositional environments.
4 Somewhat surprisingly, sediment composition was determined to be less important. This is
5 likely due to sediment composition being equally driven by the more influential variables (e.g.
6 in a negative correlation between wave and current velocities and fine sediments). All three
7 species did, however, display a strong affinity for muds and muddy sands. Primary
8 productivity, which was influential in the models for *P. phosphorea*, and *V. mirabilis*, is likely
9 to act as a proxy for coastal and enclosed water bodies. The varying responses to primary
10 productivity in the two Ecoregions may result from the inshore / offshore balance of sample
11 locations between the regions. A high number of samples in otherwise suitable habitat in the
12 Celtic Seas Ecoregion are from inlets and lochs with corresponding high primary productivity.
13 On the other hand, the deeper mud basins are associated with lower primary productivity in
14 the Greater North Sea, being located offshore in non-photoc waters.

15

16 Despite the similarities in habitat requirements, the three species displayed some divergence
17 in niche breadth, with *V. mirabilis* and *P. phosphorea* tolerating a wider range of
18 environmental conditions than *F. quadrangularis*, reflected in superior model accuracy for the
19 latter species. *V. mirabilis* and *P. phosphorea* were more tolerant of increased sand fractions,
20 and slightly higher current velocities. *V. mirabilis* showed the greatest tolerance to increased
21 wave velocities and concentration of particulate matter in the water column. This explains its
22 distribution, which extends further inshore in the North Sea than that of *P. phosphorea*, and
23 its presence in the Celtic and Irish Seas, which have higher suspended sediment loads. These
24 findings are supported by those of Greathead et al. (2015), who noted that *V. mirabilis*

1 showed the widest tolerance of a broader range of environmental conditions in Scottish
2 inshore waters, and that *F. quadrangularis* occupied the narrowest niche. It must, however,
3 be noted that we are only discussing the distribution of these species on the continental shelf.
4 *F. quadrangularis* is a cosmopolitan species that, although most commonly reported on the
5 continental shelf and at the top of continental slopes, has been recorded as deep as 2000 m
6 (De Clippele et al. 2015, Lauria et al. 2017, Bastari et al. 2018). Consequently, the models do
7 not capture the full distribution, which extends to deeper, colder water than the UKCS.

8

9 Model transferability between Ecoregions was exceptionally poor for *F. quadrangularis*, with
10 almost no overlap in the observed conditions in which this species was predicted. The Greater
11 North Sea model predicted *F. quadrangularis* in topographically elevated areas of sediment
12 with higher sand content. This result entirely contradicts the Celtic Seas model and the
13 findings of previous Scottish inshore studies (Greathead et al. 2007, 2015), which predicted
14 *F. quadrangularis* in topographically concave muddy basins such as inshore lochs and straits.
15 The apparent 'preference' of *F. quadrangularis* for sandy elevated sediments in the Greater
16 North Sea is therefore thought to reflect artefactual variation in the sea pen dataset. Such a
17 striking disparity in agreement points towards the depletion of *F. quadrangularis* within its
18 natural range in the Greater North Sea (and possibly in the Western Channel, where it has
19 been predicted but not observed), caused by a strongly influential factor which is not
20 accounted for by the environmental layers, and which varies in intensity and spatial
21 distribution between the two Ecoregions.

22

23 Demersal trawling is widely acknowledged as one of the main pressures on marine benthic
24 ecosystems (Halpern et al. 2008, Hiddink et al. 2017, Rijnsdorp et al. 2018) and is well known

1 to modify faunal assemblages and affect species distributions (e.g. Tillin et al. 2006, Hinz et
2 al. 2009). Where prevalent, this can result in sensitive species not occupying their full
3 predicted range or being restricted to suboptimal 'edge of niche' habitats (Braunisch et al.
4 2008). Evidence from the North Sea shows that otter trawlers targeting *Nephrops* and
5 demersal flatfish focus effort in muddy depressions with low bed shear stress (van der Reijden
6 et al. 2018); habitat which is predicted as optimal for sea pens by the Celtic Seas model. The
7 Fladen Grounds is one such area, which has been reported to support the largest *Nephrops*
8 fishery in the world (Ungfors et al. 2013). *F. quadrangularis* were not observed in the muddy
9 depressions of the Fladen Grounds, where gridded VMS data indicate benthic abrasion by
10 trawling (and where the Celtic Seas model predicted their occurrence). This species did,
11 however, occur in patches of interspersed sandier mixed sediment (as confirmed from
12 imagery data; see Figure 7) that do not support *Nephrops* and are therefore unlikely to be
13 targeted by fishers. We therefore hypothesise that *F. quadrangularis*, the most sensitive of
14 the three species (Ager 2003, Greathead et al. 2007), is subject to range modification in the
15 Greater North Sea, with the observed occurrences representing a remnant population
16 occupying the elevated sandy areas with lower disturbance. This finding reflects a significant
17 shift in the niche of *F. quadrangularis* in the Greater North Sea.

18

19 Estimating the accurate spatial range and intensity of demersal fishing activities is inherently
20 challenging, often limiting the extent to which pressure-state relationships can be explored
21 at finer spatial scales (Lee et al. 2010, Lambert et al. 2012). In this study we cannot
22 categorically demonstrate cause and effect, given the coarse resolution of the OSPAR VMS
23 data units in relation to the sea pen occurrence data. However, the overwhelming majority
24 of *F. quadrangularis* were observed in grid cells where the swept area ratio was zero,

1 providing anecdotal support for our hypothesis. This hypothesis is further supported by the
2 findings of Greathead et al. (2005) that the density of *F. quadrangularis* in the Scottish Cuillin
3 Sound (an area of high *Nephrops* trawling intensity), was lower in contrast to the
4 environmentally comparable (but low trawling intensity) Hebridean Sound. Further afield,
5 Pierdomenico et al. (2018) and Fabri et al. (2014) both observed low abundances of
6 *F. quadrangularis* in chronically trawled Mediterranean canyons, whilst Malecha & Stone
7 (2009) found that experimental breakage of the morphologically similar *Halipteris willemoesi*
8 resulted in extremely high mortality rates. Although the available evidence suggests that
9 *F. quadrangularis* is vulnerable to trawling impacts, there is no such indication that creel
10 fishing for *Nephrops* has a negative impact on its distribution. Eno et al. (2001) observed that
11 *F. quadrangularis* was able to re-insert itself into the sediment after uprooting by creel pots.
12 The theory that *F. quadrangularis* is more resilient to creeling impacts is supported by our
13 study, as *F. quadrangularis* was shown to proliferate in Scottish sea lochs where smaller
14 fishing vessels operate; in particular Loch Linnhe, where the *Nephrops* fishery is
15 predominantly creel-based (Berx et al. 2015).

16

17 Whilst the high sensitivity of *F. quadrangularis* to trawling appears to severely diminish the
18 transferability of the Celtic Seas and Greater North Sea models, this effect was not as
19 pronounced for *V. mirabilis* and *P. phosphorea*. Actual presences in the North Sea were
20 captured effectively by the Celtic Seas model, although this model tended to overpredict.
21 Conversely the Greater North Sea model underpredicted presences in the Celtic Seas. This
22 trend suggests that the distributions of these species are more limited in the Greater North
23 Sea and perhaps do not cover the entire natural range of environmental variability covered
24 by observations in the Celtic Seas. *V. mirabilis* and *P. phosphorea* both display far broader

1 geographical ranges than *F. quadrangularis*, being frequent in areas of demersal fishing
2 activity, both in the Celtic Seas and Greater North Sea. There is, therefore, no clear evidence
3 that trawling limits their distribution on the UKCS, corroborating the findings of previous
4 studies that *V. mirabilis* and *P. phosphorea* are less vulnerable to fishing impacts than
5 *F. quadrangularis* (Howson & Davies 1991, Tuck et al. 1998, Greathead et al. 2005). This is
6 likely due to a combination of wider niche breadth, higher abundance, physiological traits
7 (e.g. flexibility) and the ability to withdraw into a burrow (Ambroso et al. 2013, Chimienti et
8 al. 2018).

9

10 Previous publications have proposed that sea pen presence and/or abundance could serve as
11 indicators of condition in mud habitats (Greathead et al. 2007, Murray et al. 2015). This study
12 suggests that the presence of *F. quadrangularis* could potentially act as an indicator of
13 healthy, undisturbed conditions, although the absence of this species in suitable habitat
14 should not automatically be assumed to indicate poor habitat condition, as sea pens are
15 known to display patchy or aggregated distributions (Langton et al. 1990, Greathead et al.
16 2007). Although the predicted distributions of *V. mirabilis* and *P. phosphorea* did not appear
17 to be adversely affected by demersal fishing, it should be noted that these models only reflect
18 presence or absence of sea pens, offering no insight on density or condition. In their 2015
19 study of the Fladen Grounds, Murray et al. found that the density of *P. phosphorea* decreased
20 along a gradient of increasing trawling pressure, suggesting that density may have potential
21 as an indicator at higher levels of fishing intensity. We suggest that further studies are
22 conducted to explore the relationships between trawling activity and density of
23 *P. phosphorea* and *V. mirabilis*. A greater understanding of these relationships would allow

1 evaluation of sea pen management measures, in turn enabling ecologically sound
2 management decisions.

3

4 A number of UK MPAs have been designated to protect sea pens, however, management
5 measures within MPAs are spatially variable and trawling is not necessarily under active
6 management in areas of suitable habitat. The vast majority of *F. quadrangularis* observations
7 and areas of suitable habitat identified in this study are not currently covered by MPAs
8 designated to protect sea pen communities (particularly in the Greater North Sea; although
9 they are found within those managed for different habitat types and could benefit from
10 management measures in these areas). The predicted distribution models presented in this
11 study (alongside further pressure-state studies on *P. phosphorea* and *V. mirabilis*) could be
12 used to refine management of specific MPA zones and gear types, thus optimising the balance
13 between access to commercially important fishing grounds and protection of sea pen
14 communities.

15

16 This study has demonstrated the effectiveness of the Random Forest method for modelling
17 vulnerable species distributions and highlighted the utility of national (and international) data
18 platforms for addressing conservation questions at large scales. Whilst challenges still remain
19 in reconciling the spatial and temporal disparities of sample data, environmental parameters
20 and human pressures, our findings illustrate the value of broadscale qualitative comparisons
21 between SDMs and human activity data for understanding fundamental pressure-state
22 relationships. Benthic trawling is generally accepted as the greatest global threat to marine
23 benthic fauna, yet the increasing accessibility of marine data creates opportunities for
24 informed and targeted mitigation. We anticipate that the approach used in this study could

1 be widely applied to investigate trawling impacts on sensitive species at large scales, enabling
2 marine spatial planners to make better management decisions and deliver tangible
3 conservation outcomes.

4

5 **Acknowledgements**

6 We would like to thank Peter Mitchell (Cefas) for his advice on accessing and interpreting the
7 environmental data layers obtained through the Cefas Data Hub, David Clare and Michaela
8 Schratzberger (Cefas), Karema Randall and Will Hutchinson (Defra) for providing valuable
9 feedback on drafts of this article, and the Joint Nature Conservation Committee (JNCC) for
10 supplying sea pen data and network information for Scottish MPAs.

11

12 **References**

13 Ager OED (2003) *Funiculina quadrangularis* The tall sea pen. In Tyler-Walters H. and Hiscock
14 K. (eds) Marine Life Information Network: Biology and Sensitivity Key Information Reviews,
15 Plymouth: Marine Biological Association of the United Kingdom.

16 www.marlin.ac.uk/species/detail/1154 (accessed 21 Oct 2020).

17

18 Aguilar R, Perry AL, López J (2017) Conservation and Management of Vulnerable Marine
19 Benthic Ecosystems. In: Rossi S, Bramanti L, Gori A, Orejas C (eds) Marine Animal Forests.
20 Springer, Cham. https://doi.org/10.1007/978-3-319-17001-5_34-1.

21

22 Allouche O, Tsoar A, Kadmon R (2006) Assessing the accuracy of species distribution models:
23 prevalence, kappa and the true skill statistic (TSS). *J Appl Ecol* 43:1223–1232.

24 <https://doi.org/10.1111/j.1365-2664.2006.01214.x>.

25

26 Ambroso S, Dominguez-Carrió C, Grinyó J, López-González PJ, Gili JM, Purroy A, Requena S,
27 Madurell T (2013) In situ observations on withdrawal behaviour of the sea pen *Virgularia*
28 *mirabilis*. *Mar Biodivers* 43:257–258. <https://doi.org/10.1007/s12526-013-0172-5>.

1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
24
25
26
27
28
29
30
31

Baillon S (2012) Deep cold-water corals as nurseries for fish larvae. *Front Ecol Environ* 10:351–356. <https://doi.org/10.1890/120022>.

Bastari A, Pica D, Ferretti F, Micheli F, Cerrano C (2018) Sea pens in the Mediterranean Sea: habitat suitability and opportunities for ecosystem recovery. *ICES J Mar Sci* 75:1722–1732. <https://doi.org/10.1093/icesjms/fsy010>.

Beazley L, Kenchington E, Murillo FJ, Lirette C, Guijarro J, McMillan A, Knudby A (2016) Species distribution modelling of corals and sponges in the Maritimes region for use in the identification of significant benthic areas. *Canadian Technical Report of Fisheries and Aquatic Sciences* 3172. <https://doi.org/10.17632/MCB726KCBX.1>.

Berx B, Gallego A, Heath M, Lyndon, A (2015) Loch Linnhe and Firth of Lorn MASTS Case Study Workshop Report. *Scottish Mar Freshw Sci* 6 (1). doi: 10.7489/1539-1.

Braunisch V, Bollmann K, Graf RF, Hirzel AH (2008) Living on the edge—modelling habitat suitability for species at the edge of their fundamental niche. *Ecol Modell* 214:153–167. <https://doi.org/10.1016/j.ecolmodel.2008.02.001>.

Breiman L (2001) Random Forests. *Mach Learn* 45:5–32. <https://doi.org/10.1023/A:1010933404324>.

Buhl-Mortensen P, Buhl-Mortensen L (2014) Diverse and vulnerable deep-water biotopes in the Hardangerfjord. *Mar Biol Res* 10:253–267. <https://doi.org/10.1080/17451000.2013.810759>.

Chimienti G, Angeletti L, Mastrototaro F (2018) Withdrawal behaviour of the red sea pen *Pennatula rubra* (Cnidaria: Pennatulacea). *Eur Zool J* 85:64–70. <https://doi.org/10.1080/24750263.2018.1438530>.

1 De Clippele LH, Buhl-Mortensen P, Buhl-Mortensen L (2015) Fauna associated with cold
2 water gorgonians and sea pens. *Cont Shelf Res* 105:67–78.
3 <https://doi.org/10.1016/j.csr.2015.06.007>.

4

5 Conrad O, Bechtel B, Bock M, Dietrich H, Fischer E, Gerlitz L, Wehberg J, Wichmann V,
6 Böhner J (2015) System for Automated Geoscientific Analyses (SAGA) v. 2.1.4. *Geosci Model*
7 *Dev* 8:1991–2007. <https://doi.org/10.5194/gmd-8-1991-2015>.

8

9 Cutler DR, Edwards TC, Beard KH, Cutler A, Hess KT, Gibson J, Lawler JJ (2007) Random
10 Forests for classification in ecology. *Ecology* 88:2783–2792. [https://doi.org/10.1890/07-](https://doi.org/10.1890/07-0539.1)
11 [0539.1](https://doi.org/10.1890/07-0539.1).

12

13 Eigaard OR, Bastardie F, Breen M, Dinesen GE, Hintzen NT, Laffargue P, Mortensen LO,
14 Nielsen JR, Nilsson HC, O’Neill FG, Polet H, Reid DG, Sala A, Sköld M, Smith C, Sørensen TK,
15 Tully O, Zengin M, Rijnsdorp AD (2016) Estimating seabed pressure from demersal trawls,
16 seines, and dredges based on gear design and dimensions. *ICES J Mar Sci* 73:i27–i43.
17 <https://doi.org/10.1093/icesjms/fsv099>.

18

19 Eno N (2001) Effects of crustacean traps on benthic fauna. *ICES J Mar Sci* 58:11–20.
20 <https://doi.org/10.1006/jmsc.2000.0984>.

21

22 Fabri M-C, Pedel L, Beuck L, Galgani F, Hebbeln D, Freiwald A (2014) Megafauna of
23 vulnerable marine ecosystems in French Mediterranean submarine canyons: Spatial
24 distribution and anthropogenic impacts. *Deep Sea Res Part II Top Stud Oceanogr* 104:184–
25 207. <https://doi.org/10.1016/j.dsr2.2013.06.016>.

26

27 Fielding AH, Bell JF (1997) A review of methods for the assessment of prediction errors in
28 conservation presence / absence models. *Environ Conserv* 24:38–49.
29 <https://doi.org/10.1017/S0376892997000088>.

30

31 Freeman EA, Moisen G (2008) PresenceAbsence : An R Package for Presence Absence
32 Analysis. *J Stat Softw* 23:1–31. <http://dx.doi.org/10.18637/jss.v023.i11>.

1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
24
25
26
27
28
29
30
31

Gale KSP, Hamel J-F, Mercier A (2013) Trophic ecology of deep-sea Asteroidea (Echinodermata) from eastern Canada. *Deep Sea Res Part I Oceanogr Res Pap* 80:25–36. <https://doi.org/10.1016/j.dsr.2013.05.016>.

García-Matucheski S, Muniain C (2011) Predation by the nudibranch *Tritonia odhneri* (Opisthobranchia: Tritoniidae) on octocorals from the South Atlantic Ocean. *Mar Biodivers* 41:287–297. <https://doi.org/10.1007/s12526-010-0063-y>.

Greathead CF, Donnan DW, Mair JM (2005) Impact of *Nephrops* trawling on the distribution of the sea pens *Virgularia mirabilis*, *Pennatula phosphorea* and *Funiculina quadrangularis* in Scottish waters. Fisheries Research Services Internal Report No 02/05. Aberdeen, UK. <https://www2.gov.scot/Uploads/Documents/IR0205.pdf> (accessed 21 Oct 2020).

Greathead CF, Donnan DW, Mair JM, Saunders GR (2007) The sea pens *Virgularia mirabilis*, *Pennatula phosphorea* and *Funiculina quadrangularis*: distribution and conservation issues in Scottish waters. *J Mar Biol Assoc United Kingdom* 87:1095–1103. <https://doi.org/10.1093/icesjms/fsu129>.

Greathead C, González-Irusta JM, Clarke J, Boulcott P, Blackadder L, Weetman A, Wright PJ (2015) Environmental requirements for three sea pen species: relevance to distribution and conservation. *ICES J Mar Sci* 72:576–586. <https://doi.org/10.1093/icesjms/fsu129>.

Grip, K (2017) International marine environmental governance: a review. *Ambio* 46, 413–427. <https://doi.org/10.1007/s13280-016-0847-9>

Halpern BS, Walbridge S, Selkoe KA, Kappel C V., Micheli F, D’Agrosa C, Bruno JF, Casey KS, Ebert C, Fox HE, Fujita R, Heinemann D, Lenihan HS, Madin EMP, Perry MT, Selig ER, Spalding M, Steneck R, Watson R (2008) A Global Map of Human Impact on Marine Ecosystems. *Science* 319 (5865) 948-952. <https://doi.org/10.1126%2Fscience.1149345>.

1 He KS, Bradley BA, Cord AF, Rocchini D, Tuanmu M-N, Schmidtlein S, Turner W, Wegmann
2 M, Pettorelli N (2015) Will remote sensing shape the next generation of species distribution
3 models? *Remote Sens Ecol Conserv* 1:4–18. <https://doi.org/10.1002/rse2.7>.
4

5 Hiddink JG, Jennings S, Kaiser MJ (2006) Indicators of the ecological impact of bottom-trawl
6 disturbance on seabed communities. *Ecosystems* 9:1190–1199.
7 <https://doi.org/10.1007/s10021-005-0164-9>.
8

9 Hiddink JG, Jennings S, Sciberras M, Szostek CL, Hughes KM, Ellis N, Rijnsdorp AD,
10 McConnaughey RA, Mazor T, Hilborn R, Collie JS, Pitcher CR, Amoroso RO, Parma AM,
11 Suuronen P, Kaiser MJ (2017) Global analysis of depletion and recovery of seabed biota after
12 bottom trawling disturbance. *Proc Natl Acad Sci* 114:8301–8306.
13 <https://doi.org/10.1073/pnas.1618858114>.
14

15 Hill JM, Tyler-Walters H, Garrard, S (2020) Seapens and burrowing megafauna in circalittoral
16 fine mud. In Tyler-Walters H. and Hiscock K. (eds) *Marine Life Information Network: Biology
17 and Sensitivity Key Information Reviews*. Marine Biological Association of the United
18 Kingdom, Plymouth, UK. www.marlin.ac.uk/habitats/detail/131 (accessed 21 Oct 2020).
19

20 Hill JM, Wilson E (2000) *Virgularia mirabilis* Slender sea pen. In Tyler-Walters H. and Hiscock
21 K. (eds) *Marine Life Information Network: Biology and Sensitivity Key Information Reviews*.
22 Marine Biological Association of the United Kingdom, Plymouth, UK.
23 www.marlin.ac.uk/species/detail/1396 (accessed 21 Oct 2020).
24

25 Hinz H, Prieto V, Kaiser M (2009) Trawl disturbance on benthic communities: chronic effects
26 and experimental predictions. *Ecol Appl* 19:761–773.
27 <https://doi.org/10.1073/pnas.1618858114>.
28

29 Hixon MA, Tissot BN (2007) Comparison of trawled vs untrawled mud seafloor assemblages
30 of fishes and macroinvertebrates at Coquille Bank, Oregon. *J Exp Mar Bio Ecol* 344:23–34.
31 <https://doi.org/10.1016/j.jembe.2006.12.026>.
32

1 Howson CM, Connor DW, Holt RHF (1994) The Scottish Sea lochs—an account of surveys
2 undertaken for the MNCR. Joint Nature Conservation Committee Report No 164 (Marine
3 Nature Conservation Review Report No. MNCR/SR/27).
4

5 Howson CM, Davies LM (1991) Marine Nature Conservation Review, surveys of Scottish
6 sealochs. A towed video survey of Loch Fyne. Volume 1 - Report. Report to the Nature
7 Conservancy Council from the University Marine Biological Station, Millport, Scotland.
8

9 ICES (2005) ICES Statistical Areas.
10 https://gis.ices.dk/gis/rest/services/ICES_reference_layers/ICES_Areas/MapServer
11 (accessed 21 Oct 2020).
12

13 ICES (2015) ICES Ecoregions. [https://gis.ices.dk/geonetwork/srv/api/records/4745e824-
14 a612-4a1f-bc56-b540772166eb](https://gis.ices.dk/geonetwork/srv/api/records/4745e824-a612-4a1f-bc56-b540772166eb) (accessed 21 Oct 2020).
15

16 ICES (2019) Spatial distribution of fishing effort and physical disturbance of benthic habitats
17 by mobile bottom trawl fishing gear using VMS; Technical Guidelines. In Report of the ICES
18 Advisory Committee. ICES Advice 2019. <https://doi.org/10.17895/ices.advice.4683>.
19

20 ICES (2020) Definition and rationale for ICES ecoregions. In Report of the ICES Advisory
21 Committee, 2020. ICES Advice 2020. <https://doi.org/10.17895/ices.advice.6014>.
22

23 Jiménez-Valverde A (2012) Insights into the area under the receiver operating characteristic
24 curve (AUC) as a discrimination measure in species distribution modelling. *Glob Ecol*
25 *Biogeogr* 21:498–507. <https://doi.org/10.1111/j.1466-8238.2011.00683.x>.
26

27 Jones H (2008) *Pennatula phosphorea* Phosphorescent sea pen. In Tyler-Walters H, Hiscock
28 K (eds) Marine Life Information Network: Biology and Sensitivity Key Information Reviews.
29 Marine Biological Association of the United Kingdom, Plymouth, UK.
30 www.marlin.ac.uk/species/detail/1817 (accessed 21 Oct 2020).
31

1 Kafas A, McLay A, Chimienti M, Gubbins M (2014) ScotMap inshore fisheries mapping in
2 Scotland: recording fishermen's use of the sea. Scottish Mar Freshw Sci Vol 5, No 17.
3 Scottish Government, Edinburgh. doi: 10.4789/1554-1.
4

5 Kenchington E, Murillo FJ, Lirette C, Sacau M, Koen-Alonso M, Kenny A, Ollerhead N,
6 Wareham V, Beazley L (2014) Kernel density surface modelling as a means to identify
7 significant concentrations of Vulnerable Marine Ecosystem indicators. PLoS One 9 (10)
8 e109365. <https://doi.org/10.1371/journal.pone.0109365>
9

10 Kinlan BP, Poti M, Drohan AF, Packer DB, Dorfman DS, Nizinski MS (2020) Predictive
11 modeling of suitable habitat for deep-sea corals offshore the Northeast United States. Deep
12 Sea Res Part I Oceanogr Res Pap 158:103229. <https://doi.org/10.1016/j.dsr.2020.103229>.
13

14 Knudby A, Lirette C, Kenchington E, Murillo FJ (2013) Species distribution models of black
15 corals, large gorgonian corals and sea pens in the NAFO Regulatory Area. Northwest Atlantic
16 Fisheries Organization. SC Ecosystem Science and Assessment Working Group (WGESA).
17 NAFO SCR Doc. 13/078. Serial No. N6276.
18

19 Krigsman LM, Yoklavich MM, Dick EJ, Cochrane GR (2012) Models and maps: predicting the
20 distribution of corals and other benthic macro-invertebrates in shelf habitats. Ecosphere
21 3:1–16. <https://doi.org/10.1890/ES11-00295.1>.
22

23 Kröger S, Parker R, Cripps G, Williamson P (2018) Shelf Seas: The Engine of Productivity,
24 Policy Report on NERC-Defra Shelf Sea Biogeochemistry Programme. Cefas, Lowestoft, UK.
25 https://www.uk-ssb.org/shelf_seas_report.html (accessed 21 Oct 2020).
26

27 Kursá MB, Rudnicki WR (2010) Feature selection with the boruta package. J Stat Softw 36
28 (11) 1-13 <http://dx.doi.org/10.18637/jss.v036.i11>.
29

30 Lambert GI, Jennings S, Hiddink JG, Hintzen NT, Hinz H, Kaiser MJ, Murray LG (2012)
31 Implications of using alternative methods of vessel monitoring system (VMS) data analysis

1 to describe fishing activities and impacts. ICES J Mar Sci 69:682–693.
2 <https://doi.org/10.1093/icesjms/fss018>.
3
4 Langton RW, Langton EW, Theroux RB, Uzmann JR (1990) Distribution, behavior and
5 abundance of sea pens, *Pennatula aculeata*, in the Gulf of Maine. Mar Biol 107:463–469.
6 <https://doi.org/10.1007/BF01313430>.
7
8 Lauria V, Garofalo G, Fiorentino F, Massi D, Milisenda G, Piraino S, Russo T, Gristina M
9 (2017) Species distribution models of two critically endangered deep-sea octocorals reveal
10 fishing impacts on vulnerable marine ecosystems in central Mediterranean Sea. Sci Rep 7 (1)
11 8049. <https://doi.org/10.1038/s41598-017-08386-z>.
12
13 Lee J, South AB, Jennings S (2010) Developing reliable, repeatable, and accessible methods
14 to provide high-resolution estimates of fishing-effort distributions from vessel monitoring
15 system (VMS) data. ICES J Mar Sci 67 (6) 1260-1271.
16 <https://doi.org/10.1093/icesjms/fsq010>.
17
18 Lobo JM, Jiménez-Valverde A, Real R (2008) AUC: A misleading measure of the performance
19 of predictive distribution models. Glob Ecol Biogeogr 17:145–151.
20 <https://doi.org/10.1111/j.1466-8238.2007.00358.x>.
21
22 Malecha P, Stone R (2009) Response of the sea whip *Halipteris willemoesi* to simulated trawl
23 disturbance and its vulnerability to subsequent predation. Mar Ecol Prog Ser 388:197–206.
24 <https://doi.org/10.3354/meps08145>.
25
26 Manel S, Ceri Williams H, Ormerod SJ (2001) Evaluating presence/absence models in
27 ecology: the need to account for prevalence. J Appl Ecol 38:921–931.
28 <https://doi.org/10.1046/j.1365-2664.2001.00647.x>.
29
30 Marine Management Organisation (2019) UK sea fisheries statistics 2018.
31 <https://www.gov.uk/government/statistics/uk-sea-fisheries-annual-statistics-report-2018>
32 (accessed 21 Oct 2020).

1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
24
25
26
27
28
29
30
31

Marshall CE, Glegg GA, Howell KL (2014) Species distribution modelling to support marine conservation planning: the next steps. *Mar Policy* 45:330–332.

<https://doi.org/10.1016/j.marpol.2013.09.003>.

Mitchell P, Aldridge J, Diesing M (2019a) Predictor variables and groundtruth samples for north-west European continental shelf quantitative sediment analysis. Cefas, UK. V1.

<https://doi.org/10.14466/CefasDataHub.62>.

Mitchell P, Aldridge J, Diesing M (2019b) Quantitative sediment composition predictions for the north-west European continental shelf. Cefas, UK. V1.

<https://doi.org/10.14466/CefasDataHub.63>.

Mitchell PJ, Aldridge J, Diesing M (2019c) Legacy data: How decades of seabed sampling can produce robust predictions and versatile products. *Geosciences* 9:182.

<https://doi.org/10.3390/geosciences9040182>.

Murillo FJ, Kenchington E, Beazley L, Lirette C, Knudby A, Guijarro J, Benoît H, Bourdages H, Sainte-Marie B (2016) Distribution modelling of sea pens, sponges, stalked tunicates and soft corals from research vessel survey data in the Gulf of St. Lawrence for use in the identification of significant benthic areas. *Canadian Technical Report of Fisheries and Aquatic Sciences* 3170. doi: [10.13140/RG.2.1.2966.1689](https://doi.org/10.13140/RG.2.1.2966.1689).

Murray J, Jenkins C, Eggleton J, Whomersley P, Robson L, Flavell B, Hinchin H (2015) The development of monitoring options for UK MPAs: Fladen Grounds R&D case study. Joint Nature Conservation Committee/Cefas Partnership Report Series No. 9. Peterborough, UK. ISSN 2051-6711. <https://hub.jncc.gov.uk/assets/2d594d86-06f1-419d-8cb6-db2f61b5be9c>.

NBN (2020) National Biodiversity Network Atlas. <https://www.nbnatlas.org> (accessed 21 Oct 2020).

1 OBIS (2020) Global map of species distribution using gridded data. Ocean Biogeographic
2 Information System. www.obis.org (accessed 21 Oct 2020).
3
4 Pierdomenico M, Russo T, Ambroso S, Gori A, Martorelli E, D'Andrea L, Gili J-M, Chiocci FL
5 (2018) Effects of trawling activity on the bamboo-coral *Isidella elongata* and the sea pen
6 *Funiculina quadrangularis* along the Gioia Canyon (Western Mediterranean, southern
7 Tyrrhenian Sea). Prog Oceanogr 169:214–226.
8 <https://doi.org/10.1016/j.pocean.2018.02.019>.
9
10 Prasad AM, Iverson LR, Liaw A (2006) Newer classification and regression tree techniques:
11 bagging and Random Forests for ecological prediction. Ecosystems 9:181-199.
12 <https://doi.org/10.1007/s10021-005-0054-1>.
13
14 R Core Team (2018) R: A language and environment for statistical computing. R Foundation
15 for Statistical Computing, Vienna, Austria. <https://www.r-project.org> (accessed 21 Oct
16 2020).
17
18 van der Reijden KJ, Hintzen NT, Govers LL, Rijnsdorp AD, Olff H (2018) North Sea demersal
19 fisheries prefer specific benthic habitats. PLoS One 13:e0208338.
20 <https://doi.org/10.1371/journal.pone.0208338>.
21
22 Reiss H, Birchenough S, Borja A, Buhl-Mortensen L, Craeymeersch J, Dannheim J, Darr A,
23 Galparsoro I, Gogina M, Neumann H, Populus J, Rengstorf AM, Valle M, Hoey G Van, Zettler
24 ML, Degraer S (2015) Benthos distribution modelling and its relevance for marine ecosystem
25 management. ICES J Mar Sci 72:297–315. <https://doi.org/10.1093/icesjms/fsu107>.
26
27 Rijnsdorp AD, Bastardie F, Bolam SG, Buhl-Mortensen L, Eigaard OR, Hamon KG, Hiddink JG,
28 Hintzen NT, Ivanović A, Kenny A, Laffargue P, Nielsen JR, O'Neill FG, Piet GJ, Polet H, Sala A,
29 Smith C, van Denderen PD, van Kooten T, Zengin M (2016) Towards a framework for the
30 quantitative assessment of trawling impact on the seabed and benthic ecosystem. ICES J
31 Mar Sci 73:i127–i138. <https://doi.org/10.1093/icesjms/fsv207>.
32

1 Rijnsdorp AD, Bolam SG, Garcia C, Hiddink JG, Hintzen NT, van Denderen PD, van Kooten T
2 (2018) Estimating sensitivity of seabed habitats to disturbance by bottom trawling based on
3 the longevity of benthic fauna. *Ecol Appl* 28:1302–1312. <https://doi.org/10.1002/eap.1731>.
4

5 Rogers AD, Gianni M (2010) The Implementation of the UNGA Resolutions 61/105 and
6 64/72 in the Management of Deep-Sea Fisheries on the High Seas. Report prepared for the
7 Deep-Sea Conservation Coalition, International Programme on the State of the Ocean,
8 London. <http://www.savethehighseas.org/publicdocs/61105-Implementation-finalreport.pdf>
9 (accessed 21 Oct 2020).
10

11 Santika T (2011) Assessing the effect of prevalence on the predictive performance of species
12 distribution models using simulated data. *Glob Ecol Biogeogr* 20:181–192.
13 <https://doi.org/10.1111/j.1466-8238.2010.00581.x>.
14

15 Shaw RL, Booth A, Sutton AJ, Miller T, Smith JA, Young B, Jones DR, Dixon-Woods M (2004)
16 Finding qualitative research: an evaluation of search strategies. *BMC Med Res Methodol* 4:5.
17 <https://doi.org/10.1186/1471-2288-4-5>.
18

19 Smith RJ, Eastwood PD, Ota Y, Rogers SI (2009) Developing best practice for using Marxan to
20 locate Marine Protected Areas in European waters. *ICES J Mar Sci* 66:188–194.
21 <https://doi.org/10.1093/icesjms/fsn198>.
22

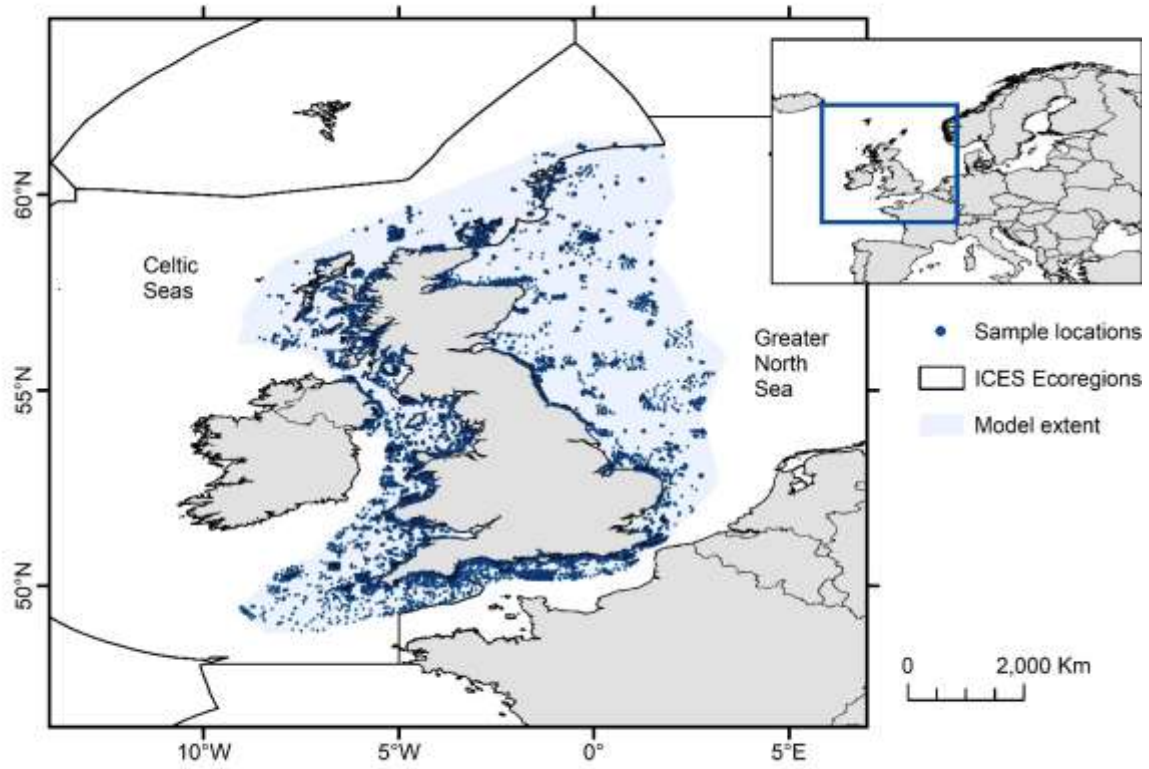
23 Sundblad G, Bergström U, Sandström A (2011) Ecological coherence of marine protected
24 area networks: a spatial assessment using species distribution models. *J Appl Ecol* 48:112–
25 120. <https://doi.org/10.1111/j.1365-2664.2010.01892.x>.
26

27 Thuiller W, Georges D, Engler R, Breiner F (2020) Biomod2: Ensemble Platform for Species
28 Distribution Modeling. R package version 3.4.6. [https://cran.r-](https://cran.r-project.org/web/packages/biomod2/biomod2.pdf)
29 [project.org/web/packages/biomod2/biomod2.pdf](https://cran.r-project.org/web/packages/biomod2/biomod2.pdf) (accessed 21 Oct 2020).
30

1 Thuiller W, Lafourcade B, Engler R, Araújo MB (2009) BIOMOD - a platform for ensemble
2 forecasting of species distributions. *Ecography* 32:369–373. [https://doi.org/10.1111/j.1600-](https://doi.org/10.1111/j.1600-0587.2008.05742.x)
3 [0587.2008.05742.x](https://doi.org/10.1111/j.1600-0587.2008.05742.x).
4
5 Thurstan RH, Brockington S, Roberts CM (2010) The effects of 118 years of industrial fishing
6 on UK bottom trawl fisheries. *Nat Commun* 1:15. <https://doi.org/10.1038/ncomms1013>.
7
8 Tillin H, Hiddink J, Jennings S, Kaiser M (2006) Chronic bottom trawling alters the functional
9 composition of benthic invertebrate communities on a sea-basin scale. *Mar Ecol Prog Ser*
10 318:31–45. <https://doi.org/10.3354/meps318031>.
11
12 Tuck I, Hall S, Robertson M, Armstrong E, Basford D (1998) Effects of physical trawling
13 disturbance in a previously unfished sheltered Scottish sea loch. *Mar Ecol Prog Ser* 162:227–
14 242. doi:10.3354/meps162227.
15
16 Tyberghein L, Verbruggen H, Pauly K, Troupin C, Mineur F, De Clerck O (2012) Bio-ORACLE: a
17 global environmental dataset for marine species distribution modelling. *Glob Ecol Biogeogr*
18 21:272–281. <https://doi.org/10.1111/j.1466-8238.2011.00656.x>.
19
20 Ungfors A, Bell E, Johnson ML, Cowing D, Dobson NC, Bublitz R, Sandell J (2013) *Nephrops*
21 fisheries in European waters. *Adv Mar Biol* 64:247–314. [https://doi.org/10.1016/b978-0-12-](https://doi.org/10.1016/b978-0-12-410466-2.00007-8)
22 [410466-2.00007-8](https://doi.org/10.1016/b978-0-12-410466-2.00007-8).
23
24 Wilson MT, Andrews AH, Brown AL, Cordes EE (2002) Axial rod growth and age estimation of
25 the sea pen, *Halipteris willemoesi* Kölliker. *Hydrobiologia* 471:133–142.
26 <https://doi.org/10.1023/A:1016509506094>.
27
28
29
30

1 **Figures**

- 2 Figure 1. Map detailing the study area, extent of distribution models, boundaries of the ICES
3 Ecoregions used to delineate the two target regions and the distribution of sea pen
4 presence / absence point observations.



5

Figure 2. Predictor variable contributions to models by species and regional model. PP = Primary productivity, UKCS = UK continental shelf, CELS = Celtic Seas and GRNS = Greater North Sea. Variable contribution is inferred from a reduction in model accuracy when each predictor variable in turn is randomised. Predictor Importance = 1 minus the correlation coefficient between model accuracies for the full model and the model with the randomised variable.

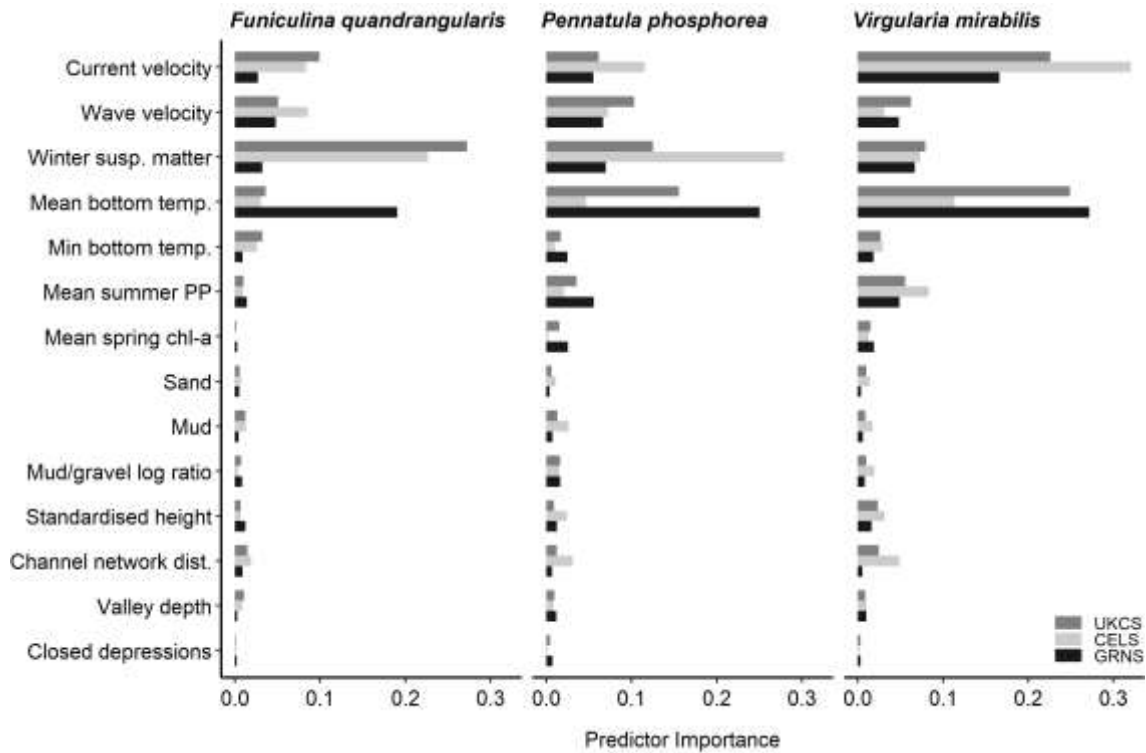


Figure 3. Random Forest model partial response curves illustrating the different response of *Funiculina quadrangularis* presence / absence to substrate and topography variables observed between the Celtic Seas (CELS) and Greater North Sea (GRNS) Ecoregions. The plots show the relative logit contribution of each variable to the probability of presence. The y-axis scale has been log-transformed to allow plotting of curves with a large range of values from both models onto the same axis.

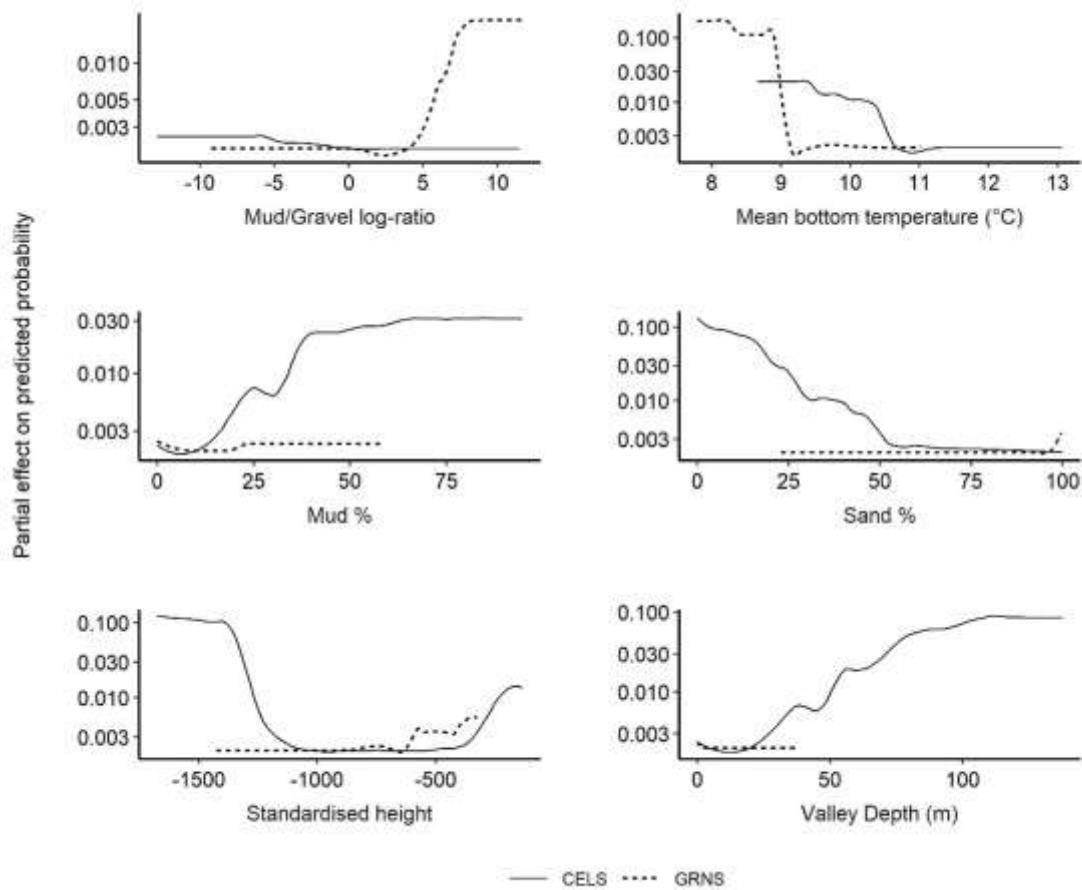


Figure 4. Distribution of *Funiculina quadrangularis* predicted by the CELS (a) and GRNS (b) models, with observed presences. Confidence in the predicted distribution is represented by two categories. Low likelihood of presence is shown where <5 and High likelihood where ≥ 5 of 10 cross-validation model iterations predicted presence.

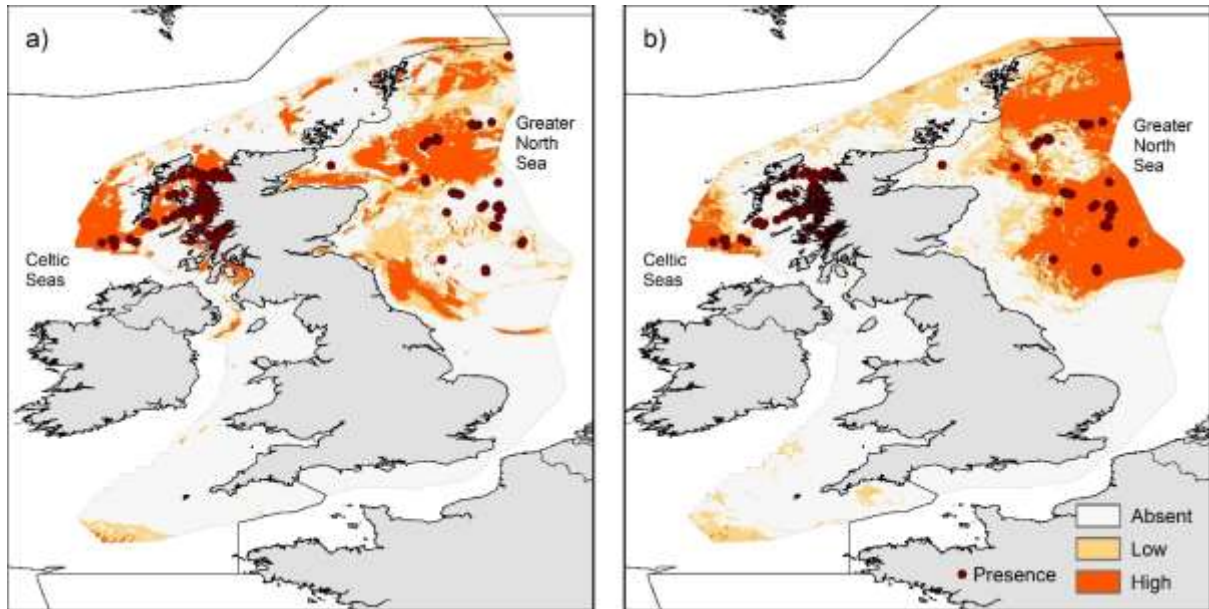


Figure 5. Distributions of (a) *Funiculina quadrangularis*, (b) *Pennatula phosphorea* and (c) *Virgularia mirabilis* predicted by the full UKCS model. Confidence in the predicted distribution is represented by two categories. Low likelihood of presence is shown where <5 and High likelihood where ≥ 5 of 10 cross-validation model iterations predicted presence.

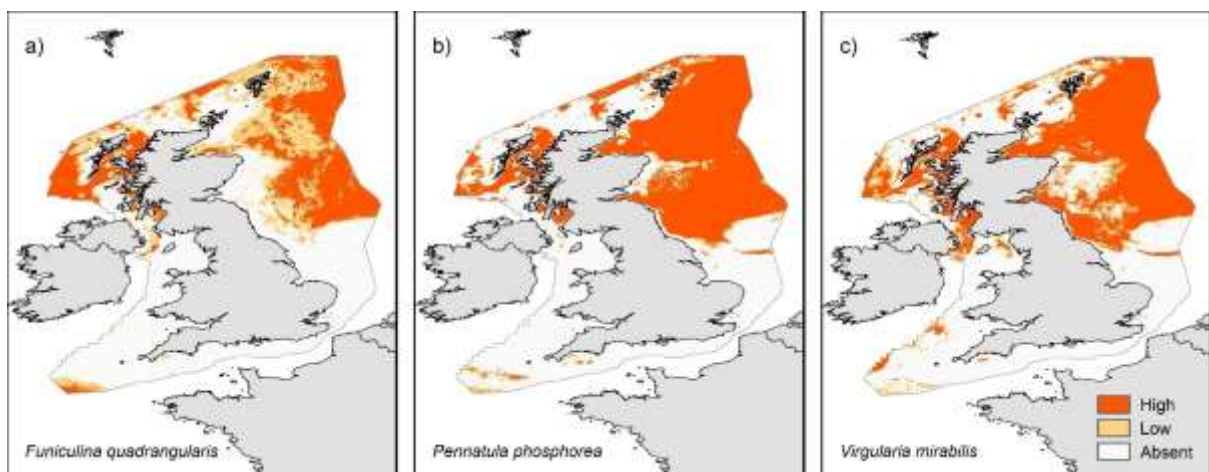


Figure 6. Distribution of *Funiculina quadrangularis* predicted by the CELS model, overlain with the extent of bottom contact fisheries (SAR >0 or >4 vessels reported fishing using bottom contacting gears) and the observed presence / absence of *F. quadrangularis* (a) and *Pennatula phosphorea* (b). Inset (i) in panel (a) shows close-up detail of *F. quadrangularis* presence in areas predicted by the CELS model in the absence of fishing pressure. Inset (ii) in panel (a) shows close-up detail of *F. quadrangularis* presence in sandier patches outside the predicted habitat.

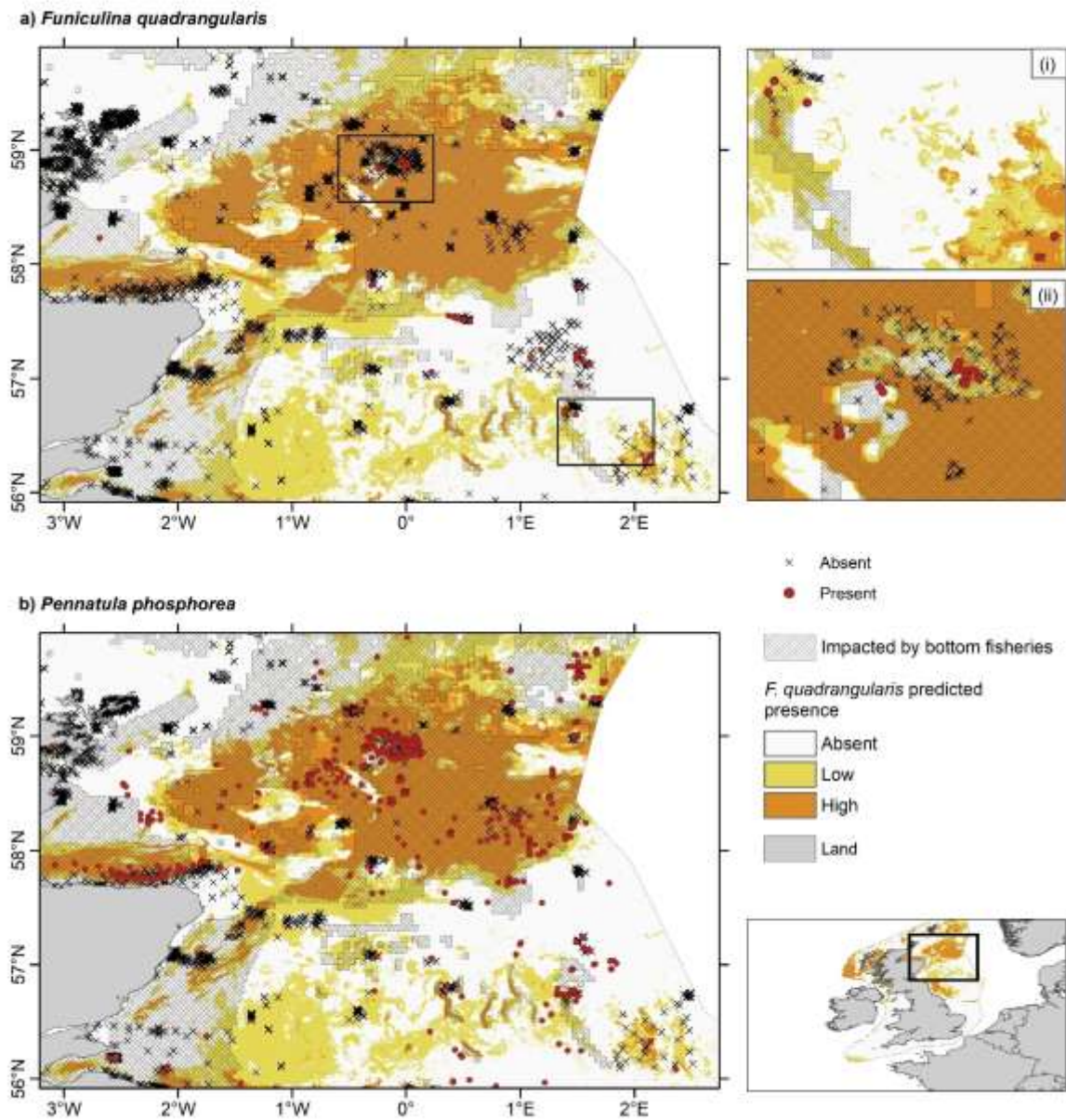
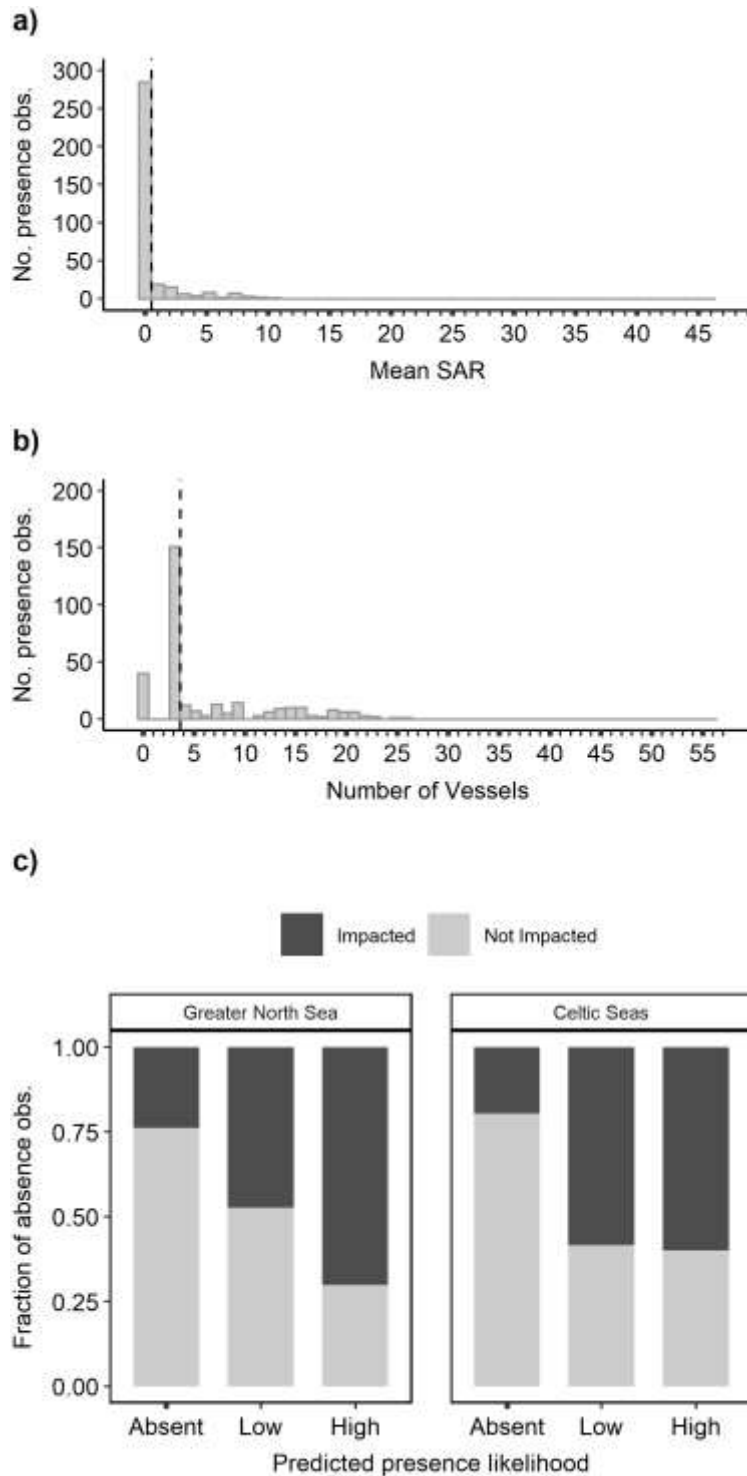


Figure 7. *Funiculina quadrangularis* (and commensal brittlestar *Asteronyx lovenii*) on sandy mixed sediment in the North Sea. Photograph: JNCC & Cefas 2013.



Figure 8. Frequency of *Funiculina quadrangularis* presences across (a) the range of observed swept area ratio (SAR) and (b) number of vessels across the full UKCS study area. Dotted lines indicate the threshold values that were used to split each fishing intensity layer into impacted and non-impacted categories. These were used to plot the fraction of *F. quadrangularis* observed absences in impacted and non-impacted areas (c), where presence was either not predicted, predicted with low likelihood, or predicted with high likelihood.



Tables

Table 1. Summary of *Funiculina quadrangularis*, *Pennatula phosphorea* and *Virgularia mirabilis* presence / absence records used in the Species Distribution Models (SDMs). Each record corresponds to a raster cell with at least one point observation. The total number of records in the full dataset is given for the whole study extent (UKCS). Number of records in the final datasets subsampled to 10 % prevalence are given for each model region.

Species	Total number of records (Presences/Absences)			
	All records	Subsampled records		
	UKCS	UKCS	GRNS	CELS
<i>Funiculina quadrangularis</i>	30103 (379/29724)	3861 (351/3510)	627 (57/570)	3234 (294/2940)
<i>Pennatula phosphorea</i>	30541 (1677/28864)	11132 (1012/10120)	7225 (656/6569)	3267 (297/2970)
<i>Virgularia mirabilis</i>	30301 (1383/28918)	9790 (890/8900)	4884 (444/4440)	4906 (446/4460)

Table 2. Environmental layers included in model development. Variables selected for the final models are indicated by a tick mark.

Variable	Unit	Selected	Source resolution	Source
Bathymetry	m		0.002 deg	EMODnet Digital Bathymetry (EMODnet 2016)
Valley Depth	m	✓		
Relative Slope Position	0-1			
Distance from Channel Network	m	✓	0.002 deg	Calculated from Bathymetry with SAGA for QGIS – Basic terrain analysis tools
Standardised height	m	✓		
Channel Network Baseline	m			
Closed Depressions	?	✓		
Current Velocity	m/s	✓		Cefas Data Hub, Mitchell et al. (2019a) https://doi.org/10.14466/CefasDataHub.62 .
Wave velocity	m/s	✓	0.002 deg	
Winter suspended particulate matter	g/m3	✓		
Sand fraction	%	✓		Cefas Data Hub, Mitchell et al. (2019b) https://doi.org/10.14466/CefasDataHub.63 .
Mud fraction	%	✓		
Gravel fraction	%		0.002 deg	
Sand to gravel log ratio	ratio			
Mud to gravel log ratio	ratio	✓		
ICES Statistical Areas	n/a		vector	ICES Data Portal (ICES, 2005)
ICES Ecoregions	n/a	✓	vector	ICES Data Portal (ICES, 2015)
Mean Spring chlorophyll-a concentration (2017-2019)	mg/m3	✓	1 km	OCEANCOLOUR_ATL_CHL_L3_NRT_OBSERVATIONS_009_036 from http://marine.copernicus.eu/
Mean Summer chlorophyll-a concentration (2017-2019)	mg/m3		1 km	
Mean Autumn chlorophyll-a concentration (2017-2019)	mg/m3		1 km	
Mean Spring primary production (2017-2019)	C mg/m3/d		7 km	NORTHWESTSHELF_REANALYSIS_BIO_004_011 from http://marine.copernicus.eu/
Mean Summer primary production (2017-2019)	C mg/m3/d	✓	7 km	
Mean Autumn primary production (2017-2019)	C mg/m3/d		7 km	
Annual Range in Bottom Temperature (2017-2019)	Deg C		1.5 km	NORTHWESTSHELF_ANALYSIS_FORECAST_PHY_004_013 from http://marine.copernicus.eu/
Maximum Bottom Temperature (2017-2019)	Deg C		1.5 km	
Mean Bottom Temperature (2017-2019)	Deg C	✓	1.5 km	
Minimum Bottom Temperature (2017-2019)	Deg C	✓	1.5 km	

1 Table 3. Model validation results (\pm standard deviation). Results for full models are shown
 2 for validation datasets covering the entire UK continental shelf (UKCS) study area, as well as
 3 separately for the Greater North Sea (GRNS) and Celtic Seas (CELS). Results for the regional
 4 models are shown for both Ecoregions. N = Number of samples included in each training
 5 data set, P = Number of presences in each training data set, TSS = True Skill Statistic.
 6

Model	N	P	Test data	Sensitivity	Specificity	Kappa	TSS
<i>Funiculina quadrangularis</i>							
UKCS	3089	281	UKCS	0.98 (± 0.02)	0.92 (± 0.03)	0.68 (± 0.09)	0.9 (± 0.03)
			GRNS	0.91 (± 0.09)	0.93 (± 0.04)	0.53 (± 0.14)	0.84 (± 0.08)
			CELS	0.98 (± 0.02)	0.92 (± 0.03)	0.72 (± 0.08)	0.9 (± 0.03)
GRNS	502	46	GRNS	0.98 (± 0.04)	0.93 (± 0.03)	0.7 (± 0.09)	0.91 (± 0.05)
			CELS	0.04 (± 0.08)	0.98 (± 0.02)	0.03 (± 0.07)	0.02 (± 0.06)
CELS	2587	235	GRNS	0.12 (± 0.14)	0.92 (± 0.04)	0.03 (± 0.12)	0.04 (± 0.12)
			CELS	0.97 (± 0.01)	0.93 (± 0.02)	0.7 (± 0.07)	0.9 (± 0.02)
<i>Pennatula phosphorea</i>							
UKCS	8906	810	UKCS	0.96 (± 0.02)	0.9 (± 0.01)	0.6 (± 0.03)	0.85 (± 0.01)
			GRNS	0.98 (± 0.01)	0.88 (± 0.01)	0.7 (± 0.03)	0.87 (± 0.02)
			CELS	0.88 (± 0.05)	0.9 (± 0.02)	0.42 (± 0.03)	0.78 (± 0.04)
GRNS	5780	525	GRNS	0.96 (± 0.02)	0.92 (± 0.02)	0.67 (± 0.07)	0.88 (± 0.01)
			CELS	0.59 (± 0.14)	0.85 (± 0.07)	0.3 (± 0.07)	0.44 (± 0.1)
CELS	2614	238	GRNS	0.96 (± 0.03)	0.83 (± 0.03)	0.46 (± 0.05)	0.79 (± 0.03)
			CELS	0.93 (± 0.03)	0.87 (± 0.03)	0.52 (± 0.06)	0.8 (± 0.04)
<i>Virgularia mirabilis</i>							
UKCS	7832	712	UKCS	0.92 (± 0.03)	0.85 (± 0.03)	0.48 (± 0.05)	0.77 (± 0.03)
			GRNS	0.95 (± 0.03)	0.89 (± 0.03)	0.63 (± 0.08)	0.84 (± 0.04)
			CELS	0.89 (± 0.04)	0.83 (± 0.03)	0.38 (± 0.04)	0.71 (± 0.04)
GRNS	3907	355	GRNS	0.91 (± 0.02)	0.95 (± 0.02)	0.73 (± 0.06)	0.86 (± 0.03)
			CELS	0.44 (± 0.12)	0.85 (± 0.07)	0.22 (± 0.05)	0.3 (± 0.07)
CELS	3925	357	GRNS	0.94 (± 0.04)	0.71 (± 0.07)	0.3 (± 0.07)	0.66 (± 0.06)
			CELS	0.9 (± 0.04)	0.82 (± 0.04)	0.41 (± 0.05)	0.72 (± 0.02)

7
8

1 Table 4. Fraction of the area predicted as suitable habitat by the CELS model (presence in >5
 2 cross-validation iterations) overlapping fished areas. Area fished by offshore vessels
 3 comprises the area covered by OSPAR gridded VMS data where swept area ratio (SAR) > 0
 4 (ODIMS, https://odims.ospar.org/odims_data_files/) and inshore vessels the area where > 4
 5 dredgers and trawlers were reported fishing in the ScotMap Inshore Fisheries Mapping
 6 Project (Kafas et al. 2014).
 7

Ecoregion	Fishing Impact	Area (km2)	Percentage of Area
Celtic Seas	Offshore vessels	10193.192	39%
	Inshore vessels	4021.080	15%
	Total Fished	14214.272	53%
	Not fished	12310.761	46%
	Total	26525.033	100%
North Sea	Offshore vessels	30388.753	71%
	Inshore vessels	2030.666	5%
	Total Fished	32419.419	76%
	Not fished	10281.401	24%
	Total	42700.820	100%

8
 9