Non-native marine species in north-west Europe: developing an approach to assess future spread using regional downscaled climate projections

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Abstract

1. Climate change can affect the survival, colonization and establishment of non-native species. Many non-native species common in southern Europe are spreading northwards as seawater temperatures increase. The similarity of climatic conditions between source and recipient areas is assumed to influence the establishment of such species, however in a changing climate those conditions are difficult to predict.

2. A risk assessment methodology has been applied to identify non-native species with proven invasive qualities that have not yet arrived in north-west Europe, but which could become problematic in the future. Those species with the highest potential to establish or be problematic have been taken forward, as well as some that may be economically beneficial, for species

distribution modelling to determine future potential habitat distributions under projected climate change.

3. In the past, species distribution models have usually made use of low resolution global environmental datasets. Here, to increase the local resolution of the distribution models, downscaled shelf seas climate change model outputs for north-west Europe were nested within global outputs. In this way the distribution model could be trained using the global species presence data including the species' native locations, and then projected using more comprehensive shelf seas data to understand habitat suitability in a potential recipient area.

4. Distribution modelling found that habitat suitability will generally increase further north for those species with the highest potential to become established or problematic. Most of these are known to be species with potentially serious consequences for conservation. With caution, a small number of species may present an opportunity for the fishing industry or aquaculture. The ability to provide potential future distributions could be valuable in prioritizing species for monitoring or eradication programmes, increasing the chances of identifying problem species early. This is particularly important for vulnerable infrastructure or protected or threatened ecosystems.

Keywords: subtidal, ocean, dispersal, alien species, invasive, invertebrate

Introduction

Non-native species can cause economic and ecological impacts in the places where they become newly established. Non-native (also non-indigenous, alien) species are considered to be those which have been introduced either directly or indirectly through human activities to areas outside their natural range (Maggs *et al*., 2010). Some of them have harmful consequences for ecosystems, industries or human infrastructure, while some can offer opportunities in certain circumstances (Molnar *et al*., 2008; Cook *et al*., 2013). It has been estimated that non-native species could cost Europe €20 billion per year in damage caused, and in subsequent eradication programmes (Kettunen *et al.,* 2008). Distributions of non-native species are often constrained by the available vectors or mechanisms of introduction, and by the environmental conditions of the receiving areas (Libralato *et al*., 2015). The survival and reproduction of marine non-native species arriving in a new area are constrained by factors such as temperature and salinity, but also depth, substrate type and food availability; and it is thought that climate change may facilitate their persistence or reproduction in locations not previously habitable (Cook *et al*., 2013). Ecosystems within north-west Europe are witnessing rapid changes as a result of anthropogenic climate change (MCCIP, 2013) but also extensive habitat modification and resource use, and so are particularly vulnerable to the added impacts of non-native species. The North Sea in particular is considered to have a high degree of environmental change (Larsen *et al*., 2014), where sea surface temperatures have risen more rapidly than the global average over the past 50 years (Hobday and Pecl, 2013).

Already many marine non-native species have spread and become established in countries within north-west Europe and are causing economic and biological damage (Cook *et al*., 2013). The impact of each ranges from negligible to catastrophic for native organisms and industries. They cause a range of impacts including outcompeting and displacing native species, affecting whole food chains and physical processes and damaging infrastructure (Molnar *et al*., 2008). Climate change is already known to have created conditions which facilitated an increased range of some non-native species in the UK and Ireland, including the Pacific oyster *Crassostrea gigas* and Asian club tunicate *Styela clava* (Cook *et al*., 2013). There is clearly potential that ranges will expand further in the future. One species not yet established in the UK is *Mnemiopsis leidyi*, a comb jelly that is native to the Atlantic coast of the USA, and which is thought to have been introduced to the Black Sea in the 1980s through ballast water (Didžiulis, 2013). In recent years, the species has spread to Scandinavia and Dutch and German coasts, potentially enabled by the rise in North Sea temperature (Oliveira, 2007). High numbers of the species have been associated with collapses in valuable fish stocks (Didžiulis, 2013) and so could be a major problem economically if it became established in north-west Europe. Increased sea temperature could also mean that invasive seaweeds such as wakame *Undaria pinnatifida* are able to establish more successfully, and the rapa whelk *Rapana venosa*, which requires warm waters to deposit egg capsules, may be able to spread to northern areas by mid-21st century (Cook *et al*., 2013). However, climate change effects are complex, and it is not certain that all species will expand their range. Heavy rainfall and resultant decreases in salinity may reduce the spread of the carpet sea squirt *Didemnum vexillum* for example (Cook *et al*., 2013) despite favourable rising seawater temperatures, and the southward spread of the red king crab *Paralithodes camtschaticus*, which requires low temperatures, may be curtailed by further climate change (Natural England, 2009). There are a large number of non-native species within Europe, and the complexities involved in understanding how climatic change may affect where these species are able to become established means that for environment managers prioritizing the monitoring and eradication of these species is not straightforward. Roy *et al*. (2014) identified that in order for preventative action to be taken, there is an urgent need to anticipate which species could arrive and cause future problems. It is necessary then to further understand which species pose the greatest threat to north-west Europe in terms of economic or biological impact, and in which areas these organisms may be able to survive and thrive in the near future and in the long term. This will enable individual species to be prioritised for management or eradication, which is particularly necessary in times of financial constraints on resources.

Some of the highest impact non-native species can be identified through risk assessment, based on information available on the effects caused in other areas and life history traits which affect likelihood of spread. Having identified these species, various species distribution modelling techniques are available to project or predict the extent of suitable environmental conditions. Some of these methodologies have been used to assess the potential future spread of non-native marine species once they have arrived at a particular destination (e.g. Herborg *et al*., 2007; Jones *et al*., 2013). Species distribution models make use of correlations between observed organism distributions and climatic or habitat variables. By looking at the current range of environmental parameters, such as depth, temperature, salinity and stratification tolerated by a species, it is possible to project future distribution using predictions of how the physical environment in an area will change in the future. Maxent is one of many, freely available, species distribution models (Phillips *et al*., 2006; Reiss *et al*. 2011) that has been applied in both terrestrial and aquatic ecosystems around the world. Previous studies using Maxent to consider future habitat distribution changes around north-west Europe have used global climate model projections (e.g. Jones *et al*., 2013). Global climate models (GCMs) incorporate the physical drivers of large-scale climate change, but are less able to resolve local-scale shelf sea processes, such as currents, stratification and mesoscale processes (Tinker *et al*., 2015). While GCMs certainly produce useful projections, a new set of north-west European shelf seas climate projections using an ensemble approach have recently been made available, providing much higher resolution (12 km cells) and more detail on the processes within the shelf seas (Tinker *et al*., 2015). Such changes include a centennial rise in annual mean sea surface temperature of 2.9 °C, and a freshening of 0.41 psu, of 2069-2098 relative to 1960-1989 (Tinker *et al*., under review). By nesting the higher resolution shelf seas projections within the broader scale GCM projections (although both are based around the same HadCM3 physical model), it is possible to train a species distribution model on global species presence data, and then make use of the more detailed regional projections to understand how that species' habitat suitability may change in the future. Such an approach is particularly useful for non-native organisms as the broad-scale training dataset encompasses the original 'native' locality of the particular species, whereas the high-resolution dataset is used to characterise the potential distribution in a new area where there may be very few observations at the moment.

The aims of this study were to make use of these new climate model outputs and determine which non-native species pose the biggest risk of spread and conservation impact within north-west Europe by:

- Conducting a thorough risk assessment to identify species with the greatest potential to spread and consequently cause environmental or economic harm, or which could potentially be economically exploited in the UK, in the future; and
- Determining future habitat distributions of non-native species by combining GCM projections with the most recently available north-west European shelf seas climate projections.

Methods

This study involved four key steps:

1. Identify non-native species with the potential to spread and establish within the north-west European shelf seas, or that are already present and have the potential to spread further;
2. Prioritise these species, using a risk assessment framework, to identify which to take forward for distribution modelling on the basis of potential impacts and invasiveness;
3. Build, train and assess the species distribution model for the present day on these key species. Combine high-resolution north-west European shelf seas projections with global projections using a nesting approach to produce a useable global dataset; and
4. Use the model to project future distribution change using the nested projections of future environmental conditions which have increased complexity within the shelf seas region of concern.

Step 1. Species identification

There are a number of mechanisms by which non-native species may become established, which were considered when identifying species for the risk assessment. Non-native species already present in the UK and nearby European countries, may be able to spread by natural processes (secondary colonisation) if environmental conditions (e.g. climatic changes in the future) allow. Secondary colonisation and spread can be facilitated by the movement of recreational boats between harbours, and by translocation of stock between aquaculture facilities. As seawaters warm in the future, it is likely that novel aquaculture species will be cultured in Europe and that some of these will escape captivity. Species were not identified from this route as it is difficult to anticipate which species might be brought to Europe deliberately in the future.

In order to produce a list of non-native species to include in detailed modelling for the UK, the following qualities were considered: Non-native species already present in the UK and known to cause problems either here or in other countries, and non-native species present in north-western European countries including those with a slightly warmer climate than the UK (e.g. Atlantic coasts of Portugal, Spain, France, Belgium, Holland, Germany, Denmark), and known to cause problems in those countries.

The list was compiled from a number of sources. These were the UK Technical Advisory Group on the Water Framework Directive (UKTAG, 2014), the GB Non-native Species Secretariat (NNSS, 2014); UK species lists provided by Natural England and The Marine Biological Association of the United Kingdom (MBA), the database of Delivering Alien Invasive Species Inventories for Europe (DAISIE, 2015), the global database of marine invasive species on the Conservation Gateway of The Nature Conservancy (Molnar *et al*., 2008), Cefas Priority Species Report (Cefas, 2015a), and the report on horizon scanning for invasive species in Great Britain (Roy *et al*., 2014)

From the subsequent short-list of non-native species already present in Europe (89 species in total, see Table S1), the species that were scored highest by The Nature Conservancy database (Molnar *et al*., 2008) (ecological impact scores of 3 or 4) or identified on the Cefas Priority Species Report (Cefas, 2015a) were taken forward for risk assessment as described below (40 species in total, see Table S2). For the Nature Conservancy database, Molnar *et al*. (2008) determined ecological impact scores by assessing how a species affects the viability and integrity of native species and biodiversity, based on documented evidence. The highest score is 4, which is achieved if a species disrupts ecosystem processes and wider abiotic influences. A score of 3 means that the species disrupts multiple species, and some wider ecosystem functions, or it affects keystone or species of conservation value. A lower score is achieved if impacts are less, for example, if only one taxon is affected. The Cefas Priority Species Report (Cefas, 2015a) contains species which have previously been assessed by Cefas or other European institutes as of high or moderate impact, or which are on European horizon scanning lists or listed by Roy *et al*. (2014).

A species which arrives by ship (either through hull fouling or in ballast water) and is not currently able to survive in UK waters, may potentially be able to establish populations if conditions become more suitable with further climate change. To identify species which are not already established in Europe, current shipping routes to the UK were investigated to determine the likely ports of origin from which species could arrive and become established. Tidbury *et al*. (2014) analysed the different shipping routes for their potential to act as a vector for introduction of non-native species, and found that the ports of origin for commercial shipping to the UK with the highest number of voyages (greater than 500 voyages to the UK per year) were all in Europe or elsewhere in the UK which suggested heightened risk of secondary colonization primarily by organisms already present in the region. Regarding recreational boating, nearly all cruising routes were to continental Europe or within the UK. Thus species most likely to be introduced to the UK through current shipping and boating practices (the vast majority of which are within Europe) are highly likely captured in the species search as described above and no other species were put forward for risk assessment based solely on long-distance shipping routes.

Step 2. Risk assessment and prioritization

A thorough risk assessment was carried out on the 40 invertebrate species identified during the first process. An online "*Marine Invertebrate Invasiveness Screening Kit (MI-ISK)*" (Cefas, 2015b) was developed based on the widely used non-native freshwater fish toolkit "*Risk Identification and Assessment Methodology*" (Copp *et al.,* 2005). These toolkits include protocols and questionnaires by which species can be screened to determine their relative 'invasiveness', and thus the potential threat that they might pose in the wild. Forty-nine questions are answered about the species life history, evidence of invasiveness elsewhere in the world and whether or not they cause impacts to ecosystems or infrastructure where established. An associated confidence level and a numeric score is calculated. Animals receive a high score (19 to 40) for invasiveness if there is a history of repeated introductions outside their natural range, large impacts to ecosystems or infrastructure where they become established, and/or if their life history characteristics suggest that they could be easily spread and become established in new areas. A medium score (13 to 18) is given if there is some history of invasion and some associated impacts. The species are characterized according to whether they possess undesirable traits, including reproductive strategies that enable rapid proliferation and broad dietary characteristics such as generalised feeding, that both enable species to out-compete native populations. Species information used in the risk assessment came from the GB Non-native Species Secretariat (NNSS, 2014), database of Delivering Alien Invasive Species Inventories for Europe (DAISIE, 2015), Global Invasive Species Database (2015), the Invasive Species Compendium (CABI, 2015), Cefas Priority Species Report (Cefas, 2015a) and the broader scientific literature.

Only invertebrates could be assessed due to the nature of the tool, and so aquatic plants were prioritised based on their ecological impact scores (Conservation Gateway of The Nature Conservancy, 2008) and listing on the Cefas Priority Species Report (Cefas, 2015a). Lastly, freshwater/brackish species were removed from the list, as the resolution of the climate models was considered not sufficientl to allow reliable projections or outputs. Marine/brackish species were however included. The invertebrate species taken forward for distribution modelling were those with MI-ISK scores of 13 or over (a 'medium' or 'high' score of invasiveness), and are listed in Table 1, along with their impact scores and additional detail. The full MI-ISK question scores are available in Table S3. The algal and angiosperm species taken forward were those which either had an Ecological Impact Score of 3 or 4, or were listed in the Cefas Priority Species Report (Cefas, 2015a).

Step 3. Build, train and assess the model for the present day

Using high-resolution shelf seas projections on their own would not cover a sufficiently large area or provide a broad enough range of experienced climate conditions to enable assessments of habitat suitability. For example, the shelf seas area does not include the environmental conditions experienced by a species which is native to the sub-tropics or the Arctic, and so these conditions would be excluded from any habitat suitability assessment. Therefore in order to make use of these newly-available high-resolution projections while taking account of the conditions experienced globally by each species, it was necessary to nest high-resolution regional data within a grid of coarse GCM outputs. This resulted in habitat suitability functions encompassing the entire range of each species and in particular the 'native' range of the non-native species, that could subsequently be applied at the local scale to a focal location where the species may not yet be fully established.

A set of environmental, marine climate parameters available in standard climate projections were chosen to drive the species distribution model (Maxent) as in previous work (Cheung *et al.*, 2009; Jones *et al*., 2013). To build and test the model under present day conditions the outputs of the global and regional model were used as averaged annual means from 1980–2009 (hereafter termed 'present day') taking the parameters: i)bathymetry, ii) near bottom temperature iii) sea surface temperature, iv)near bottom salinity, v) sea surface salinity, vi) bulk thermal stratification (difference between sea surface and near bottom temperature) and vii) bulk haline stratification (difference between sea surface and near bottom salinity).

Projections were obtained from the Met Office Hadley Centre. Global 1.25 degree resolution projections were from a Perturbed Physics Ensemble (PPE) (Collins *et al*., 2011) of the Atmosphere-Ocean Global Climate Model HadCM3 (Gordon *et al*., 2000; Pope *et al*., 2000). This PPE consisted of the standard version of the model (the unperturbed ensemble member) with 10 ensemble members with a number of atmospheric parameters perturbed in order to span the range of uncertainty in Climate Sensitivity (the amount of global mean warming associated with a doubling of CO_2). In this study the unperturbed ensemble member is used, which is equivalent to the standard version of HadCM3 and HadRM3. The unperturbed member of this ensemble has been dynamically downscaled with the shelf seas model POLCOMS (Proudman Oceanographic Laboratory Coastal Ocean Modelling System; Holt and James, 2001; Holt *et al*., 2001) to produce the north-west European shelf seas projection (Tinker *et al*., 2015, in review) used in this study, with a resulting resolution of 12 km (1/9° latitude by 1/6° longitude), covering 43°N – 63°33'20"N and 18°20'W – 13°E (see Fig. 1).

The downscaled shelf seas projections were nested within the driving global projections using Python 2.7 (Python Software Foundation, 2010) (packages netCDF4 and numpy) with a resulting global dataset at 0.5 degree resolution. The global ocean fields were bi-linearly interpolated from the native 1.25° resolution to the 0.5°, while the downscaled regional fields were aggregated up (averaged) from their native 1/6°x1/9° resolution to the required 0.5°. They were then copied into the global data. As the regional data and the global data are consistent (the global data is from the run that forced the regional model), the two datasets match at the boundary. This intermediate resolution was necessary as it still captures the local-scale processes of the shelf seas model while not reducing the resolution of the GCM more than is appropriate. This intermediate resolution (0.5 degree) grid of present day environmental parameters was then used as the driver for the species distribution model.

The Maximum Entropy (Maxent) species distribution model was used (Phillips *et al*., 2006) because it provides a robust method for assessing habitat suitability (e.g. Vierod *et al*., 2015; Reiss *et al.* 2011) compared to other, similar modelling methodologies. Maxent randomly selects training data points and generates habitat suitability by combining presence-only occurrence data and chosen environmental variables and predicting the potential distribution of a species, or habitat suitability. The remaining presence data points are used to test the model fit. Projected environmental conditions are then used to force the model to predict future habitat suitability, based on the same environmental preferences. Maxent estimates the probability distribution of the grid by finding the distribution that has the maximum entropy (i.e. most uniform), subject to the constraints of incomplete information (Phillips *et al*., 2006). The probability distribution is defined by the environmental variables used in to the model. The term "habitat suitability" is used here to describe the bathymetry and the environmental hydrographic conditions of the area, and does not include characteristics of bottom substrate, or local species interactions within communities (i.e. food availability etc.).

Species occurrence data were downloaded from two databases: the Ocean Biogeographic Information System (OBIS) (http://www.iobis.org) and the Global Biodiversity Information Facility (GBIF, 2015) (http://data.gbif.org). The data were cleaned using the statistical software R (version 3.0.3 (R Core Team, 2013), to remove duplicates, occurrences outside the accepted depth and Food and Agriculture Organisation of the United Nations (FAO) area ranges, and to remove reported occurrences on land (due to mis-recording of locations). This was done by taking FAO areas and depth ranges from OBIS and Sea Life Base (http://www.sealifebase.org), with depth being rounded up to the nearest 100 m to ensure that all reasonable presence data were included. Cleaning and sense-checking the data in this way reduced the chance that species which were misidentified or mis-recorded were included in the presence dataset. The data were aggregated to the intermediate resolution 0.5 degree grid, with a value in each cell for presence or absence. This aggregation reduced the number of presence points within a small area (e.g. at a regularly sampled beach or marina). Maxent was then run for each species using the model interface (version 3.3.3k) downloaded from http://www.cs.princeton.edu/~schapire/maxent/. The presence data was uploaded into 'Samples' and the current environmental data (climate and bathymetry) into 'Environmental layers'. Auto features were used along with 'jacknife' which checks variable importance. Maxent automatically chooses the number of training values based on the number of presence data points available. The number of training points used across the different species varied considerably, from the highest for the American lobster (202) to the lowest for the seaweed wakame (16), with those with the higher value likely to be a better fit to reality than others. The number of training and presence points are given in Table 2. Maxent then tests the 'skill' of the resulting relationships using the Area Under the Curve (AUC) value. The AUC value (from 0 to 1) is a measure of the performance of the model; the higher the value the better the model fit. A threshold value of 0.8 or above was chosen, based on a review of published habitat suitability models by Mercks *et al*., 2011). It should be noted that this type of modelling can be subject to autocorrelation due to biased and opportunistic species sampling, and so this value of 0.8 is used as a guide rather than an absolute value of a robust output.

Step 4. Using the model: future distribution change

Projections from climate model output were obtained from the same unperturbed member of the downscaled HadCM3 model as described above, under an SRES A1B business as usual scenario, characterised as 'medium' emissions. As described above these were nested within the global climate model outputs to produce a set of intermediate resolution projections for two future timeslices: 2040–2069 and 2069–2098. Hereafter we refer to these timeslices by their middle year: 2055 or 'near future'; and 2084 or 'end of century'.

Data outputs from these projections were included as inputs into species distribution model (Maxent), as described above, but with the future environmental data entered into 'Projection layers'. This was carried out for each species in each future time scenario. This gave a global half-degree resolution grid of habitat suitability ranging from 0 to 1 for the present and future scenarios. The latitudinal centroid for each time period and species was then calculated, both globally and for the extent of the shelf seas model alone, giving the centre of the latitudinal range for each species and a measure of how it has changed from the current to the future period, both globally and around the UK. The centroid *C* was calculated using the equation from Cheung *et al*., (2009):

$$C = \frac{\sum_{i=1}^{n} Lat_i \cdot Abd_i}{\sum_{i=1}^{n} Abd_i}$$

where Lat_i is the central latitude of the spatial cell *i*, *Abd* is the predicted relative habitat suitability of the same cell, and *n* is the total number of cells. The difference between the two latitudinal centroids in the current and projected years was then calculated in kilometres (Cheung *et al*., 2011):

$$Latitudinal\ shift\ (km) = (Lat_m - Lat_n) \times \frac{\pi}{180} \times 6378.2$$

where Lat_m and Lat_n are the latitudinal centroids in the projected (*m*) and current (*n*) years, and 6378.2 is the approximate equatorial radius of the Earth in km.

Results

Steps 1 and 2. Risk assessment and prioritisation

The MI-ISK scores for the marine invertebrates showed that the Pacific oyster, the slipper limpet *Crepidula fornicata,* the Asian club tunicate, the Asian shore crab *Hemigrapsus sanguineus* and the northern Pacific starfish *Asterias amurensis* all scored ‘high’ and had the highest potential risk for spread and subsequent impact. All other invertebrates scored ‘medium’. Full MI-ISK scores are included in supplementary material (Table S3).

Step 3: Validation of the present day species distribution models

For five species, as listed in Table 1, there were insufficient presence data to either run the model or produce the robust output with an AUC greater than 0.8, and so these were not taken forward to the final modelling stage. For the remaining species which were taken forward for the future modelling, the AUC values, the variable with the highest percent contribution, the total number of presence data points and the number of training points used are all presented in Table 2.

All AUC values are above 0.9 showing good predictive power of the models. Further detail on the modelling results is provided in supplementary Table S4.

Step 4. Future distribution change

Species distribution modelling found that habitat suitability ranges for all species would move poleward at a global scale by up to 843 km (9.5 km/yr) (Fig. 2), and generally northward within the European shelf seas by up to 115 km (1.3 km/yr) by the end of the century (Fig. 3), although American lobster *Homarus americanus* and Conrad’s false mussel *Mytilopsis leucophaeata* were exceptions with predicted southwards movement. The American lobster was projected to have a distribution shifted south by 2055 and then north by 2084 while Conrad’s false mussel’s habitat suitability shifts south over both time periods. The species with latitudinal centroid projected to move the furthest globally by the end of the century are kuruma prawn *Penaeus japonicus* (843 km, 9.5 km/yr), American hard-shelled clam *Mercenaria mercenaria* (620 km, 7.0 km/yr), slipper limpet (615 km, 6.9 km/yr), American razor clam *Ensis directus* (572 km, 6.4 km/yr) and Manila clam *Ruditapes philippinarum* (703 km, 7.9 km/yr). Within the shelf seas area, the species with the greatest northward latitudinal centroid change by 2084 are cord grass *Spartina townsendii var.*

anglica (115 km, 1.3 km/yr), wireweed *Sargassum muticum* (110 km, 1.2 km/yr), Asian club tunicate (90 km, 1.0 km/yr), Pacific oyster (86 km, 1.0 km/yr), Asian shore crab and kuruma prawn both (81 km, 0.9 km/yr).

For the four highest MI-ISK scoring species (Pacific oyster, Asian shore crab, Asian club tunicate and slipper limpet) and most of the species assessed, habitat suitability was projected to shift northwards by 2055 and 2084 compared with 1995, particularly in the southern North Sea and along the Scandinavian coastline (Fig. 4). The American lobster showed higher habitat suitability in deeper waters, particularly along the shelf edge and in the Bay of Biscay. Conrad's false mussel showed a decrease in suitability around the northern UK and Scandinavia in 2055 and 2084. Within these plots, the difference in resolution from the global and shelf seas models can be seen by looking at the outline of the coast. The land area covered by the GCM only is based on data at 1.25 degree resolution and so is not highly detailed. However, the area derived from the shelf seas model can be clearly seen by the more detailed coastal outline.

Discussion

The risk assessment found a range of non-native species which are either already impacting marine environments within the UK or north-west Europe or which pose a significant threat. The species distribution models suggest a change in habitat suitability around the shelf seas over time with predicted climate change scenarios. This will potentially result in the majority of the species included in the risk assessment responding to this with a northward shift within the next 50 to 100 years and establishing in new areas. The models predict how far these non-native species may be able to spread including to areas where conditions are not currently suitable. Further spread may occur via natural dispersal or facilitation by further shipping and other human activities, but the environmental conditions that currently limit survival and reproduction will become less restrictive in the coming decades.

The risk assessment and distribution modelling identified Pacific oyster, Asian shore crab, Asian club tunicate, wireweed and cord grass as species of particular concern due to their potential future suitable habitat and the impact that they have on ecosystems or industries; evidenced by high MI-ISK scores and the greatest anticipated latitudinal shifts in habitat suitability. Changing environmental conditions could allow these species to increase their range substantially, with ecologically and economically damaging impacts. For example, the Asian shore crab is anticipated to spread north around the British Isles and along the Scandinavian coasts, where it has the potential to outcompete the native green shore crab *Carcinus maenas* (Epifanio, 2013). In addition to effects on individual species, there are also likely to be changes to whole food webs, and this too can be modelled given scenarios of projected spread, population growth and ecological characteristics (e.g. Pinnegar *et al*., 2014). The Pacific oyster forms reefs when it occurs in high numbers, and there is concern that this could happen in the UK, which could prevent certain protected areas from meeting ecological status levels required by legislation (Herbert *et al*., 2012). Economic problems which could be envisaged include wireweed and Asian club tunicate fouling man-made structures such as aquaculture facilities, with consequential declines in mussel production in the case of the tunicate (NNSS, 2015).

It has been suggested that in some circumstances non-native species may enrich ecosystems rather than causing harm (Libralato *et al*., 2015). Additionally, some of the species considered in this study could represent a hither-to unexploited commercial opportunity where they have invaded. For example, shellfish such as the American razor clam, the American lobster, the Pacific oyster and the Manilla clam and seaweed such as wakame, are edible species which could be commercially exploited, either through wild harvest or aquaculture. With very careful management, wild capture could provide a mechanism to limit population sizes and subsequent impacts while also providing short-term commercial gain although much caution should be taken with this approach. Detailed cost benefit analyses would be required, especially in relation to the possible loss of revenue from native species potentially impacted either directly or indirectly by climate change and the introduced non-native organisms, before the exploitation of these species should be really considered. The Pacific oyster has been harvested in the UK for a number of decades in areas where it is abundant (Davison, 1976). Herbert *et al*. (2012) state that in certain areas where wild settlement is inevitable due to the volumes of boat traffic, harvesting the species may be the only way to manage the stock. The authors suggest that fisheries support schemes could be appropriate to develop the new fishery. In the Bay of Biscay, the American razor clam is collected for human consumption and as bait (Arias and Anadon, 2012), and it is considered that densities in certain areas are high enough to sustain a fishery (Witbaard *et al*., 2013). In an ICES Alien Species Alert Gollasch *et al*. (2015) note caution with regard to establishing such a fishery due to the potential to cause further spread. Cord grass can spread rapidly within soft sediments and so its ability to thrive in new areas has been of benefit with regard to stabilising coastlines (Davidson, 1991). However, this benefit needs to be balanced with the reduced biodiversity within the cord grass monoculture, in comparison with biodiversity among the native saltmarsh plants which are slower to establish (Davidson, 1991).

The new high-resolution north-west European shelf seas climate projections suggest a geographic pattern of sea temperature changes, with greater winter/spring warming in the southwestern North Sea, and summer/autumn warming in the Celtic Sea and North Sea (Tinker *et al*., in review). The use of the downscaled model outputs allows tides, regional currents and stratification to be represented across the north-west European shelf seas area (Tinker *et al*., in review), which are important for modelling the physical conditions in this region, and for the survival and reproduction of a number of species. The GCM does not represent these processesand so if used to represent certain shelf regions, there may be deficiencies in the ability to model the underlying species distribution-habitat relationships.

It should be noted that this study is not indicative of an inevitable spread of a range of non-native species, but that it demonstrates the potential spread based on the projected environmental suitability (Jarnevich *et al*., 2015). The habitat suitabilities were compared to the present day (averaged time period), and were based on recorded occurrences and not absolute distributions. Thermal niche alone does not fully predict invasive species distributions (Parravincini *et al*., 2015), and for a complete picture there are many factors to consider other than those included in this study. For example, it is unlikely that species will thrive in large numbers at the boundaries of projected areas of habitat suitability although they may be present. Additional factors such as local hydrodynamics, substrate type and food supply may mean that these areas remain unsuitable (Cook *et al*., 2013).

Species distribution models must be interpreted with appropriate caution (Jarnevich *et al*., 2015). By practicality, presence data used in the models are incomplete and are likely to be biased to areas where there is greater sampling effort, creating autocorrelation errors. There are more mechanistic modelling approaches available (Jennings and Brander, 2010), however the Maxent approach offers the opportunity to screen large numbers of species relatively quickly and easily and so should be viewed as complementary to more complex approaches. A study comparing different species distribution modelling techniques of benthic species found Maxent to be one of the most robust, including for small sample sizes (Bučas *et al*., 2013) and others have found it compared well against other techniques (*et al*Elith *et al*., 2006; Phillips *et al*., 2006; Pearson *et al*., 2007; Reiss *et al*., 2011; Padalia *et al*., 2014). This study focused on species of interest to the UK and north-west Europe, and the climate projection dataset used was designed to be of highest possible resolution around the shelf seas. Therefore caution should be if interpreting habitat suitability predictions for elsewhere in the globe where model detail is lower. While the half-degree resolution used in the models is high relative to global data, much of the coastal and intertidal species presence data points are lost as a result of this action. Therefore for species that occur very close to the coast, this missing zone must be considered when using the habitat suitability scores.

Aspects of climatic change not included within the models here are changes in pH or oxygen saturation. Ocean acidification is predicted to have diverse effects on organisms. It is possible that algae and jellyfish, that do not have calcareous skeletons, may benefit while molluscs and some crustaceans may be at a disadvantage (Hall Spencer *et al*., 2015). Therefore, with increased ocean acidification later in the century, the predicted habitat suitability for the Pacific oyster may be an overestimate, while that for the comb jelly *M. leidyi,* and seaweeds *C. fragile*, wireweed and wakame it may be overly conservative. Greater intensity and frequency of storms may also favour the spread of non-native species, particularly seaweeds and animals that attach to seaweeds (Cook *et al*., 2013). The effects of these parameters on individuals and ecosystems are complex and so further research will help to understand the complexities affecting spread, survival and population persistence of species.

There are a number of sources of uncertainty that will affect these results. Full quantification of this uncertainty is outside the scope of this study, however, we briefly discuss them here. These sources broadly fall into a three of categories: the underlying climate projections; the species distribution modelling approach; and the observations used to train it. Climate projection uncertainty typically includes choice of emission scenario (here we use a single emission scenario, A1B), model structure uncertainty (we use a single GCM and shelf seas model (HadCM3 and POLCOMS), and model parameter uncertainty (we use the standard (unperturbed) member of a perturbed physical ensemble). We therefore note that these results give a plausible estimate of possible future invasive species distribution but not necessarily characterise the full range of possibilities. However, we recommend future work to explore the implications of these underlying uncertainties, and to explore the uncertainties in distribution modelling such as through using a multi-model approach (e.g. Jones *et al*., 2013). The limitations of observation sampling should be particularly highlighted. If a species has not realised its full fundamental niche (i.e. it does not yet occur in all of the places where it could survive; a situation that is highly likely in an invasive species), then it is difficult to make predictions about its future distribution, as the predicted niche may be smaller than the full 'realisable' potential niche (Phillips *et al*., 2006). This is also a problem when species occurrence records come from only one part of the global distribution (for example if many records occur close to a research station) which does not represent the whole species niche, or when there are too few occurrence data points. Sufficient sampling effort is required to ensure that the models are robust (Phillips *et al*., 2006). For some of the species which had a high MI-ISK score, it was not possible to carry out Maxent modelling due to a low number of presence records globally. As more records are digitised and made publically available, this will help to increase the accuracy of modelling techniques and the forecasts that they give.

Prevention of establishment or arrival is recognised as the most effective management approach to combat non-native species (Caffrey *et al*., 2014; Caplat and Coutts, 2011). The Convention on Biological Diversity (CBD) and new European legislation on the prevention and management of invasive species (IAS regulations) both focus on identifying and managing the pathways and vectors of introduction and spread. These pathways and vectors can be varied and complex, such as international shipping, recreational boating and trans-shipment of aquaculture species, therefore individual countries cannot stop the spread of introduced non-native species alone, making international cooperation vital. As such, there are a number of initiatives aimed at sharing information and prioritising species for further research and monitoring, such as the North European and Baltic Network on Invasive Alien Species (NOBANIS, 2015), Delivering Alien Invasive Species Inventory for Europe (DAISIE, 2015) and Reducing the Impact of Non-Native Species in Europe (RINSE, 2014). The IAS Regulation has a target to have identified, prioritised and controlled or eradicated species which are highlighted by risk assessments as a priority and to manage the pathways of introduction and spread by 2020, likewise the European Marine Strategy Framework Directive (MSFD), which also targets the management of non-native species, has an aim of achieving Good Environmental Status by 2020. Conservation agencies and scientists are working together to try to achieve this. However further regulations such as The International Convention for the Control and Management of Ships' Ballast Water and Sediments (BWM Convention), which has yet to be ratified, are required to help prevent further introductions of new species. Once introduced, it is very difficult to prevent spread of a species in the aquatic environment, although not impossible given sufficient resources.

This study contributes to the growing knowledge-base available, aimed at informing the measures required to monitor, prevent introduction or slow the spread of non-native species in the marine environment, and potentially eradicate them altogether. For species that have arrived recently, their impact within European ecosystems is not yet fully understood. Ecosystems can be resilient to some changes, and the addition of one species may not always mean the loss of others. However, the impact can only be determined by sufficient monitoring and screening of both the introduced species, and the ecosystem that has been invaded. Novel techniques such as analysis of environmental DNA (eDNA) may facilitate the rapid screening of potential introduction sites (e.g. ports and harbours) for particular species (Goldberg *et al*., 2015). It is clear from these models that the habitat around north-west Europe will become more suitable for certain non-native species in the coming century, and so environment managers need to be mindful of this. Early detection of non-native species is crucial to stop them becoming established (Roy *et al*., 2014; Cefas, 2015a). The risk assessments and modelling projections in this study could be used to prioritise the species for monitoring surveys and impact assessments, increasing the chances that the most dangerous species are identified early. The results of this study will enable managers of protected areas or important infrastructure, such as marinas and power stations, to identify high risk areas and priority species as soon as they arrive, and activate eradication programmes before they become fully established, thus saving money and conferring a higher chance of success. However eradication of such species may be an ongoing process until the species source or pathways of spread are removed.

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References

Arias A, Anadon N. 2012. First record of Mercenaria mercemnair (Bivalvia: Veneridae) and Ensis directus (Nivalvia: Pharidae) on Bay of Biscay, Iberian Peninsula. *Journal of Shellfish Research* **31**: 57-60.

Bučas M, Bergström U, Downie A-L, Subdblad G, Gullstöm M, von Numers M, Šiaulys A, Lindegarth M. 2013. Empirical modelling of benthic species distribution, abundance, and diversity in the Baltic Sea: evaluating the scope for predictive mapping using different modelling approaches. *ICES Journal of Marine Science* doi: 10.1093/icesjms/fst036.

CABI. 2015. The Invasive Species Compendium http://www.cabi.org/isc/datasheet/87296 [5 January 2015].

Caffrey J, Baars J-R, Barbour J, Boets P, Boon P, Davenport K, Dick J, Early J, Edsman L, Gallagher C. *et al*. 2014. Tackling Invasive Alien Species in Europe: the top 20 issues. *Management of Biological Invasions* **5**: 1–20.

Caplat P and Coutts SR. 2011. Integrating Ecological Knowledge, Public Perception and Urgency of Action into Invasive Species Management. *Environmental Management* **48**: 878–881.

Cefas. 2015a. Development of priority species lists for monitoring and surveillance of marine non-natives in the UK. Cefas contract report C6486 Issued 08/05/15. Cefas, UK.

Cefas. 2015b. MI-ISK Marine Invertebrate Invasiveness Screening Kit. Decision support tools for the identification and management of invasive non-native aquatic species. https://www.cefas.co.uk/services/research-advice-and-consultancy/invasive-and-non-native-species/decision-support-tools-for-the-identification-and-management-of-invasive-non-native-aquatic-species/ [January 2015].

Cheung WWL, Lam VWY, Sarmiento JL, Kearney K, Watson R, Pauly D. 2009. Projecting global marine biodiversity impacts under climate change scenarios. *Fish and Fisheries* **10**: 235–251.

Cheung WWL, Dunne J, Sarmiento JL, Pauly D. 2011. Integrating ecophysiology and plankton dynamics into projected maximum fisheries catch potential under climate change in the Northeast Atlantic. *ICES Journal of Marine Science* **68**: 1008–1018.

Collins M, Booth BBB, Bhaskaran B, Harris GR, Murphy JM, Sexton DMH, Webb MJ. 2011. Climate model errors, feedbacks and forcings: a comparison of perturbed physics and multi-model ensembles. *Climate Dynamics* **36**: 1737-1766.

Cook EJ, Jenkins S, Maggs C, Minchin D, Mineur F, Nall C, Swell J. 2013. Impacts of climate change on non-native species. *MCCIP Science Review 2013*: 155-166.

Copp GH, Garthwaite R, Gozlan RE. 2005. Risk Identification and Assessment Methodology: concepts and perspectives on protocols for the UK. Cefas Science Series Technical Report No. 129. Cefas, UK.

DAISIE. 2015. Delivering Alien Invasive Species Inventory for Europe http://www.europe-aliens.org/default.do;jsessionid=8D5020C8900486C2630B81CB8D798B5A [25 November 2015].

Davidson. 1976. Oyster Fisheries of England and Wales. Ministry of Agriculture, Fisheries and Food. Directorate of Fisheries Research. Laboratory Leaflet No. 31. Fisheries Laboratory, Lowestoft, Suffolk, UK.

Davidson NC, Laffoley Dd'A, Doody JP, Way LS, Gordon J, Key R, Drake CM, Pienkowski MW, Mitchell R and Duff KL. 1991. Nature conservation and estuaries in Great Britain. Peterborough, Nature Conservancy Council.

Didžiulis, V. 2013. NOBANIS – Invasive Alien Species Fact Sheet – *Mnemiopsis leidyi* Online Database of the European Network on Invasive Alien Species – NOBANIS www.nobanis.org, [June 2016]

Elith J, Graham CH, Anderson RP, Dudík M, Ferrier S, Guisan A, Hijman RJ, Huettmann F, Leathwick JR, Lehmann A, Li J, Lohmann LG, Loiselle BA, Manion G, Moritz C, Nakamura M, Nakazawa Y, Overton JM, Peterson AT, Phillips SJ, Richardson K, Scachetti-Pereira R, Schapire RE, Soberón J, Williams S, Wisz MS, Zimmermann NE. 2006. Novel methods improve prediction of species' distributions from occurrence data. *Ecography*, **29**: 129-151.Epifanio E. 2013 Invasion biology of the Asian shore crab *Hemigrapsus sanguineus*: A review. *Journal of Experimental Marine Biology and Ecology* **441**: 33-49.

Global Invasive Species Database. 2015. http://www.issg.org/database/welcome/ [2014/2015].

GBIF. 2015. GBIF.org (27th August 2015) GBIF Occurrence Download http://doi.org/10.15468/dl.or5iyi; http://doi.org/10.15468/dl.3jk6le; http://doi.org/10.15468/dl.yp87nl; http://doi.org/10.15468/dl.tzhek4; http://doi.org/10.15468/dl.7hnz9w; http://doi.org/10.15468/dl.rztjy0; http://doi.org/10.15468/dl.vt985x; http://doi.org/10.15468/dl.vx3ycm; http://doi.org/10.15468/dl.7ji3nw; http://doi.org/10.15468/dl.vxqqyi; http://doi.org/10.15468/dl.qnyex8; http://doi.org/10.15468/dl.w1uljq; http://doi.org/10.15468/dl.gsotxi; http://doi.org/10.15468/dl.ahx5nb; http://doi.org/10.15468/dl.izebvc; http://doi.org/10.15468/dl.a3fjvj; http://doi.org/10.15468/dl.1umi2u; http://doi.org/10.15468/dl.9dnbrz; http://doi.org/10.15468/dl.7hjd20; http://doi.org/10.15468/dl.ffnudw; http://doi.org/10.15468/dl.pgpxwk; http://doi.org/10.15468/dl.2wazhc; http://doi.org/10.15468/dl.gveic9; http://doi.org/10.15468/dl.xof2li; http://doi.org/10.15468/dl.um6ght.

Goldberg CS, Strickler, KM, Pilliod, DS. 2015. Moving environmental DNA methods from concept to practice for monitoring aquatic macroorganisms. *Biological Conservation* **183**: 1-3.

Gordon C, Cooper C, Senio, CA, Banks H, Gregory JM, Johns TC, Mitchell JFB, Wood RA. 2000. The simulation of SST, sea ice extents and ocean heat transports in a version of the Hadley Centre coupled model without flux adjustments. *Climate Dynamics* **16**: 147-168.

Hall Spencer JM, Allen R. 2015. The impact of CO_2 emissions on 'nuisance' marine species. *Research and Reports in Biodiversity Studies* **4**: 33-46.

Hobday AJ, Pecl GT. 2013 Identification of global marine hotspots: sentinels for change and vanguards for adaptation action. *Reviews in Fish Biology and Fisheries* **24**.

Herbert RJH, Roberts C, Humphreys J, Fletcher S. 2012. The Pacific Oyster (*Crassostrea gigas*) in the UK: Economic, Legal and Environmental Issues Associated with its Cultivation, Wild Establishment and Exploitation. Report for the Shellfish Association of Great Britain.

Herborg L-M, Rudnick DA, Siliang Y, Lodge DM, MacIsaac HJ. 2007. Predicting the range of Chinese mitten crabs in Europe. *Conservation Biology* **21**: 1316-1323.

Hollasch S, Kerckhof F, Craeymeersch J, Goulletquer P, Jensen K, Jelmert A, Minchin D. 2015. Alien Species Alert: *Ensis directus*. Current status of invasions by the marine bivalve *Ensis directus.* ICES Cooperative Research Report No. 323. 32 pp.

Holt JT, James ID. 2001. An s coordinate density evolving model of the northwest European continental shelf - 1, Model description and density structure. *Journal of Geophysical Research-Oceans* **106**: 14015-14034.

Holt JT, James ID, Jones JE. 2001. An s coordinate density evolving model of the northwest European continental shelf 2, Seasonal currents and tides. *Journal of Geophysical Research-Oceans* **106**: 14035-14053.

Jarnevich CS, Stohlgren TJ, Kumar S, Morisette JT, Holcombe TR. 2015. Caveats for correlative species distribution modeling *Ecological Informatics* **29**: 6-15.

Jennings S, Brander K. 2010. Predicting the effects of climate change on marine communities and the consequences for fisheries. *Journal of Marine Systems* **79**: 418–426.

Jones MC, Dye SR, Pinnegar JK, Warren R, Cheung WWL. 2013. Applying distribution model projections for an uncertain future: the case of the Pacific oyster in UK waters. *Aquatic Conservation: Marine and Freshwater Ecosystems* **23**: 710-722.

Kettunin M, Genovesi P, Gollasch S, Pagad S, Starfinger U, ten Brink P, Shine C. 2008. Technical support to EU strategy on invasive species (IAS) - Assessment of the impacts of IAS in Europe and the EU (final module report for the European Commission). Institute for European Environmental Policy (IEEP), Brussels, Belgium. 44 pp. + Annexes.

Larsen KMH, Gonzalez-Pola C, Fratantoni P, Beszczynska-Möller A, Hughes SL (Eds). 2016. ICES Reprot on Ocean Climate 2014. *ICES Cooperative Research Report* No. 329. 139 pp.

Libralato S, Caccin A, Pranovi F. 2015. Modelling species invasions using thermal and trophic niche 2009dynamics under climate change. *Frontiers in Marine Science* **2**: 1-14.

Maggs CA, Mineur F, Bishop JDD, McCollin T. 2010. Non-natives in MCCIP Annual Report Card 2010-11 *MCCIP Science Review* 11pp. www.mccip,org.uk/arc

MCCIP. 2013 Marine Climate Change Impacts Report Card 2013 (Eds. Frost M., Baxter J.M., Bayliss-Brown G.A., Buckley P.J., Cox M., Withers Harvey N.) Summary Report, MCCIP, Lowestoft, 12pp.

Mercks B, Steyaert M, Vanreusel A, Vincx M, Vanaverbeke J. 2011 Null models reveal preferential sampling, spatial autocorrelation and overfitting in habitat suitability modelling. Ecological Modelling. 222: 588-597.

Molnar JL, Gamboa RL, Revenga C, Spalding, MD. 2008 Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment* **6**: 485-492.

Natural England. 2009. Horizon scanning for new invasive non-native animal species in England. Natural England Commissions Report NECR009. http://publications.naturalengland.org.uk/publication/43005 [25 November 2015].

NOBANIS. 2015. NOBANIS European Network on Invasive Alien Species https://www.nobanis.org/ [25 November 2015].

NNSS. 2014. Great Britain Non-native Species Secretariat Species Information Portal http://www.nonnativespecies.org/factsheet/index.cfm [January to December 2014].

Oliviera OMP. 2007 The presence of the ctenophore *Mnemiopsis leidyi* in the Oslofjorden and considerations on the initial invasion pathways to the North and Baltic Seas. European Research Network on Aquatic Invasive Species. *Aquatic Invasions* (2007) **2:** 185-189.

Padalia H, Srivastava V, Kushwaha SPS. 2014 Modeling potential invasion range of alien species, *Hyptis suaveolens* (L.) Poit. In India: Comparison of MaxEnt and GARP. *Ecological Informatics* **22**: 36-43.

Paravincini V, Azzurro E, Kulbicki M, Belmaker J. 2015. Niche shift can impair the ability to predict invasion risk in the marine realm: an illustration using Mediterranean fish invaders. *Ecological Letters* **18:** 246-253.

Pearson RG, Raxworthy CJ, Nakamura M, Peterson AT. 2007 Predicting species distributions from small numbers of occurrence records: a test case using cryptic geckos in Madagascar. *Journal of Biogeography* **34**: 102-117.

Phillips SJ, Anderson RP, Schapire RE. 2006. Maximum entropy modelling of species geographic distributions. *Ecological Modelling* **190**: 231-259.

Pinnegar JK, Tomczak MT, Link JS. 2014. How to determine the likely indirect food-web consequences of a newly introduced non-native species: A worked example *Ecological Modelling* **272**: 379-387.

Pope VD, Gallani PR, Rowntree PR, Stratton RA. 2000. The impact of new physical parametrizations in the Hadley Centre climate model: HadAM3. *Climate Dynamics* **16**: 123-146.

Python Software Foundation. 2010. Python Language Reference, version 2.7.

R Core Team. 2012. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org/

Reiss H, Cunze S, König K, Neumann H, Kröncke I. 2011. Species distribution modelling of marine benthos: a North Sea case study. *Marine Ecology Progress Series.* **442**: 71-86.

Roy HE *et al*. 2014. Horizon scanning for invasive alien species with the potential to threaten biodiversity in Great Britain. *Global Change Biology* doi: 10.1111/gcb.12603.

RINSE. 2014. http://www.rinse-europe.eu/ [25 November 2015].

The Nature Conservancy. 2008. The Nature Conservancy. Conservation Gateway https://www.conservationgateway.org/Pages/default.aspx [5 January 2014].

Tidbury H, Taylor N, Copp G, Garnacho E, Stebbing P. 2014. Introduction of Marine Non-Indigenous Species into Great Britain and Ireland: Hotspots of Introduction and the Merit of Risk Based Monitoring. Cefas contract report C5995 (Objective 1). Cefas, UK.

Tinker J, Lowe J, Holt J, Pardaens A, Wiltshire A. 2015. Validation of an ensemble modelling system for climate projections for the northwest European shelf seas. *Progress in Oceanography* **1338A**: 211-237.

Tinker J, Lowe J, Holt J, Pardaens A, Barciela R. in review. Uncertainty in climate projections for the 21st century northwest European shelf seas. *Progress in Oceanography.*

UKTAG. 2014. Water Framework Directive UK TAG. Alien Species Alarm List http://www.wfduk.org/resources/alien-species-alarm-list [5 January 2014].

Vierod DT, Guinotte JM, Davies AJ. 2015. Predicting the distribution of vulnerable marine ecosystems in the deep sea using presence-background models. *Deep-Sea Research* **99**: 6-18.

Witbaars R, Duineveld GCA, Bergman M. 2013 The final report on the growth and dynamics of *Ensis directus* in the near coastal zone off Egmond, in relation to environmental conditions in 2011-2012. NIOZ Report 2013-2. 79 pp.

Fig. 1. The extent of the dynamically downscaled regional climate projections.

Fig. 2. Poleward shifts in global habitat suitability, referenced against a baseline from 1995, as predicted for the years 2055 (light bars) and 2085 (dark bars).

Fig. 3. Poleward (northerly) shifts in habitat suitability in the shelf seas area, referenced against a baseline from 1995, as predicted for the years 2055 (light bars) and 2085 (dark bars).

Fig. 4. Habitat suitability (from 0 to 1) within the north-west European shelf seas area for five species with particularly high MI-ISK risk scores, as predicted for the years 1995 (left), 2055 (middle), and 2085 (right). Species, from top to bottom: Pacific oyster *C. gigas*, Asian shore crab *H. sanguineus*, Asian club tunicate *S. clava*, slipper limpet *C. fornicata*, American lobster *H. americanus*.

Table 1. Species selected for distribution modelling. Species of potential commercial value are shown in bold, and species that were not modelled due to insufficient data are highlighted in grey. Information summarised from NNSS (2014), Roy *et al*. (2014), Cefas (2015a) and DAISIE (2015).

Scientific name	Common name	The Nature Conservancy Ecological impact score	DAISIE 100 worst	Cefas Priority list?	Impact Detail	MI-ISK Score
Crassostrea gigas	Pacific oyster	3	Y	Monitoring	**Commercially valuable.**	25
Hemigrapsus sanguineus	Asian shore crab	4	N	Monitoring	One of highest ranking by Roy *et al.*, 2014.	23
Styela clava	Asian club tunicate	4	Y	-	Outcompetes other filter feeders and causes declines in mussel production. Spray causes respiratory condition in humans. Can foul structures, shellfish and fish cages.	23

Crepidula fornicata	slipper limpet	4	Y	Monitoring	High densities, causes trophic competition, reducing growth of commercial bivalves. Changes sediment structure. Reduces diversity of maerl beds. May reduce recruitment of fish. Fouls port structures.	22.5
Asterias amurensis	northern Pacific starfish	-	-	Surveillance	Voracious predator, reducing numbers of native species.	20
Amphibalanus improvisus	bay barnacle, acorn barnacle	4	Y		Dominate community and compete for space and food. Foul native mussels and oysters. Fouls water intake pipes, hulls, structures.	19
Penaeus japonicus	kuruma prawn	3	Y	-	Competes with native prawn species for food and space. May change structure of native benthos, and sediment structure.	19
Tricellaria inopinata	bryozoan	4	Y	-	Outcompetes native bryozoans. Fouls buoys, boats, ropes. Insufficient presence data to model.	19
Ruditapes philippinarum	Manila clam	4	N	-	Outcompetes native bivalves. **Could be commercially exploited.**	17.5
Garveia franciscana	rope grass hydroid	4	N	-	Blocks cooling systems in Chesapeake Bay. Insufficient presence data to model.	17
Mytilopsis leucophaeata	Conrad’s false mussel	3	N	-	Brackish biofouler of coolant systems.	17

Mercenaria mercenaria	American hard-shelled clam	3	N	-	Displaces native clams.	16
Watersipora subatra	bryozoan	-	-	Monitoring	Fouling organism. Insufficient presence data to model.	16
Crassostrea angulata	Portuguese oyster	-	-	Monitoring	**Commercially valuable.** Insufficient presence data to model.	15
Mnemiopsis leidyi	sea walnut, comb jelly	4	N	Surveillance	Reduces species in lower trophic levels, and reportedly can cause collapse of planktivorous fish, dolphins and seals. One of highest ranking in Roy *et al*., 2014.	15
Rapana venosa	rapa whelk	4	Y	Monitoring	Risk to oyster cultures in high densities. May compete with native *Buccinum undatum*. Mussels in Black Sea severely affected. One of highest ranking in Roy *et al*., 2014.	14.5
Ensis directus	American razor clam	3	Y	Monitoring	May impact sediment structure. Shallower water than native species so can affect bathers. Can damage trawls and nets. **Could be commercially valuable.**	14
Didemnum vexillum	carpet sea squirt	4	N	Monitoring	Insufficient presence data to model.	13
Homarus americanus	American lobster	-	N	Monitoring	One of highest ranking future alien invasive species in Roy *et al*., 2014.	13
Bonnemaisonia hamifera	red alga	-	Y	Monitoring	Dominant alga in some regions, outcompeting native species.	N/A

Codium fragile	green alga	4	N	-	Alters benthic communities and increases sedimentation. Fouls shellfish beds, clogs dredges, interferes with nets, jetties etc.	N/A
Sargassum muticum	wireweed	4	N	Monitoring	Outcompetes native seaweeds, fouls harbours.	N/A
Spartina townsendii var. *anglica*	cord grass	4	Y	Monitoring	Environmental modifier. Replaces *S. maritima* and excludes native *Salicornia spp*. and *Zostera spp*.. **Used to stabilise mudflats for land reclamation. May be used as biofuel, paper and animal feed.**	N/A
Undaria pinnatifida	wakame (seaweed)	3	Y	Monitoring	Outcompetes native seaweeds. Can grow on shellfish and impair aquaculture harvests. **Could be commercially valuable.**	N/A

Table 2. The Area Under the Curve (AUC) value, the variable with the highest percent contribution to the model, and the number of presence records used for training

Species	AUC value	Variable with the highest percent contribution	Number of presence records	Number of presence records used for training
A. improvisus	0.989	Near bed temperature	308	136
A. amurensis	0.987	Bathy	95	70
B. hamifera	0.992	Bathy	191	145
C. fragile	0.992	Near bed temperature	202	132

C. gigas	0.988	Near bed temperature	283	147
C. fornicata	0.988	Near bed temperature	394	191
E. directus	0.990	Near bed temperature	266	133
H. sanguineus	0.9980	Bathy	83	38
H. americanus	0.988	Near bed temperature	291	202
M. mercenaria	0.992	Near bed temperature	147	60
M. leidyi	0.994	Near bed temperature	133	86
M. leucophaeata	0.994	Near bed temperature	89	26
P. japonicus	0.985	Near bed temperature	63	32
R. venosa	0.971	Near bed temperature	84	25
R. philippinarum	0.984	Bathy	77	29
S. muticum	0.991	Near bed temperature	230	123
S. townsendii var. anglica	0.995	Bathy	143	78
S. clava	0.990	Bathy	97	65
U. pinnatifida	0.999	Bathy	27	16

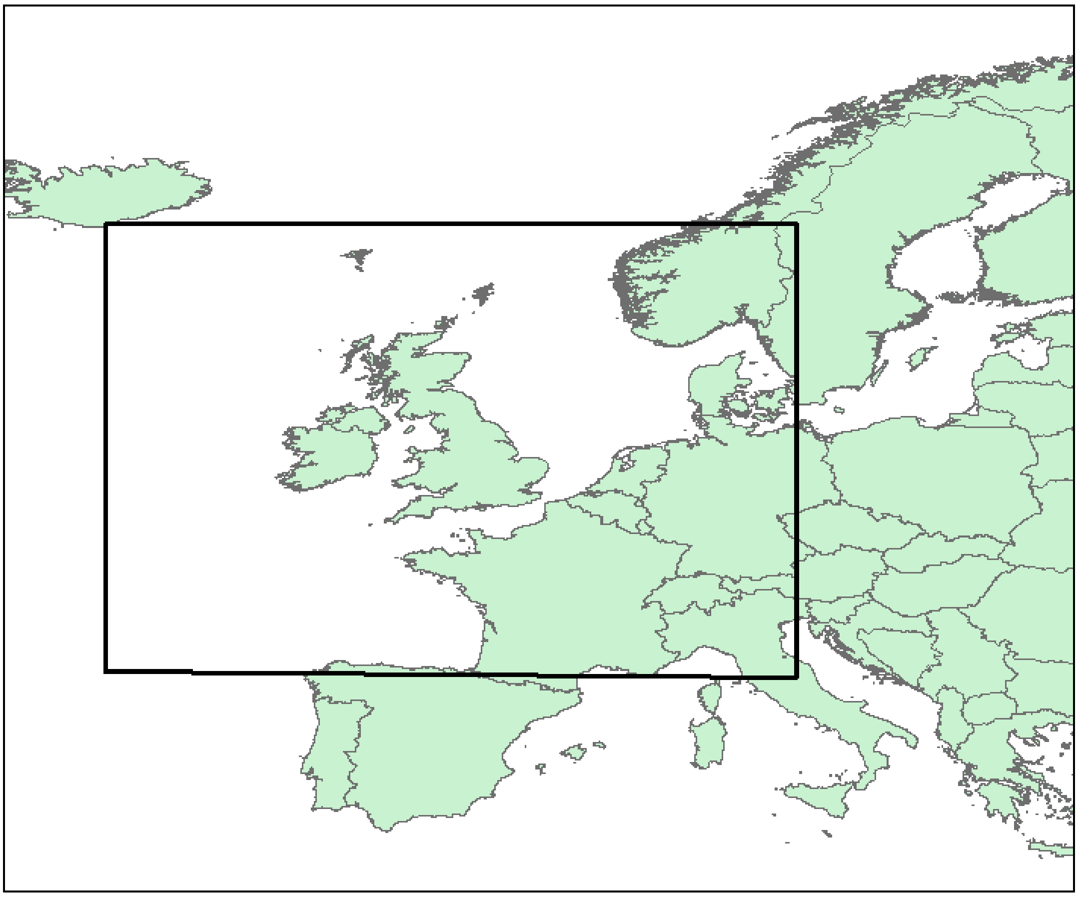

Fig. 1

Extent of the north-west European shelf seas projections.

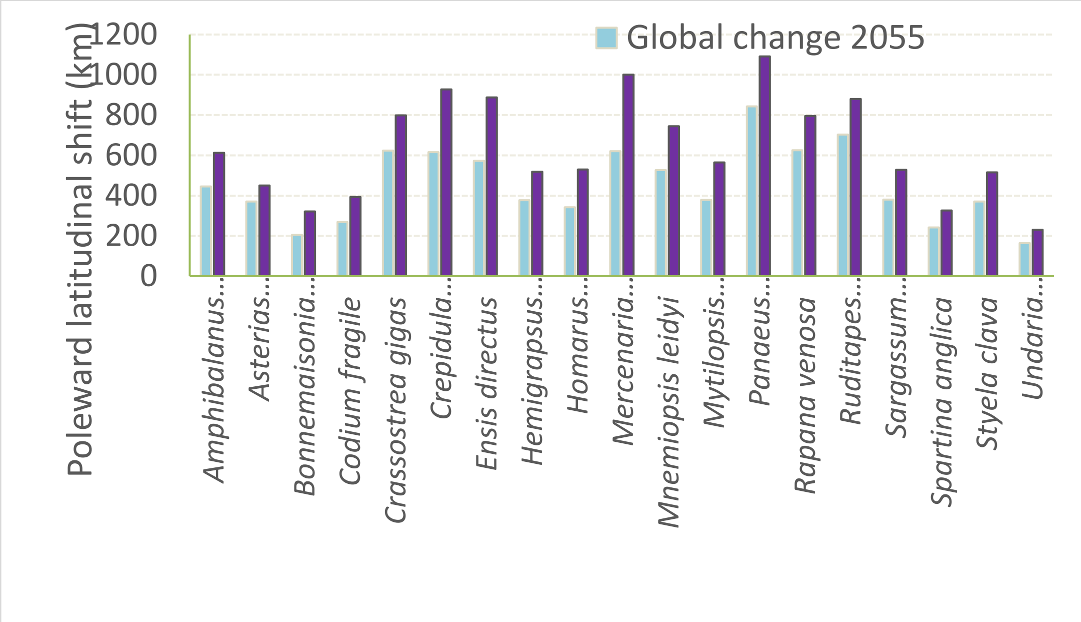

Fig. 2

Poleward shifts in global habitat suitability, referenced against a baseline from 1995, as predicted for the years 2055 (light bars) and 2084 (dark bars).

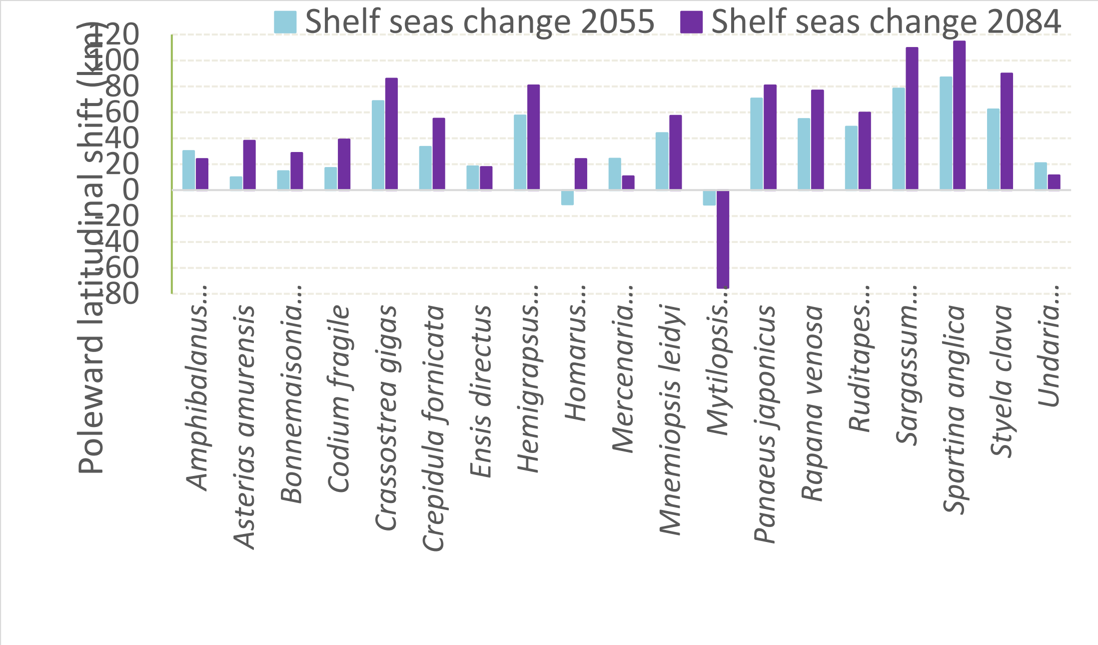

Fig. 3

Poleward (northerly) shifts in habitat suitability in the shelf seas area, referenced against a baseline from 1995, as predicted for the years 2055 (light bars) and 2084 (dark bars).

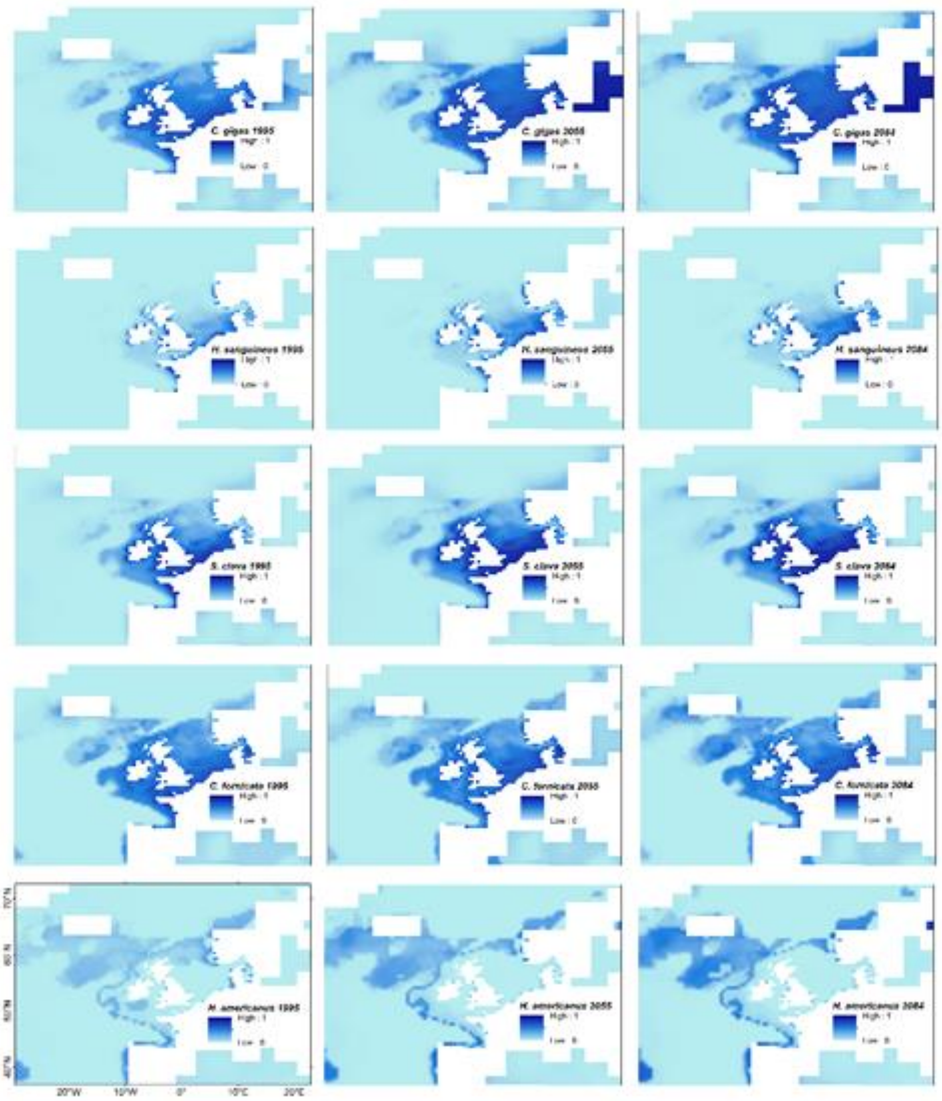

Fig. 4

Habitat suitability (from 0 to 1) within the northwest European shelf seas area for five species with particularly high MI-ISK risk scores, as predicted for the years 1995 (left), 2055 (middle), and 2085 (right). Species, from top to bottom: Pacific oyster *C. gigas*, Asian shore crab *H. sanguineus*, Asian club tunicate *S. clava*, slipper limpet *C. fornicata*, American lobster *H. americanus*.