JNCC Report & Advisory Note

No: 517

Revised Phase III Data Analysis of Joint Cetacean Protocol Data Resources

C.G.M. Paxton, L. Scott-Hayward, M. Mackenzie, E. Rexstad & L. Thomas

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Advisory Note on JNCC projects using the Joint Cetacean Protocol data and their purpose

1. **Background**: Two JNCC contracts have recently been completed that utilise the Joint Cetacean Protocol datasets from around the UK but with very different objectives.

2. The relevant reports are:
   a. Revised Phase-III Data Analysis of Joint Cetacean Protocol Data Resources (Phase-III; Paxton *et al* 2016); and
   b. The identification of discrete and persistent areas of relatively high harbour porpoise density in the wider UK marine area (DHI; Heinänen & Skov 2015 http://jncc.defra.gov.uk/page-6991).

3. **Objectives:** The Phase-III analysis further developed methods for the production of distribution maps and to provide estimates of both abundance and changes in abundance for seven regularly occurring cetacean species. One of the aims was to deliver products that would assist with marine industry Environmental Impact Assessments in UK waters.

4. The DHI analyses helped identify discrete and persistent areas of high harbour porpoise density in the UK marine area. The aim was for the work to assist in the identification of potential harbour porpoise Special Areas of Conservation.

5. **Data similarities:** The Phase-III analysis brought together 38 data sources from at least 542 distinct survey platforms (ships and aircraft) representing over 1.05 million km of survey effort. The first stage of the Phase-III analyses standardised the datasets so that they could be combined for analyses.

6. The standardised datasets of the Phase-III analyses were subsequently used as the input data for the DHI analysis, having first removed some datasets for which permission had not been granted for use in the DHI work.

7. **Differences in analytical models:** The Phase-III analysis is aimed at predicting cetacean abundance from a model that fits observed densities of animals on the European continental shelf (to water depths of 300m) to a small suite of explanatory variables (mainly time and latitude/longitude).

8. DHI developed predictive distribution models based on the relationships between habitats and harbour porpoise density for each Management Unit (MU) (http://jncc.defra.gov.uk/pdf/Report_547_webv2.pdf). The descriptors of habitat included static features, such as water depth and seabed sediment type, and dynamic variables that describe the locations, timing and movement of salinity fronts, eddies and upwelling.

9. **Comparison of results:** The harbour porpoise modelled density surfaces from both analyses are broadly similar.

10. However, in areas of low survey effort, the Phase-III predictions are influenced by the number of porpoises seen whereas DHI predictions are driven by associations between observed numbers and habitat characteristics within the respective MU.

11. DHI predictions are, therefore, more likely to provide a realistic picture of abundance in areas where effort is low.

12. **JNCC Advice**: The approach undertaken by DHI is explicitly for the purposes of identifying habitats that support persistently high densities of porpoise for spatial protection; the approach undertaken by the Phase-III analysis is not suitable for this purpose.

13. With regard future use of the outputs of these analyses for environmental impact assessments, JNCC concludes that the DHI density surfaces better represent the expected distribution and abundance of harbour porpoise for any given area of interest and should, therefore, be used preferentially.

14. JNCC must now seek agreement from data providers to make the DHI density surfaces available for wider use.

15. In the interim, Phase-III density surfaces for harbour porpoises may be used.

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JNCC recommends that before you consult this report you read the guidance in Appendix 7: The Joint Cetacean Protocol Project: current status, applications and limitations

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Executive Summary

Effort-linked sightings data contained within the Joint Cetacean Protocol data resource were used to estimate spatio-temporal patterns of abundance for seven species of cetacean over a 17-year period from 1994 – 2010 over a 1.09 million km^2 prediction region from 48° N to c. 64° N and from the continental shelf edge west of Ireland to the Kattegat in the east. The species were harbour porpoise *Phocoena phocoena*, minke whale *Balaenoptera acuturostrata*, bottlenose dolphin *Tursiops truncatus*, short-beaked common dolphin *Delphinus delphis*, Risso's dolphin *Grampus griseus*, white-beaked dolphin *Lagenorhynchus albirostris* and Atlantic white-sided dolphin *Lagenorhynchus acutus*. There were 38 data sources with data from at least 542 distinct survey platforms (ships and aircraft) representing over 1.05 million km of effort.

The analysis consisted of the following stages, for each species:

1. Estimates were derived of the probability of detecting a group of animals as a function of covariates affecting detectability measured on all surveys. This detection probability had up to three components: (a) probability of detecting a group given that it was available for detection on the surface and assuming all groups on the survey trackline were seen with certainty; (b) probability of detecting a group on the trackline given it was available for detection ("perception bias"); and (c) probability of a group being available for detection ("availability bias"). The first component was estimated using available line transect data, modelling detection probability as a function of available covariates such as group size and vessel type. The second component for some vessel types and species was estimated from a limited amount of double-observer line transect data, as well as previous published analyses. The third component was obtained from a very limited amount of published data combined with expert opinion.
2. The survey data were divided into short (target approx. 10 km) segments of effort. The observed number of animals per segment was converted into an estimated abundance of animals per segment using the estimated detection probabilities/availabilities.
3. The resulting spatially and temporally referenced abundance estimates were modelled as functions of predictor variables including environmental covariates, space and time (day of year and year) sometimes with additional interactions. A recently-developed method was used to deal with complex topographies using spatial predictors.
4. The density surface models then were used to predict over a fine spatial grid (25km^2 resolution) and for one day of each season for each year. Subsets of the grid were used to produce abundance estimates for four large regions and 19 areas of commercial interest.
5. Estimates of uncertainty in the predictions were generated using a parametric bootstrap procedure.
6. Temporal trends were estimated for each region by estimating an average summer abundance for each of three time periods, corresponding with the EU Habitats Directive reporting periods (1994 – 2000, 2001 – 2006, 2007 – 2010). Results were summarized as the average annual rate of population change between periods, using the most recent period as the baseline.
7. The estimated variability about the trend estimates (derived from the parametric bootstrap) was used in a retrospective power analysis to estimate the rate of population change that is detectable with high power (0.8, given an α-level of 0.05).

The modelling of detection probability and availability bias proved successful in allowing the generation of spatially referenced densities. The density surfaces proved complex to model, but spatial density surfaces that faithfully reflected the patterns in the input data were generated for all species. A spatio-temporal interaction model was selected for harbour porpoise, allowing for changes in spatial distribution over time. For all other species, the selected models included space and time as separate effects, giving a proportionately constant spatial distribution over time (except for the limited effect of a spatio-temporally varying covariate, sea surface temperature).

The models generally produced realistic density estimates, and estimated abundances over large areas were broadly similar to equivalent estimates from SCANS-II but were normally greater than those of SCANS.

Confidence intervals on the fitted density surfaces were sometimes wide in regions and times where there was little survey effort. Models with a spatio-temporal interaction produced more accurate results, in the sense of allowing spatial distribution to vary over time, but with wider confidence intervals on the associated predictions. This demonstrates a commonly-encountered trade-off between accuracy and precision.

Four species showed detectable temporal trends in their populations when comparing the first and/or second reporting periods with the most recent period: harbour porpoise showed increases in some regions, while minke whale, Risso’s dolphin and white-sided dolphin showed declines.

The power analysis indicated that in almost all cases annual population changes of between 6% and 40% per year were detectable with good power (>0.8), depending on the species (and for harbour porpoise the region). This means that populations would have to decline to between approximately 10% and 50% of their original size between reporting periods for there to be a good chance of detecting this trend.

It is important to note that there are some issues inherent in the data that make reliable estimation problematic especially at fine scales. Firstly, we have attempted to standardize surveys by converting effort-linked counts into abundance estimates, using data from surveys where a line transect protocol was followed, and by obtaining other conversion factors from the literature. Our primary aim was to make the survey counts comparable so that we could model them to estimate spatial and temporal patterns; abundance estimation was not the primary goal. In standardizing the data, we made strong assumptions about factors, such as availability at the surface and how this changes for different pod sizes, for which there is little information. We assumed line transect surveys that collect distance data have the same detection probabilities as surveys that do not record distances, and that detection probabilities within survey type do not change with time. Further we assume distances are accurately recorded and detection probabilities are the same across survey types (i.e. dedicated survey vs. platform of opportunity). If these assumptions are incorrect then the conclusions will not be valid. Secondly, despite having large quantities of data, survey effort is distributed very patchily over space and time, and there are significant spatio-temporal gaps (particularly in autumn and winter seasons) and confounding between survey type and location/time. For these reasons, we do not believe our estimates of abundance can be as reliable as those coming from a well-designed dedicated abundance survey. Although the Joint Cetacean Protocol data resource is a highly valuable dataset, it should be complemented by regular, large-scale designed surveys for reliable estimation of region-wide abundance.

There are six appendices to this report. Appendix 1 describes how to submit data to the Joint Cetacean Protocol data resource. Appendix 2 gives details of the component datasets of the Joint Cetacean Protocol data resource. Appendix 3 gives the parameters of the detection function models. Appendix 4 gives example plots of the estimated segment-level densities and predictions. The last two appendices were added during revisions to the original report. Appendix 5 gives an examination of the accuracy of estimates, in terms of the average size of model residuals, as a function of the area over which inferences are made. Appendix 6 demonstrates a possible alternative output from the analysis: proportion of the population in some larger management area that is estimated to be within smaller areas of developer interest. These latter outputs may potentially be used to provide context to a proposed development, for example.

How to Use the Results from This Document

Our results are provided in three forms: maps of density, numerical estimates of abundance for particular species-location combinations, and rates of change between reporting periods. A fourth output was added during revisions: numerical estimates of the proportion of the total estimated abundance for a cetacean management unit (IAMMWG, 2015) that is estimated to be within a smaller area of interest (see Appendices 6 and 7).

Graphical depictions of cetacean density (e.g. Figures 11 – 18 & Appendix 4) are provided along with their cell-by-cell upper and lower 95% confidence limits with a map of the adjusted counts for each species used as input to the models; so all maps are presented with four panels. The point estimates of density should be assessed in the context of the associated uncertainty (depicted in the confidence interval maps) as well as the fidelity of the modelled spatial pattern to the adjusted counts. It is evident in each instance that the predicted spatial patterns do not exactly mimic the adjusted counts upon which the models were based. That is to be expected because the plots of the adjusted counts depict the result of samples of the population; our models are intended to discern the underlying general processes that give rise to the pattern seen in the sampled sightings rather than local fine scale processes. Our modelling philosophy was to employ smooths to the adjusted counts rather than accommodate the sharp discontinuities present in the data. A consequence of this philosophy is that small scale differences in cetacean density are smoothed over. This enables us to fill in the noteworthy gaps in data over space and time, but inhibits our ability to make inferences about cetacean densities at small spatial scales.

The numerical estimates provided in Tables 10 – 16 are similar to those of SCANS-II but higher than those of SCANS but always with greater uncertainty. Surveys specifically designed for large-scale abundance estimation, such as SCANS, should produce more reliable estimates of abundance over a wide area for a particular time than those given here, and should be used in preference where possible (see Discussion and Appendix 7). Where such estimates are not available, the abundance estimates given here can be used as a rough gauge of the number of animals likely to be found in delineated areas in the absence of further information. Given the spatial smoothing required to produce these estimates, inferences are unlikely to be reliable at scales of less than approximately 500-1000km^2, and only in areas where data exist. This is confirmed by a preliminary analysis of average residuals as a function of prediction area (Appendix 5). Consequently, we are more confident of the numerical abundances we provide for large areas (e.g. mATL and the OSPAR areas) than for the smaller areas of commercial interest (e.g. the Dogger Bank). In addition, for all species except harbour porpoise (which had an order of magnitude more sightings than the other species), the selected models did not allow for changes in spatial pattern over time; hence the estimated spatial patterns are long-term averages and may not be accurate in any particular short period of time such as a single year.

Another output of our modelling efforts was the measures of population change between Habitats Directive reporting periods. For harbour porpoise, the data supported a spatio-temporal interaction model, and there is the possibility of detecting differences in trends across regions. Based on our results (i.e. the smallest area of conservation interest for which detectable change in harbour porpoise density was noted), we suggest that trends for harbour porpoise might be feasibly estimated at the scale of 570 km^2 over a period of approximately a decade. For the other species, the models fitted here are suitable only for making inferences about population trend over the whole area.

A final output was added during revisions to this document (Appendix 6): numerical estimates of the proportion of the total estimated abundance for a cetacean management unit (IAMMWG, 2015) that is estimated to be within a smaller area of interest (see Appendices 6 and 7). We believe that potential biases in the estimates of absolute abundance may at least partially cancel out in these relative abundance measures, and so they may be more reliable. However, as with the absolute abundance measures, they are unlikely to be useful for areas of less than 500 – 1000 km^2.

Please also refer to Appendix 7 for guidance from the JCP Steering Group on the uses and limitations of the JCP Phase III outputs

1 Introduction

1.1 Background (previous phases)

EU Member States have a legal obligation under Article 11 of the Habitats Directive to undertake surveillance of all cetacean species occurring in their waters to determine their “conservation status”, and to report on this every six years (European Commission 2009). A species is in “favourable” conservation status if: “population dynamics data indicate that the species is maintaining itself on a long-term basis as a viable component of its natural habitats, the natural range of the species is neither being reduced nor is likely to be reduced in the foreseeable future, and there is, and will probably continue to be, a sufficiently large habitat to maintain its populations on a long-term basis.” The exact measures reported are open to interpretation by Member States (see Council of the EEC 1992), but the above guidance leads naturally to a focus on (1) changes in species’ range; (2) trends in species’ abundance (3) designation and monitoring of suitable habitat (albeit not practical for cetaceans); and (4) future species prospects.

In the UK and Ireland, one cost-effective method for addressing the first two of the above measures for marine mammals is the Joint Cetacean Protocol (JCP) data resource. In 2004, a working group was set up to look for ways to update the *Atlas of Cetacean Distribution in North-West European Waters* (Reid *et al* 2003) and from this arose the idea that a publically managed database of voluntarily submitted data would allow consideration of historical trends in species distributions. The JCP data resource has thus developed as a collection of effort-related survey data that have been gathered by various governmental organizations, educational organisations, private sector companies and non-governmental organizations. These data are collected through dedicated surveys, systematic observing from platforms of opportunity, and more casual watches.

Thomas (2009) reviewed the potential of the JCP data resource to allow estimation of trends in species abundance and pointed out that whilst the threshold for detectable population change recommended by the Habitats Directive (monitoring should be able to detect a decline equivalent to 1% per annum) was clearly unrealistic, there was the potential to extract relative trend data from the JCP data resource. He reviewed a number of potential statistical methods that could be used to investigate the data and made recommendations for future research.

Paxton & Thomas (2010) provided a preliminary analysis of the JCP data resource for a core area consisting of the Irish Sea. This Phase I analysis used line transect data from designed and platform of opportunity surveys, where distance sampling data were gathered, and applied the resultant detection probabilities from these analyses to sightings data where distance measurements were not present. Once sightings data had been corrected, a spatio-temporal density surface was fitted to the data for five species using Generalized Additive Models (GAMs, Wood 2011, 2006). Paxton & Thomas (2010) found they could create time series of species abundance for harbour porpoise *Phocoena phocoena*, minke whale *Balaenoptera acuturostrata*, bottlenose dolphin *Tursiops truncatus*, short-beaked common dolphin *Delphinus delphis*, and Risso’s dolphin *Grampus griseus*.

Paxton *et al* (2011) extended the analysis southwards into the Celtic Sea and northwards to the Minches and fitted flexible spatial models in a generalized estimating equation (GEE) context to fit density surfaces to the above species as well as white-beaked dolphin *Lagenorhynchus albirostris* and Atlantic white-sided dolphin *Lagenorhynchus acutus,* and obtained estimates of abundance for the period 1985-2010. Strong seasonality was found for all species but longer term changes were less consistent although decadal large fluctuations in abundance were found for harbour porpoise, minke whale, bottlenose and short-beaked common dolphins.

We expanded on JCP Phase II region and updated the existing data resource to the entire North Sea, to latitude 64°N and to the west to 300 m depth (Figure 1). The species considered for analysis were those that were most common in the JCP data resource: harbour porpoise, minke whale, bottlenose dolphin, short-beaked common dolphin, Risso's dolphin, white-beaked dolphin and Atlantic white-sided dolphin. Estimates of abundance are provided for large spatial regions of interest (OSPAR 2, OSPAR 3, mATL and the UK EEZ, Figure 1, see section 2.1 for an explanation of the acronyms) as well as, where possible, a set of 19 smaller areas of conservation and/or offshore development interest (Figure 2, see section 2.1).

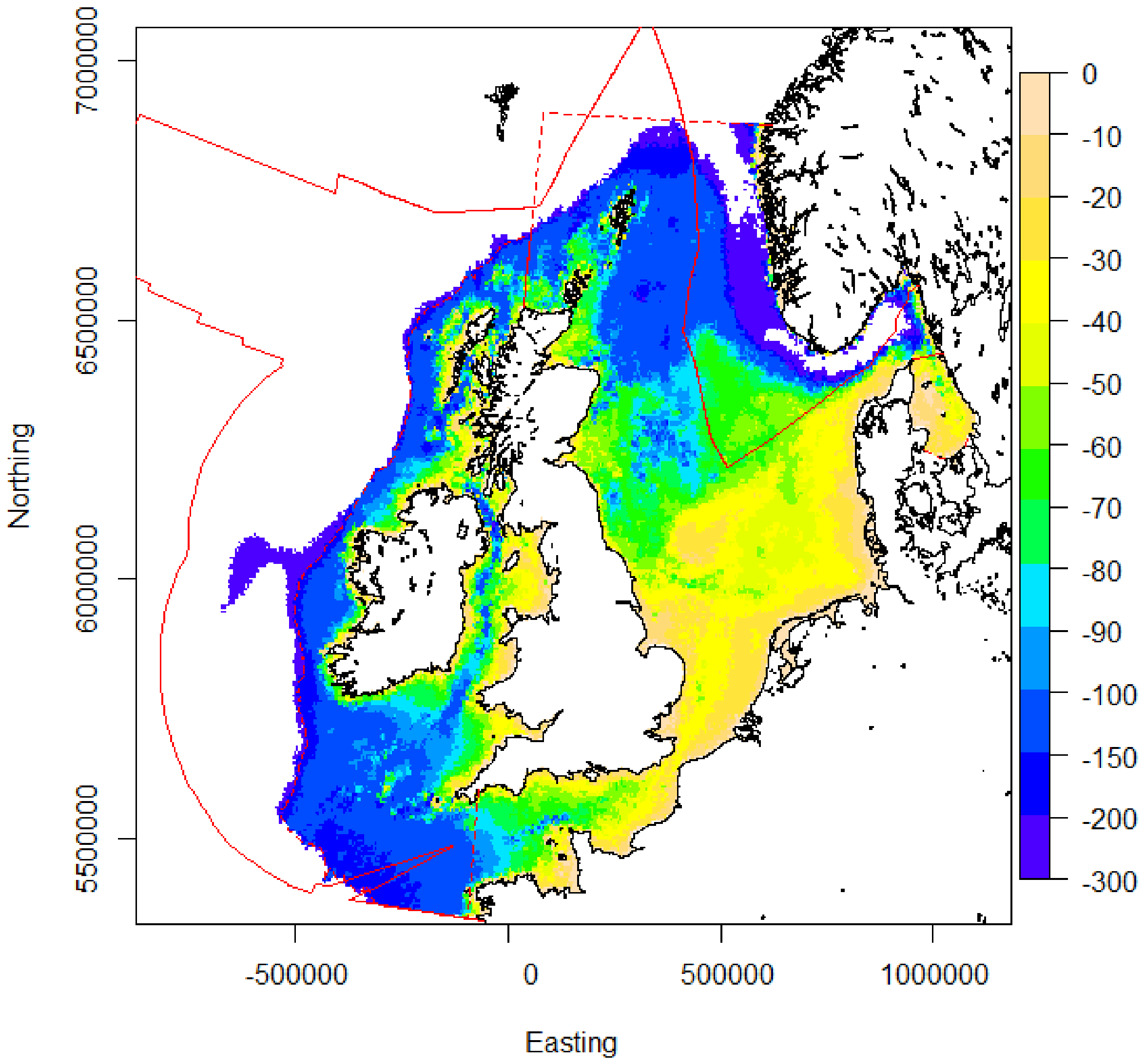

Figure 1. The JCP Phase III region. The coloured area (with depth shaded in m) indicates the region of collected survey effort and the prediction region. The area outlined in red is the mATL biogeographic region. The areas delineated in dashed red are OSPAR 3 to the west and OSPAR2 to the east divided at 5° Lon. W.

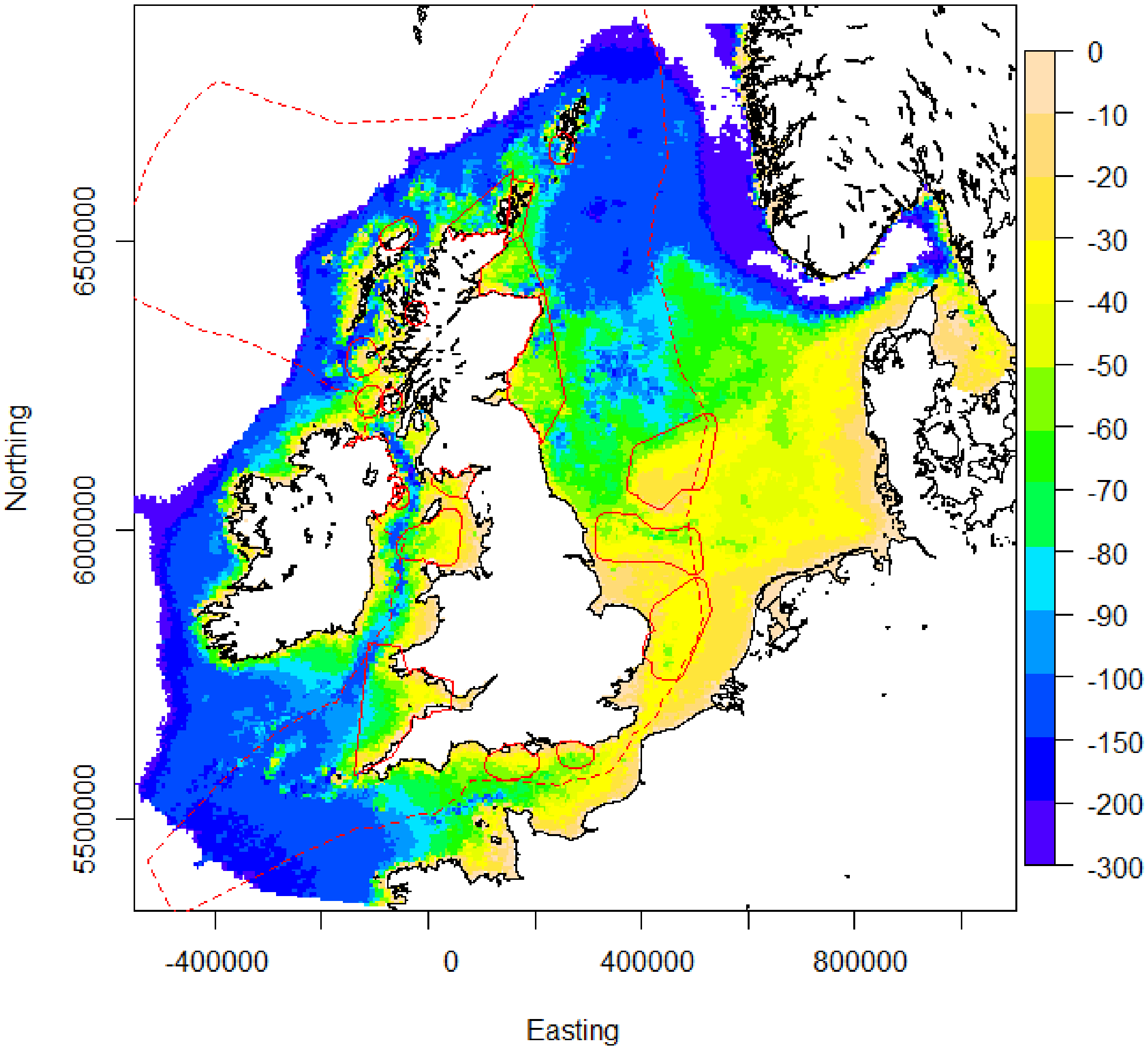

Figure 2. A close up of the core JCP Phase III region showing (red) areas of interest for offshore development where estimates of abundance are of special commercial interest. The red dashed line indicates the British exclusive economic zone (EEZ).

1.2 Purpose of This Analysis

The primary goals of this research are described here, and linked to the parts of the report where these goals are addressed.

1. To produce guidance to individuals/organisations wishing to contribute to the JCP. Details of this are provided in Appendix 1.
2. To identify potential problems in the data. Details of the data with reasons for inclusion are provided in summary in Section 2.1, 2.2.1 and 3.4 with a detailed description of the individual data sets in Appendix 2.
3. To render the different data types comparable. Correction of the supplied data for under-detection and surface availability has been undertaken (see Section 2.3 for methods and Section 3 for results).

4. To produce comparable (relative) density/abundance estimates across various regions (see Sections 2.1 and 2.5, Figure 1 and Figure 2) of scientific and/or commercial interest. Estimates for 2010 can be found tabulated in Section 3.5, along with density surface maps. Further historical maps of density can be found in Appendix 4. Caution should be exercised in the interpretation of the results, as discussed in Section 4. Further relative abundance measures are given in Appendix 6.
5. To provide accessible predictions for consideration by others. The relevant files have been deposited with the JNCC.
6. To determine the power of the final data resource/analyses for detecting changes in abundance and range. Sections 2.7 and 3.7 address this.
7. To produce guidance on how outputs could be used. See above.
8. To suggest the most robust resolution and repeatable methods for the future repeated synthesis of these and future datasets. This is discussed in Sections 4.6 and 4.9, and in Appendix 5.
9. To suggest how power to detect change in distribution and abundance can be improved, e.g. through development of methods and increased coverage. See Section 4, particularly 4.9.

1.3 Framework of the Joint Cetacean Protocol Data Resource Analysis

Here we present a summary of the statistical methods and the general approach used, prior to a more detailed description in the later sections. Figure 3 provides a schematic overview of the stages of the analysis. The data under consideration (see below) consisted of spatially and temporally referenced sightings of cetaceans, coming from a variety of different data sources. For some of the data sources, each sighting was associated with estimated observed distances of the observer to the group of animals seen, allowing estimation of the detection probability. We modelled effort associated sighting data using the number of animals detected in segments of survey effort ("the count method"; see Hedley 2000, Hedley & Buckland 2004, Hedley *et al* 2004).

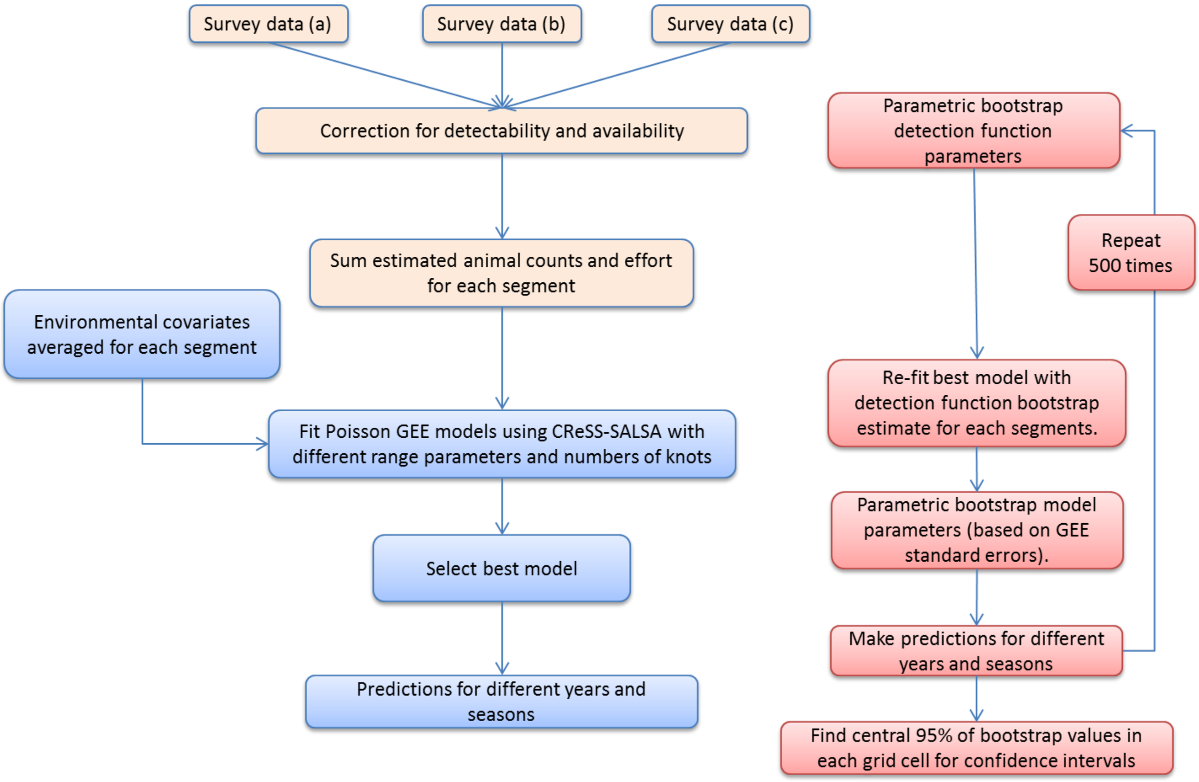

Figure 3. A schematic of the analysis of the JCP data resource.

The aim of this analysis was to estimate the overall relative density of animals in the region of interest, based on the observed numbers seen by the observers and corrected for under-detection (i.e. not all animals at the surface are seen) and availability (i.e. not all animals are at the surface). The first stage consisted of correcting the data both for imperfect detection and availability at the surface; the second stage involved modelling the resultant estimated numbers as a function of space, time and other relevant explanatory variables.

The data did not have to come from dedicated line transect surveys, although there had to be systematic observation associated with defined effort, i.e. both location and whether observers were truly observing should be known, as opposed to casual ad-hoc observations when the observers' primary task is something else. It is well established in the psychological literature that objects can easily be missed under such circumstances (e.g. Simons & Chabris 1999). As detection by observers is always imperfect, correction should be made where possible for missed animals. This can be done in part using distance sampling methods (see Buckland *et al* 2001, 2004) that can infer detectability by considering the distribution of distances to observed animals. Data without distances can be incorporated into the analysis by assuming that detection probabilities obtained from the distance analysis also apply to the sightings without distances. This is further justification for requiring dedicated survey effort from the surveys that do not include distances.

In the simplest case, detectability was estimated assuming detection on the trackline was perfect. If the probability of detection on the trackline, $g(0)$ is actually less than 1 and this can be estimated, or if $g(0)$ actually is 1, then an absolute index of the abundance of animals at the surface can be made (i.e. the actual numbers present at that moment). In this specific context, we considered $g(0)$ as a correction for perception bias only i.e. it is the probability of detecting an animal on the trackline given it is at the surface, not considering, at this stage, animals that are submerged. For some of the data considered here *g*(0) could be estimated and then applied to the subset of data where *g*(0) was unknown and assumed not to be one, so abundance of surface animals could indeed be estimated.

Cetaceans are not always available to be detected. However, if information about diving times is obtainable, a correction for availability can be made. Here availability was calculated as a function of surfacing rates of the animals and the transit time of the viewing vehicle. We caution, however, that knowledge of diving behaviour is very sparse, particularly the level of synchrony of surfacing individuals within groups, and so such corrections are somewhat speculative. For larger slow moving vessels, availability was assumed to be one.

The survey effort was divided into segments and the numbers observed, corrected for detectability and availability, were summed for each segment. Data for different surveys were combined at this stage. This created a spatially referenced density or estimated abundance index (in this case the latter) that could then be modelled in the next stage of the analysis. Our primary objective in making the detectability and availability corrections was to put the count data from different data sources onto the same scale so that they could be combined. The best way to do this is to correct as far as possible to absolute density or abundance. However, this required us to assume that animals counted on surveys without distances have the same detectability as those on dedicated line transect surveys; it also required us to make strong assumptions about availability. For these reasons, we do not anticipate that the estimated numbers produced at this stage will be as reliable as those arising from surveys designed from the ground up estimate abundance. See Section 4.9 for further discussion of this.

The abundance modelling methods (Complex Region Spatial Smoother, CReSS, Scott-Hayward *et al* 2011 and Spatially Adaptive Local Smoothing Algorithm, SALSA, Walker *et al* 2011) used here accounted for the spatially complex patterns in animal distribution, hugely variable animal counts across the survey area and spatio-temporal autocorrelation. The resultant models were used to interpolate into regions and times without survey data, creating estimated density surfaces over a grid of temporally geo-referenced points. Additionally, to allow for inter-annual changes in spatial distribution, the data were modelled using a (relatively smooth) spatio-temporal interaction where possible.

Uncertainty in the modelling process was incorporated using parametric bootstrap techniques (Davison & Hinkley 1997). Values were simulated from the fitted parameters of the detection functions to create a new simulated density data set, which was then spatially modelled as described above. Values were then simulated from the fitted parameters of the density surface models leading to replicate model prediction surfaces with which percentile confidence intervals could be constructed. There was an additional nonparametric component in that inclusion of sightings without distances was decided by sampling (see Section 2.2.3). This two stage process resulted in 500 density estimates for each grid cell (given a set of covariates and point in time) and the central 95% of these values were used to define the upper and lower 95% confidence limits in each case (i.e. by identifying the 2.5 and 97.5 percentiles).

The fitted density surfaces were used to estimate abundance for a set of 23 regions of interest (four large regions and 19 smaller localities of commercial or conservation interest), by summing densities in grid cells within each relevant region, and multiplying by grid cell area. Ninety-five percent confidence intervals on these abundances were derived by taking the lowest 2.5th percentile and highest 97.5th percentile as before.

Temporal trends were estimated for each region by estimating an average summer abundance for each of three time periods (1994 – 2000, 2001 – 2006, 2007 – 2010), corresponding with Habitats Directive reporting periods. Results were summarized as the average annual rate of population change between reporting periods, using the most recent period as the baseline. The estimated coefficients of variation from this analysis were used in a statistical power analysis to determine what rates of population change are detectable for each species with good power (target power of 0.8).

ESAS protocol boat data

ESAS data were collected either by boat or aircraft, although no sightings of the species analysed in this phase were made during the aerial effort. Sightings were allocated to one of four distance bins (0 – 50 m, 50 – 100 m, 100 – 200 m 200 – 300 m) from the track line. The ESAS boat data were combined with the E.ON Atlantic Array seabird sighting boat survey data, the Mainstream RP boat survey data and the SMartwind data, all of which were collected using the ESAS survey method. Therefore, the data were treated as single platform distance data with initially 300 m truncation. Some data were not binned into distance categories but were coded as only within 300 m. Other data had no distances at all. Fitting a realistic detection function over more than 100 m proved problematic for harbour porpoise and dolphins, so the data for these species were right truncated at 100 m. Some ESAS surveys, unlike all the other surveys considered here, are undertaken from one side of the platform only. Effort (see below) was altered to consider this.

2 Methods

2.1 Overview of Data

The core region of interest, which is shown by the coloured area in Figure 1, covers the epicontinental North Sea and beyond with boundaries corresponding to either the marine Atlantic biogeographic region (mATL) or to the Oslo and Paris Conventions for the protection of the marine environment of the North-East Atlantic region (OSPAR) (Ministerial Meeting of the Oslo and Paris Commissions 1992) boundaries except in the west and north (see below) where the boundary is defined by the 300 m depth contour. The southern boundary is at 48° N corresponding to the southern boundary of OSPAR regions 2 and 3 and the northern boundary (which corresponds to the northern boundary of the mATL) is mostly at 62° N. The eastern boundary corresponds to the eastern boundary of OSPAR region 2. Thus predictions could be made for a subset of OSPAR2, OSPAR3 and the mATL region to 300 m depth. Within the covered region are parts of the exclusive economic zones of Ireland, the UK (including the Channel Islands and the Isle of Man), Norway, Sweden, Denmark, Germany and France. The EEZs of the Netherlands and Belgium are entirely within the covered region. Predictions can be made of the British EEZ to 300 m depth.

Sightings and effort data were available for all of the above regions but with much of the effort concentrated within the North Sea, English Channel and Celtic Sea - Irish Sea - Minch axis. The available data span was from 1968 to 2010, but with no effort on the periphery of the shelf between 1969 and 1984. The first large scale synoptic survey was in 1994 (SCANS, Hammond *et al* 2002), so only data from that year onwards were considered here.

A large variety of data was available but only a subset of this was amenable to analysis within the time frame of the study. Only ship and aircraft based data were used. All ship and aircraft sightings required effort (i.e. times and locations of observing) to make them usable in the analysis and observers had to have observation as their primary task when on effort. There were 38 distinct datasets in Phase III that met these criteria. These data sets contained data from *at least* 542 vessels (not all vessels are identified by name). Duplicate data were identified and deleted. The individual data sets are described and summarised in Appendix 2. In terms of information, the sightings data could be classified to one of four classes:

- sightings from a line transect survey, with distances from two observers (“double platform configuration”) allowing estimation of $g(0)$, the probability of detection on the trackline, for an available pod;
- sightings with distances from a single observation platform;
- sightings from count only data collections with no distances;
- data from aerial imaging surveys where numbers at the surface are, in theory, known without error

Effort data were checked for anomalous positions and speeds. There was insufficient time to consider how erroneous reporting of platform position could affect effort. Data with noticeably erroneous positions that could not be easily corrected (i.e. by reference to adjacent effort) were removed.

To be included in the analysis, segments of effort had to be within the geographic region of interest, within the time period of interest, not be associated with an anomalous speed greater than those possible by the participating vessel (based on consultation with the data suppliers and also less than 75 km/h in the case of aeroplanes) and not be based on stationary observations on land or sea. Anomalous speed must be caused by erroneously recorded position or time data. Effort operating at speeds less than 5 km/h for shipboard surveys was also removed. Because of a scarcity of available effort in deeper waters, only waters at depths of 300 m or less were considered in the analyses, and any data from deeper waters were excluded.

In the case of harbour porpoise, because of their low detectability at even moderate sea states (e.g. Laake *et al* 1997, Northridge *et al* 1995 & Barlow 1988), only effort and sightings data recorded during sea state 2 or less were used.

2.2 Detection Function Modelling

2.2.1 Sightings Classes

Surveys were carried out using a variety of platforms, from rigid inflatables and other small boats to ferries and aeroplanes. Sightings data that had the same truncation distance, type of platform and binning regime were grouped (see below for details) to estimate detection probability. Data were only considered together if they came from the same *survey mode* (boats as opposed to aeroplanes) with identical truncation with identical binning. Three classes of boat were classified: *littleboats* (observer eye height < 5 m above the water level, *bigboats* (observer eye height between 5 and 10 m above water level) and *ferries* (observer eye height > 10 m above the water). This enabled a single platform function (see below) to be fitted to these data with boat class considered as a covariate. Note that *ferries* were not necessarily functionally ferries, they could be large research boats. Also observer eye height was not always available for all platforms so heights, and hence *VesselType* had to be occasionally inferred. Some data classes did not have detection functions fitted because perpendicular distances were not available. We briefly describe the sightings data classes below.

Swansea University and University of Aberdeen Surveys

These continuous distance aerial data were considered combined as single platform data. For harbour porpoise and dolphins the data were left truncated at 100 m and right truncated at 400 m. However because of a paucity of sightings this function could not be fitted for minke whales and so a strip transect was assumed.

Wildfowl and Wetlands Trust

The WWT data aerial data were treated as distinct from the university aerial surveys because of the unique bins into which the distances were recorded. The data were divided into two groups dependent on whether the distances were binned in 2 or 3 categories (old: 44 – 163 m, 163 – 426 m, new: 44 – 163 m, 163 – 282 m, 282 – 426 m). A distinct single platform detection function was created for each set of data. The data for both functions were left truncated at 44 m because the region closest to the aeroplane could not be observed.

Other boat data

The remaining boat sightings, from all the other surveys (including SCANS, SCANS-II, CODA etc.) were collectively analysed as single platform data although a subset of these data was collected in trial mode (see below) allowing estimation of $g(0)$.

SCANS & SCANS-II aerial surveys

In the case of these data sets, the data were amalgamated and treated as single platform distance data allowing estimation of a detection function. The exception to this was for the harbour porpoise sightings which were treated differently (see below).

Aerial Photo Surveys

In the case of these data, no detection function was fitted and the detection probability was assumed to be one. The width of sea photographed varied within and between surveys.

2.2.2 Fitting Detection Functions

When fitting detection functions, the effects of covariates, other than perpendicular distance, were incorporated into the detection function model ('Multiple Covariate Distance Sampling', MCDS, Marques & Buckland 2004, Marques *et al* 2007). The probability of detection becomes a multivariate function, *g*(y,v), representing the probability of detection at perpendicular distance y and covariates v (v = v1,..,vQ where Q is the number of covariates). Using either a hazard-rate (1-exp(-y/σ)-b) or half-normal detection function ($\exp(-y^2/2\ \sigma^2)$) the covariates were incorporated via the scale term, σ, where for sighting j, σ has the form:

$$\sigma_j = \exp\left(\beta_0 + \sum_{q=1}^{Q}(\beta_q v_{j\,q})\right)$$

where b, β_0 and β_q (q=1,…,Q) are parameters to be estimated (Marques & Buckland 2004). Separate detection functions were generated for harbour porpoise, minke whale and dolphin species (i.e. the five dolphin species combined, although species was used as a potential covariate in this case, see below).

Potential covariates were Beaufort sea state *SeaState*, cetacean group size *Size* and *VesselType* in the case of boat surveys. *Survey* (i.e. the survey the data came from) was also considered as a covariate in some analyses. In the case of the collective dolphin detection function, *Species* was also considered as a factor. The number of observers was not considered as an independent variable (and was often not available), but was presumably correlated with the size of the boat so was partially accounted for by *VesselType*. Also inter-observer differences in detection could not be considered. On the rare occasions *SeaState* was not available for a given sighting, *SeaState* was taken from adjacent effort. Occasionally *Size* was missing from a sighting in which case a *Size* was taken as the mean of all relevant sightings.

The *R* (R Developmental Core Team, 2011) library mrds v.2.0.1. (Laake *et al* 2011, Burt *et al* in prep.), was used for fitting and selection of detection functions.

2.2.3 Detections Without Distances

For individual detections or surveys without distance measurements, the estimated detection probabilities were obtained from the detection function for the appropriate covariate combination. This assumes that these surveys have the same detection probability, given their covariates, as those with distances. A proportion of sightings with missing distances was discarded at random, this proportion being the same as the proportion of the detections of known distance that were beyond the truncation distance. This is because the detection probability calculated from surveys with distance data are the average probability of detecting an animal group between the trackline and a given truncation distance.

2.3 Adjustment in Addition to Detectability

2.3.1 Perception Bias (*g*(0))

Some of the boat sightings data (i.e. the Sea Watch data collected in 2008 in Cardigan Bay, SCANS, SCANS-II, CODA and a survey in the Atlantic Array) were collected under a double platform (Laake & Borchers 2004) protocol. The Cardigan Bay survey had independent observers and thus allowed the opportunity to estimate *g*(0), the probability of detection on the trackline, for the *littleboats* used in the survey. The SCANS, CODA and Atlantic Array surveys with data collected in trial (or also called Buckland-Turnock) mode (Buckland & Turnock 1992), with a tracker platform observing at a distance and a primary searching normally; this allowed estimation of a *g*(0) for *bigboats*.

The double platform data, suitably edited, were amalgamated with the remaining single platform boat data to estimate a single platform detection function for the vast majority of reported sightings (see non-ESAS boat section above).

In the case of the SCANS aerial detections of harbour porpoise, an integrated estimate of overall detection probability incorporating the probability of detection on the trackline and availability, was available (SCANS-II 2008, Hiby pers. comm.). No adjustment for perception bias was made for the other aerial surveys.

2.3.2 Availability Bias

Surfacing and Diving Times

The estimated detection probabilities consider the probability of detection, given that the animals are at the surface to be observed. To obtain an estimate of the total population of cetaceans, the proportion of animals available at the surface has to be considered. An index of availability at the surface for each sighting was made by considering the reported proportion of time the animals spend at the surface. The probability of an individual being available at the surface was given by

$$P(Avail) = \frac{E[s]}{(E[s]+E[d])} + E[d] \times \frac{(1-e^{-\frac{t}{e[d]}})}{(E[s]+E[d])}$$

after Laake *et al* (1997) where s = surface time, d = dive time and t = window of time during which an animal is within the visual range of an observer. The time period that the animal was within the visual range of the observer was normally taken to be the quotient of the perpendicular truncation distance and speed. However in some cases where the truncation distance was thought to be unrealistically short, a longer distance was used based on consultation with a sample of the data providers (i.e. we asked how far forward the observers were looking).

Given individual availability above, group availability (*Group avail*) was calculated as follows

$$P(Group\ avail) = 1 - (1 - P(Avail))^{Size}$$

where the right hand side represents the probability at least one member of the group is at the surface assuming no synchrony between individuals within the group (i.e. the group is available at the surface to be detected). The availability correction was not applied to *bigboats* and *ferries* where, as per SCANS and SCANS-II (Hammond *et al* 2002, Hammond *et al* in press) we assumed availability was 1 as the availability to observers on these large vessels was assumed to be corrected for by the use of the *g*(0) correction as trackers would see animals at such distance that the abilities of primaries to see the animal would be compromised not only by perception but also by availability.

2.4 Density Surface Modelling

2.4.1 Partitioning Data into Segments

The effort data were provided either as waypoints or as segments of transect lines. In the former case, the data were segmented using the waypoints as end points. Segments were then split or amalgamated to achieve final segments of length as close to 10 km as possible. To be amalgamated into a 10 km segment, the existing segments had to be adjacent in space and time and have identical sea states. Sometimes the target 10 km segment lengths were not achieved. Segments above 15 km in length were split. Ten kilometres was chosen as an appropriate length based on the previous experience of analysing survey data by the authors. Objective criteria for choosing segment lengths are not yet available.

Appendix table A2.1 gives the realized effort (i.e. effort after removal of unusable effort) for each data set used. The estimated number of individuals in each *i*th covered segment, $\widehat{N}_c$ was estimated via a method similar to the Horvitz-Thompson estimator (Horvitz and Thompson, 1952), i.e.:

$$\hat{N}_{ci} = \sum_{j=1}^{n_i} \frac{s_{ij}}{P_{ij} G_{ij} g_{ij}(0)}, \qquad i = 1, \ldots, T,$$

where, for each segment *i* containing at least one sighting, P_{ij} is the estimated probability of detection (from the line transect analysis) of the *j*th detected pod in the segment, n_i is the number of detected pods in the segment and s_{ij} is the size of the *j*th pod. G_{ij} is the probability of the given pod being available at the surface to be seen, and $g_{ij}(0)$ is the probability of the given pod being detected on the trackline. Sightings were allocated to segments by reference to their time of observation.

$\hat{N}_{ci}$, was modelled with a_i (i.e. the area associated with each segment) as an offset (see below). The resulting estimated density of individuals (corrected where necessary for perception and availability bias) in segment *i* $\hat{N}_{ci}/a_i$ is hereafter referred to as $\hat{D}_{ci}$. The total number of segments varied between harbour porpoise and the other species because of the sea state ≤ 2 criteria for inclusion of harbour porpoise data.

2.4.2 Environmental Covariates

Because of the crude resolution of the analyses here, fine scale predictors such as tidal levels, salinity, sediment types, etc. could not be used as predictors. Only predictors that could be adequately assigned to all the available data and for the entire temporal and spatial range of the predictions could be used. Thus the fine scale analyses of habitat preferences as undertaken, e.g. for harbour porpoise (Embling *et al* 2010, Skov & Thomsen 2008) and minke whale (Anderwald *et al* 2012), were not possible in the context of the analysis of the whole JCP.

Covariates considered in the models were *Dayofyear* of survey, *Year* of survey, *Depth, Slope* and *SST* sea surface temperature. Depth for each segment was obtained either from depth data collected on survey itself or from the ETOPO2, 2 minute resolution relief data available from the National Oceanic And Atmospheric Administration http://www.ngdc.noaa.gov/mgg/global/etopo2.html. In the latter case, *Depth*s were selected for each segment based on the nearest great circle distance from the segment to a datum in either one of the two depth sources. *Slope* was estimated from the available depths as function of the north-south and east-west gradient using the standard slope function used by GIS (http://webhelp.esri.com/arcgisdesktop/9.2/index.cfm?TopicName= How%20Slope%20works). *SST* was obtained from the optimum interpolation (OI) sea surface temperature (SST) analysis of Reynolds & Smith (1994, also Reynolds *et al* 2002) available at 1 degree and weekly resolution since 1982 (http://www.esrl.noaa.gov/psd/data/gridded/data.ncep.oisst.v2.html).

2.4.3 Spatial Data Processing

The spatially referenced survey data (segment midpoints) and environmental data (*Depth*, *Slope* and *SST*) were indexed using latitude and longitude, assumed to be in the WGS1984 geo-coordinate system. Coastline information was obtained from ESRI (2002), and boundaries of regions of interest used for estimating abundance from JNCC – again these were stored in WGS1984. For spatial modelling, all of these were projected to the UTM31N coordinate system (Transverse Mercator projection, false easting 500000.0, false northing 0.0, central meridian 3.0, scale factor 0.9996, latitude of origin 0.0, linear unit metre). To reduce the number of waypoints in the coastline file, islands smaller than 10 km^2 were deleted from the file, and a maximum error of 2 km was allowed when projecting from the geographic to projected coordinate system. Geographic processing took place in ArcMap 10 (ESRI 2010) and MapObjects 2.1 (ESRI 2004).

2.4.4 Abundance Modelling

Modelling of estimated numbers in each segment was carried out using a flexible surface fitting procedure which accommodates both local surface features and wide reaching (global) trends. Additionally, the procedure used was designed for areas with complex topography to prevent any ‘hotspots’ in marine mammal densities ‘leaking’ unnecessarily across islands and coastlines. Cetaceans occur at low frequencies so the modelling method used had to account for the low proportion of non-zero data in the models.

As a part of the modelling process careful attention was also paid to quantifying the various uncertainties that arise, in order to help ensure any geo-referenced confidence intervals in the species-specific surfaces are realistic. The modelling process used here accounts for both the uncertainty in the abundance estimates used as inputs (after Distance sampling analysis) and the uncertainty in the model fitting process, including any adjustments for residual spatio-temporal autocorrelation identified in the model residuals. We do not, however, account for uncertainty in model selection, in the sense of which covariates are included in the final model.

Surface Fitting Methodology

The smoothing method implemented here was based on point-to-point (i.e. centres of each segment) distances relevant to the cetaceans. All smoothers operate using distances between points: points deemed to be close together in space are assigned values which are more similar than values assigned to distant points. Most smoothing methods are based on Euclidean/straight-line (‘as the crow flies’) distances however, this can mean that distances between points across islands appear artificially close when there is a greater distance for an animal to travel.

The method implemented here employs geodesic (‘as the whale swims’) distances to ensure distances across islands or around coastlines more closely reflect cetacean swimming distances. Specifically, the smoothing was carried out using local exponential basis functions ($\exp(-d/R^2)$), where d represents pointwise geodesic distance between each observation and each knot location and R is the range coefficient. .

The model flexibility available was necessarily distributed over an extensive spatial area (1.09 million km^2) and time period (17 years) and it was considered likely that the modelled area requires a variable degree of flexibility across the surface. For these reasons, the flexibility in the model surfaces was targeted into areas with the greatest need.

To accommodate the potentially patchy numbers of animals across the survey area, the spatial component to the modelling was undertaken using the CReSS method (Complex REgion Spatial Smoother; Scott-Hayward *et al* in press) with targeted smoothing capabilities using the SALSA method (Spatially Adaptive Local Smoothing Algorithm; Walker *et al* 2011). While the methods used are spatially adaptive (and thus allow a wide range of surfaces to be accommodated) the method itself does not induce local structure in the data and the models considered were given sufficient scope to adequately describe surfaces with extremely local surface features (e.g. patchy surfaces with locally acting hotspots) and/or global surface features (e.g. flat surfaces or far-reaching trends).

A runs test (Mendenhall, 1982) on the Pearson’s residuals from the final harbour porpoise and minke whale models showed significant levels of positive correlation (H0: independent residuals, $p < <0.001$). This means that there are fewer runs of residuals than would be expected (each run is long, resulting in fewer runs) if the residuals were independent. To accommodate this spatio-temporal autocorrelation in model residuals, Generalized Estimating Equations (GEEs) (e.g. Hanley *et al* 2003) were employed. This approach requires the specification of a ‘panel’ variable, and model residuals are permitted to be correlated within ‘panels’ while independence between panels is assumed (Hardin & Hilbe 2002). In this case, model residuals pertaining to segments from the same day of survey from the same observation vessel (survey-day-vessel), were permitted to be correlated while independence was assumed between survey-day-vessels. The GEEs allow the estimation of standard errors to be adjusted for the autocorrelation in the ‘panel’ residuals, and to ensure that model standard errors were not based on an inappropriate correlation structure, empirical standard errors were used; these are robust to mis-specification of the correlation structure and are based on the observed correlation in the Pearson residuals within panels.

We have chosen to model these data using GEEs which are population-average (marginal) models for correlated data which are robust to misspecification of the correlation structure. An alternative, mixed-effects model, approach could have been used instead to model the correlation in the data however these are `conditional’ models and generally speaking are concerned with modelling the `average individual’ (transect-vessel-days in this case) rather than the average of the data as required here which are naturally produced by population-averaged/marginal models.

While one can obtain marginal results from conditional models, the quality of these predictions depends entirely on the validity of model assumptions – assumptions which are very difficult to check in practice. Specifically, generalized linear (or additive) mixed models (GLMMs/GAMMs) are based on strict assumptions which, when violated, can result in both biased parameter estimates (Litiere *et al* 2008, Litiere *et al* 2007, Heagerty & Kurland 2001) (and therefore biased predictions) and biased estimates of variance (Darcy 2013, Jeong 2012) for non-normal data.

These often strict assumptions regarding the random effects are almost impossible to check, and therefore the extent of the bias is typically unknown. This issue is of particular concern for more realistic models with several random effects which typically assume the random effects belong to a multivariate Normal distribution (or even some variant of this). The validity of this assumption is critical when `marginalising' the results – which is what would be required to convert conditional/mixed model results into population averaged results, as desired here. This would involve integrating over the random effects distribution which in practice, could be done by feeding-in/imputing an enormously large number of realisations from the chosen multivariate distribution. This assumption might well be unreasonable and virtually impossible to check, leaving the extent of the bias unknown without carrying out a full scale simulation exercise. In contrast, GEEs (as implemented here) are marginal models which are robust to misspecification of the correlation structure, and so are likely to result in less bias.

The data are estimated counts per segment and are non-negative so Poisson errors with a log link function were assumed. The following equation is an example of the GEE model for one covariate, X_1, and a set of coordinates for each data point.

$$s(\lambda_{ij}) = \log(\lambda_{ij}) = \eta_{ij} = \beta_0 + \beta_1 X_1 + \sum_{t=1}^{T} \delta_t b_{tij}$$

where λ_{ij} is the estimated cetacean count for panel i and time j, η_{ij} represents the additive predictor and b_{tij} represents a set of basis functions (b_{tij}; t = 1; … ; T) for panel i at time j for a two-dimensional smoother. Additional covariates are added prior to the summation.

Due to the high numbers of zeros in the data, the mean-variance relationship for a Poisson model was not likely to hold (i.e. $V(y) >> \mu$). A dispersion parameter, which forms part of the GEE parameter estimation process, adjusts the variance appropriately.

Appropriate fit criterion to govern model selection for GEE models is still an area of active research, and while quasi-likelihood versions of the AIC and BIC statistics can be used (Pan 2000) there is still a great deal of debate about the adjustment of these for the correlation observed in model residuals. For instance, it is well known that model selection in the presence of non-independent residuals is likely to result in over-fitted models and thus any complexity penalty (2 per parameter for the AIC and log(N) for the BIC) needs to guard against this tendency. Further, because correlated data offer less information than independent data, the effective sample size for correlated data is typically less than the apparent sample size (N).

In this analysis, a conservative measure for model selection (a quasi-likelihood analogue of the BIC) was used to guide model selection and guard against fitting models which are too complicated (over-fitting) for the underlying process. The penalty per parameter applied is log(N) based on the apparent sample size, (~12 per parameter in this case); this penalty would have been smaller if the effective sample size had been used instead, and thus it would have been likely to result in more complicated models being selected. *K*-fold cross validation based on folding the data into five sets of correlated panels was also considered for model selection. However, we found in practice this returned overly simplistic models that failed to identify cetacean concentrations in certain areas that persisted over time.

Model Selection

To ensure any signal between the environmental covariates and the response was not inappropriately allocated to the spatial surface, model selection proceeded by first considering the environmental covariates alone, followed by the addition of a spatial surface and then (data-permitting) with a spatio-temporal interaction terms.

Initially, models were fitted to each one-dimensional covariate in turn (*Depth*, *Slope*, *SST*, *Year* and *Dayofyear*) to establish the strength of any relationship between animal abundance estimates and each covariate. These covariates were considered as cubic *B*-splines (see Faraway 2006), except in the case of day of year, which was considered as a cyclic cubic regression spline (see Wood 2006). The SALSA method (governed by the BIC statistic) was used to choose the number and location of knots (with some constraints: *df*=3-5, 1-3 knots). These models combined with GEE-based confidence intervals gave an order of best predictors. Collinearity between predictors was also identified at this stage.

To combine the covariates in one model, covariates were added one by one (conditional on an improvement in the quasi-likelihood based BIC score), using the order of best predictors, to an initial model containing a smooth function for year (in anticipation of the temporal trend assessment to follow the modelling). As each covariate was added, SALSA was used to re-distribute the knots, for the added term (if this was considered necessary using the fit score).

Model flexibility for the spatial smoother is determined by both the number of knots (anchor points/points of maximum flexibility) used for the surface and the range coefficient (R) of each knot (the spatial extent to which each knot influences the fitted surface). The SALSA method was used to target model flexibility and identify the locations on the surface in greatest need. The candidate knot locations were chosen to provide good coverage of the survey area and in this case the candidate knots were sourced from a regular grid of knots (60 x 60 km). SALSA initiates the model selection process by space-filling (Johnson *et al* 1990) some specified number of knots from this grid, and these knot locations are subsequently moved to areas on the surface which are poorly fitted by the model as determined by the designated fit criterion. As a part of the SALSA method, once knot locations were determined, different range coefficients (R) were trialled for each knot and chosen using the same fit criterion. The SALSA algorithm is heuristic and thus does not search the full model space and for this reason, several start points (6-12 knot models, in increments of two) were considered for the numbers of knots in the spatial surfaces.

The BIC score (based on a quasi-likelihood value) was also used to discriminate between four candidate values (R_{min} to R_{max}) trialled for each range coefficient to accommodate a wide variety of local to global smoothing gradients. Values for R_{min} were chosen to achieve basis function values close to zero (i.e. with extremely local influence) and R_{max} was chosen to achieve basis function values close to one (i.e. with global influence).

Once a model containing one and two-dimensional terms was determined, an interaction term between the spatial surface and *Year* was added, should this be deemed necessary using the quasi-likelihood BIC statistic. While there was some re-distribution of some (or all) of the species across time in the corrected counts, the sampling effort was extremely patchy over the 17 years and some areas were only surveyed on a small number of occasions. This made it difficult to fit spatial-temporal interactions for some species. Notably, for some species the fitted surfaces (based on these interactions) showed a greatly improved fit to the data but the uncertainty in these fitted surfaces was prohibitively high. Therefore for those species the spatial-temporal interaction between *Year* and the two dimensional surface smooth was removed from the model and the model refitted.

2.5 Prediction

Predictions from the models were made on a 5 by 5 km resolution easting and northing grid covering the coloured area in Figure 1. If the models contained year or season covariates (i.e. *Dayofyear* or *SST*), the predictions were *Year*/*DayofYear* specific.

Predictions were made from the following regions: *OSPAR2* (not including the deep region to the south-east of Norway (699526 km^2), *OSPAR3* in entirety (364608 km^2), the *mATL* region truncated to 300 m depth (936372 km^2) (Figure 1) and the British *EEZ* up to 300 m depth (538009 km^2) (Figure 2). In addition to the above regions, estimates were made for specific localities of commercial/conservation interest/offshore renewable interest (see Figure 2 and Table 1).

Predictions from the individual grid cells (25 km^2 area) adjusted for the intruded land area were summed to obtain overall estimates for the relevant area of interest.

Seasons for the purposes of the JCP analysis were assumed to be January, February, March (winter), April, May, June (spring), July, August, September (summer) and October, November, December (autumn), therefore predictions were made for day 227 (mid-August) for each year 1994 – 2010 and also for days 45 (mid-February), 136 (mid-May) and 315 (mid-November) as representative of winter, spring and autumn respectively for 2010 (SSTs were means from the relevant week).

Table 1. Description of areas of commercial interest.

Area	***Description***	***Area (km^2)***
Lewis	North of Lewis	1842
Kyle Rhea	Kyle of Lochalsh and environs	307
Argyll Array	Around Tiree	3286
Sound of Islay	Jura and Islay combined	574
Islay	An area to the west of Islay	2081
Solway Firth	Solway Firth	2293
Strangford Lough	Strangford Lough and an area around the Ards peninsular	639
*Irish Sea**	A region between Anglesey and the Isle of Man	8227
Atlantic Array	Bristol channel and adjacent Irish Sea	19649
IOW	a region to the west of the Isle of Wight	4459
Hastings	A region to the south of Sussex	2488
Norfolk Bank	A region to the east of East Anglia	14295
South Dogger Bank	A region south of the Dogger Bank	14265
Dogger Bank,	Dogger Bank	17884
Firth of Forth	An area off the east coast of Scotland	14241
Moray Firth	Moray Firth	7899
North	a region immediately north of Sutherland and Caithness (including the west Orkneys)	6047
East Orkney	the east Orkneys	3027
Shetlands	Southern Shetland Islands	1836

*Not the same as the geographic Irish Sea. From hereon commercial area indicated by italics.

2.5.1 Parameter Uncertainty

The uncertainty in the parameter estimates at both the input stage (i.e. estimating abundances for each segment based on the observed counts) and the modelling stage (i.e. estimating the distribution of these estimates across space and time) was combined using a two-stage parametric bootstrap (Davison & Hinckley 2007) process, with 500 bootstrap re-samples being generated as described below. Model selection uncertainty was not considered in this process.

The first stage captured uncertainty in the corrected counts ($\widehat{D}_c$). For each bootstrap, new estimates of the distance sampling detection probability were generated by sampling from the variance-covariance matrix of the estimated detection function parameters, assuming these parameters followed a multivariate normal distribution. New estimates of $g(0)$ were generated by sampling from a normal distribution with mean and standard deviation taken from the fitted values of $g(0)$. New estimates of availability were generated by sampling the observed surface and/or diving times from a gamma distribution, with mean and standard deviation obtained from reported values in the literature (see section 3.3.2.); for minke whales a nonparametric sampling of reported surface frequencies was made as the raw surfacing data were available (see section 3.3.2) in preference to assuming a distribution. Finally, for surveys that did not contain distance data, the sightings to be truncated were randomly chosen independently for each bootstrap replicate.

The second stage captured uncertainty in the density surface modelling, and involved re-fitting each selected species-specific model for each of the 500 bootstrap replicates generated from the first stage, and from each of these new GEE-based fits, generating a single parametric bootstrap realisation from the model. Specifically, a parametric bootstrap replicate was generated from a multivariate normal distribution with variance-covariance matrix estimated using the GEE-based model each case. In cases where the variance-covariance matrices were not symmetric or positive definite (due to numerical optimisation rather than non-convergence) these matrices were coerced to be symmetric and positive definite to allow random generation from each multivariate normal distribution.

This two-stage process resulted in 500 density/abundance estimates for each grid cell (given a set of covariates and point in time). The central 95% of these values was used to define the upper and lower 95% confidence limits (this is often called the `percentile method').

Abundances could then be estimated for the 23 regions of interest (four large regions and 19 smaller localities of commercial or conservation interest). These were obtained by summing the predicted abundances in the relevant grid cells within each region and calculating 95% confidence intervals on these abundances. This was done using the same procedure as described above, using the central 95% of values from the 500 sets of the geo-referenced predictions.

The bootstrap replicates also formed the basis for uncertainty estimation in the trend analysis (see next section).

2.6 Temporal Trend Estimation

The term “trend” has no objective definition (Thomas *et al* 2004), but is generally defined as smooth change over the long-term, which in the current application means over multiple years. The density surface models contained up to two covariates with inter-annual variation. All models contained *Year*, modelled as a smooth, and in some cases allowed to interact with *Easting* and *Northing* so allowing changes in temporal pattern over space. Models were also given the opportunity to select a smooth of sea surface temperature, *SST*, a covariate that varied over space and time (at weekly resolution). Hence, predictions from the model over particular years and regions can be taken as estimates of long-term trend.

Predicted abundance was calculated for each region of interest (the four larger regions and 19 smaller localities of commercial and conservation interest described in Section 2.5), for each year, using day 227 (i.e. mid-summer).

One issue that arises is how to summarize the trend in a single number, for reporting purposes. This was done here by defining a population change metric: the ratio of estimated average abundance in a baseline period divided by estimated average abundance in some earlier period of interest:

$$\Delta_x^* = \hat{N}_b / \hat{N}_x$$

where $\hat{N}_b$ and $\hat{N}_x$ are the average abundance estimates in the baseline period (*b*) and earlier period of interest (*x*) respectively. A value of 2, for example, indicates a population doubling over the intervening time, while a value of 0.5 indicates a population halving and 1 indicates no change. One disadvantage of such a measure is that its size is in some way related to the number of years in the time intervals; hence it may be better scaled into a measure of average annual rate of change:

$$\Delta_x = \left(\hat{N}_b / \hat{N}_x\right)^{1/(t_b - t_x)}$$

where t_b is the midpoint of the baseline period and t_x is the midpoint of the earlier period of interest. Here, a value of 1.05 would be interpreted as the population growing by an average of 5% per year over the intervening time, while a value of 0.95 indicates a decline of 5% per year on average and 1.0 indicates no change. Note that if each period is a single year, then this measure can also be calculated by taking the geometric mean of the annual population changes. If the periods are more than one year, the average abundance estimated is calculated as the geometric mean of the abundances estimate in the period.

The primary interest is in population change between the three reporting periods: 1, 1994-2000; 2, 2001-2006; and 3, 2007-2010. Given that most interest focusses on recent population numbers, the third period was taken as the baseline, and hence two trend estimates were calculated for each area:

$$\Delta_1 = \left(\hat{N}_3 / \hat{N}_1\right)^{1/11.5}$$

$$\Delta_2 = \left(\hat{N}_2 / \hat{N}_1\right)^{1/5}$$

These are average annual average rate of population change between most recent reporting period and the mid-points of reporting periods 1 and 2.

Estimates of uncertainty in each of the trend estimates were calculated by repeating the analysis with each of the 500 bootstrap re-samples of abundance, and using these to estimate a coefficient of variation in trend and confidence intervals (using the percentile method).

2.7 Power Analysis

Because Δ is the ratio of two zero-bounded random quantities, its distribution is expected to be approximately log-normal. Hence, a simple test for trend is a one-sample, two-sided *z*-test of the null hypothesis that the natural log of Δ is zero (i.e. that Δ is 1.0). Given an estimate of the variance in log(Δ) and the α-level (here assumed to be 0.05) then it is straightforward to calculate the power of the test for various levels of rate of change, Δ, that are considered biologically relevant (the relevant formulae are given in Steidl & Thomas, 2001). Alternatively, given a desired target level of power, the detectable Δ can be calculated. In this report, the latter approach was taken. For the purposes of illustration, the value of rate of population change Δ detectable with a power of 0.8 was calculated for each species, region and pair of reporting periods. The population rate of change could be either a decline or increase (i.e. a 2-tailed test was assumed), but since decline is of more concern, it is the detectable decline that is reported (i.e. largest rate of change less than 1.0 that is detectable with a power of 0.8). The required input, variance in log (Δ), was calculated from the estimated coefficient of variation of Δ using the following relationship:

$$\mathrm{var}(\log(\Delta)) = \log\left(1 + CV(\Delta)^2\right).$$

3 Results

3.1 Sightings

The breakdown of sightings by species within each data set is given in Table 2.

Table 2. Sightings within the truncation distance (see section 3.2) for each species under consideration. Includes distanceless sightings randomly assigned to within truncation distance.

Data set	**Harbour porpoise**	**Minke whale**	**Bottlenose dolphin**	**Short-beaked common dolphin**	**Risso's dolphin**	**White-beaked dolphin**	**White-sided dolphin**
APEM	14	0	0	0	0	0	0
CBMWC	302	1	593	0	0	0	0
Centrica	122	5	2	1	0	0	0
CODA	3	9	3	0	0	0	0
Cornish	26	0	10	23	0	0	0
E.On surveys	7	2	0	52	0	0	0
ESAS	502	148	19	161	7	199	38
ESASaircraft	0	0	0	0	0	0	0
Forewind	50	41	0	0	0	18	0
HIDEF	89	0	0	0	0	5	0
HWDT	1352	200	13	73	14	43	2
IFAW	0	0	0	0	0	0	0
IWDG	19	22	2	121	3	0	0
IWDG Ferry	436	35	5	283	12	0	0
MANW	221	0	0	0	0	0	1
MANX	122	24	0	0	33	0	0
Marinelife	485	65	78	292	15	17	2
Moray Offshore Renewables (MORL) boat	149	19	1	1	1	2	0
NORCET	265	75	56	2	9	52	9
North Wales Windfarm Environmental Assessment	39	0	0	0	0	0	0

Orca	221	48	21	46	5	14	1
Pembrokeshire Porpoise 2007	145	2	0	10	0	0	0
Pembrokeshire Porpoise 2008	52	1	1	10	0	0	0
MainstreamRP	17	0	0	0	0	0	0

Table 2. Cont. Sightings within the truncation distance (see section 3.2) for each species under consideration. Includes distanceless sightings randomly assigned to within truncation distance.

Data set	**Harbour porpoise**	**Minke whale**	**Bottlenose dolphin**	**Short-beaked common dolphin**	**Risso's dolphin**	**White-beaked dolphin**	**White-sided dolphin**
RWE NPower (Atlantic Array)	25	1	0	128	0	0	0
SCANS II	700	97	16	46	6	55	49
SCANS II (air)	421	16	0	36	4	0	0
SCANS	1052	168	11	20	10	108	5
SCANS (air)	120	1	10	0	0	0	0
Sea Energy Renewables Inch Cape Argyll boat	1	0	0	0	0	0	0
Sea Energy Renewables Inch Cape boat	30	13	0	12	0	1	0
Scottish Power & EAOW	43	0	3	0	2	4	0
Smartwind Hornsea	287	12	0	0	0	15	0
SWF	3973	666	2121	690	152	68	13
University of Aberdeen aerial	142	2	15	5	0	1	0
University of Aberdeen boat	52	28	0	0	0	0	0
University of Aberdeen boat SAC	35	8	27	0	0	0	0
University of Aberdeen/MORL/HiDef	57	0	0	1	0	0	0
University of Aberdeen/MORL/HiDef/Crown Estate	14	1	0	0	0	0	0
University of Aberdeen ferry	650	111	19	49	5	6	0
University of Swansea	11	0	4	13	0	0	0
WDCS	12	0	0	252	0	0	0
WWT	7769	39	35	84	6	90	1
Total	**20032***	**1860**	**3065**	**2411**	**284**	**698**	**120**

The breakdown of sightings by species by year is given in Table 3.

Table 3. Sightings within the truncation distance (see section 3.2) for each year and species under consideration. Includes distanceless sightings randomly assigned to within the truncation distance.

Year	**Harbour porpoise**	**Minke whale**	**Bottlenose dolphin**	**Short-beaked common dolphin**	**Risso's dolphin**	**White-beaked dolphin**	**White-sided dolphin**
1994	2071	284	14	100	36	157	6
1995	436	54	12	85	3	25	9
1996	297	52	84	58	30	23	7
1997	383	83	132	73	59	44	22
1998	104	56	43	101	2	33	5
1999	165	42	193	32	23	9	2
2000	186	84	133	43	3	2	1
2001	506	191	207	66	5	16	0
2002	531	88	220	61	11	12	1
2003	839	118	414	84	10	27	2
2004	964	89	133	252	6	12	2
2005	2785	179	269	468	16	69	53
2006	1819	89	263	293	8	36	5
2007	1702	84	237	113	43	22	1
2008	2163	72	431	163	12	21	4
2009	3700	124	191	187	9	129	0
2010	1381	171	54	232	8	61	1
Total	**20032**	**1860**	**3065**	**2411**	**284**	**698**	**121**

Table 4. Sightings within the truncation distance (see section 3.2) for each year and species under consideration. Includes distanceless sightings randomly assigned to within the truncation distance.

Vessel Type	**Harbour porpoise**	**Minke whale**	**Bottlenose dolphin**	**Short-beaked common dolphin**	**Risso's dolphin**	**White-beaked dolphin**	**White-sided dolphin**
Littleboats	6115	853	2756	623	192	125	13
Bigboats	3053	518	63	791	29	341	67
Ferries	2227	430	182	858	53	136	40
Planes	8637	59	64	139	10	96	1
Total	**20032**	**1860**	**3065**	**2411**	**284**	**698**	**121**

3.2 Detection Function Results

3.2.1 Model Selection

Estimated detection probabilities and related statistics are given in Table 5. Parameters of the models can be found in Appendix table A3.1.

3.2.2 Harbour Porpoise

The single platform hazard rate detection function for the non-ESAS boat sightings (*n* = 7706) over a range of 0 – 800 m is given in Figure 4a (goodness of fit test not possible – too many parameters compared with the number of distance bins). Additional variables included in the function were *VesselType* and *SeaState*. Harbour porpoise were more detectable from bigboats than littleboats and increased *SeaState* lowered detectability. For boat sightings collected under ESAS survey protocols, the detection function (*n* = 871) was modelled as a half-normal function with *SeaState* and *VesselType* over the range 0 – 100 m (a goodness of fit test not possible, Figure 4b). There were 153 sightings of harbour porpoise groups in the University of Aberdeen and Swansea University aerial surveys within the truncation boundaries. The model with the lowest AIC was half normal with distance only (goodness of fit: $\chi^2 = 0.30$, df = 1, $P = 0.581$ Figure 4c). In the case of the SCANS and SCANS-II aerial data, an integrated estimate of detection probability, detection on the trackline and availability for each survey was taken from the estimates made in the SCANS reports (SCANS-II 2008, Hammond *et al* 2002) from the method of Hiby & Lovell (1998, Hiby pers. comm.). The overall probability of detection for SCANS was 0.260 over a truncation distance of 500 m, the overall probability of detection for SCANS-II was 0.623 over a truncation distance of 300 m. A variance was available for each estimate which could be incorporated into a future bootstrap. There were a total of 123 group sightings in the original SCANS aerial survey and 420 sightings in the SCANS-II aerial survey.

Table 5. Estimated detection probabilities for the JCP sightings data.

Species	***Survey Analysis (sample size)***	***Truncation Distance (m)***	***Detection Probability***
Harbour porpoise	Non-ESAS boats (7706)	800	0.341 (SE= 0.009)
	ESAS mode boats (871)	100	0.808 (SE= 0.031)
	University aerial (153)	300 (left truncated)	0.565 (SE = 0.042)
	WWT old binning (100)	382	0.597 (SE =0.072)
	WWT new binning (7669)	382	0.371 (SE = 0.003)
	SCANS aerial (123)	500	0.260 (includes availability and *g*(0)
	SCANS-II aerial (420)	300	0.623 (includes availability and *g*(0)
Minke whales	Non-ESAS boats (1025)	1000	0.370 (SE = 0.025)
	ESAS mode boats (152)	100	0.619 (SE = 0.056)
	University aerial (2)	100 (left truncated)	1
	WWT old binning (0)	-	-
	WWT new binning (39)	382	0.807 (SE = 0.136)
	SCANS aerial (1)	150	1
	SCANS-II aerial (14)	150	1
Dolphins	Non-ESAS boats (3970)	800	0.331 (SE=0.013)
	ESAS mode boats (512)	100	0.441 (SE = 0.019)
	University aerial (38)	300 (left truncated)	0.541 (SE = 0.078)
	WWT old binning (1)	382	-
	WWT new binning (215)	382	0.409(SE=0.021)
	SCANS aerial combined (60)	180	0.494 (SE=0.051)

In the case of harbour porpoise sightings in the WWT aerial surveys, in both the old and new binning regimes (see above) there were very few sightings further than 382 m (a bin margin) away so the sightings were truncated at this distance (Figure 4d and Figure 4e). In both cases the model with lowest AIC was half-normal with no additional covariates (old goodness of fit not possible, new goodness of fit: $\chi^2 < 0.01$, df = 1, $P = 0.987$). The number of sightings were 100 and 7669 respectively for each binning regime.

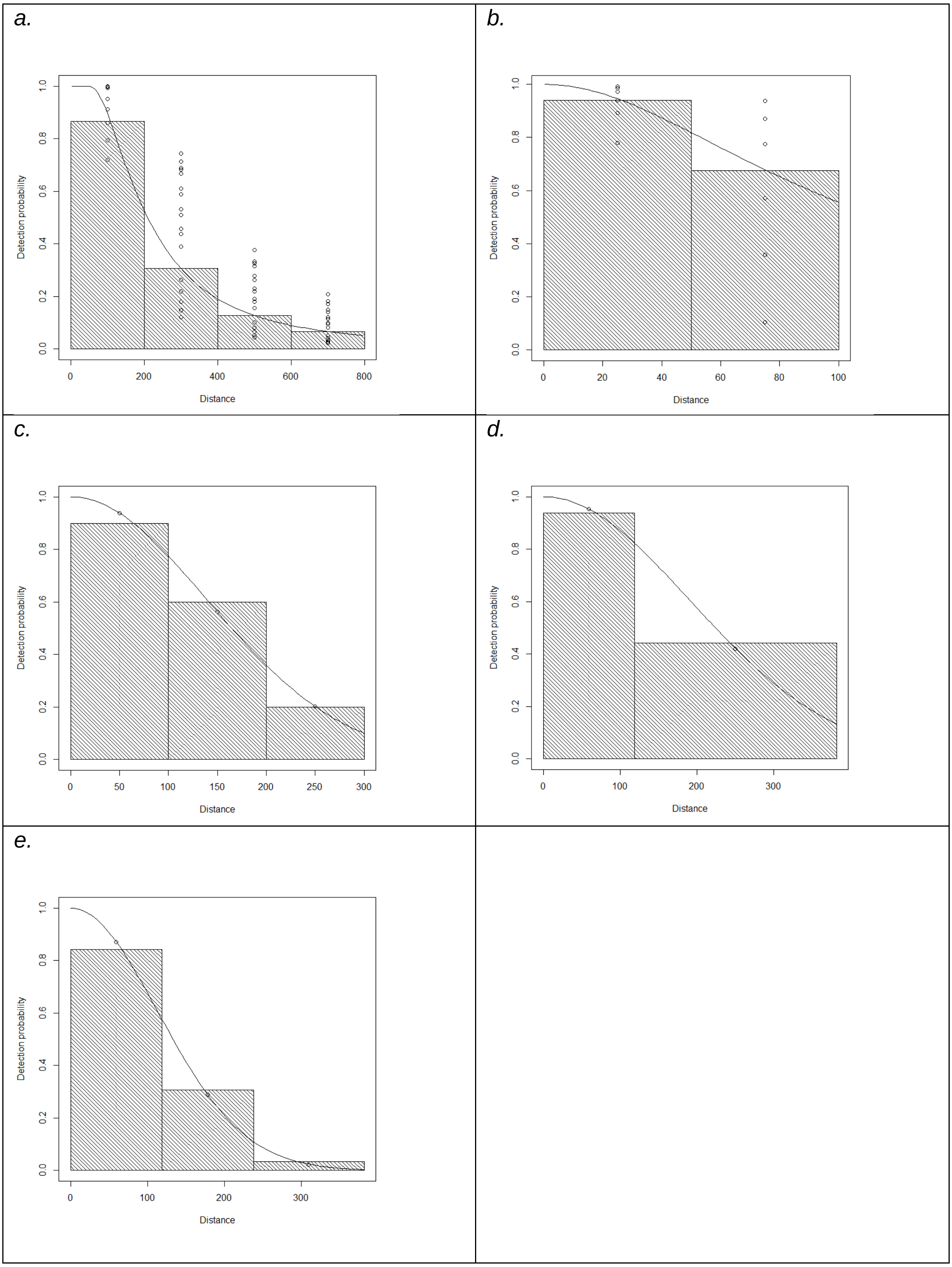

Figure 4. Detection functions for harbour porpoise for a. non-ESAS boat sightings, b. ESAS mode boat sightings, c. Swansea University and University of Aberdeen aerial surveys (after left and right truncation) d. WWT aeroplane sightings old binning (after left truncation) e. WWT aeroplane sightings new binning (after left truncation). The dots represent the fits of the different covariate values and the solid line is the mean fit.

3.2.3 Minke Whale

The single platform detection functions for minke whales are shown in Figure 5. Over a thousand (*n* = 1025) sightings were made by non-ESAS boats to generate the single platform half normal detection function (goodness of fit: $\chi^2 = 0.960$, df = 1, *P* = 0.327, Figure 5a) with a truncation distance of 1000 m. *Seastate* also contributed to the variation in detection probability. One hundred and fifty-two whale groups were seen by ESAS boats within the range 0 – 100 m (Figure 5b). No covariates in addition to distance were selected for the half-normal detection function derived from this dataset. A goodness of fit test was not possible. In the case of the university aerial surveys, only two minke whales were seen so no detection function could be fitted. The sightings were treated as a strip transect of half-width 100 m (i.e. detection probability = 1 within 100 m, 0 thereafter). In the case of the WWT aerial surveys, no minke whales were seen under the old binning regime. In the case of the newer WWT distance binning regime, there were 39 sightings, modelled using a half normal detection function (goodness of fit: $\chi^2 = 0.41$, df = 1, P = 0.524, Figure 5c). The two SCANS aerial surveys saw a total of 15 minke whales. The data were assumed to come from a strip with a half-width of 150 m.

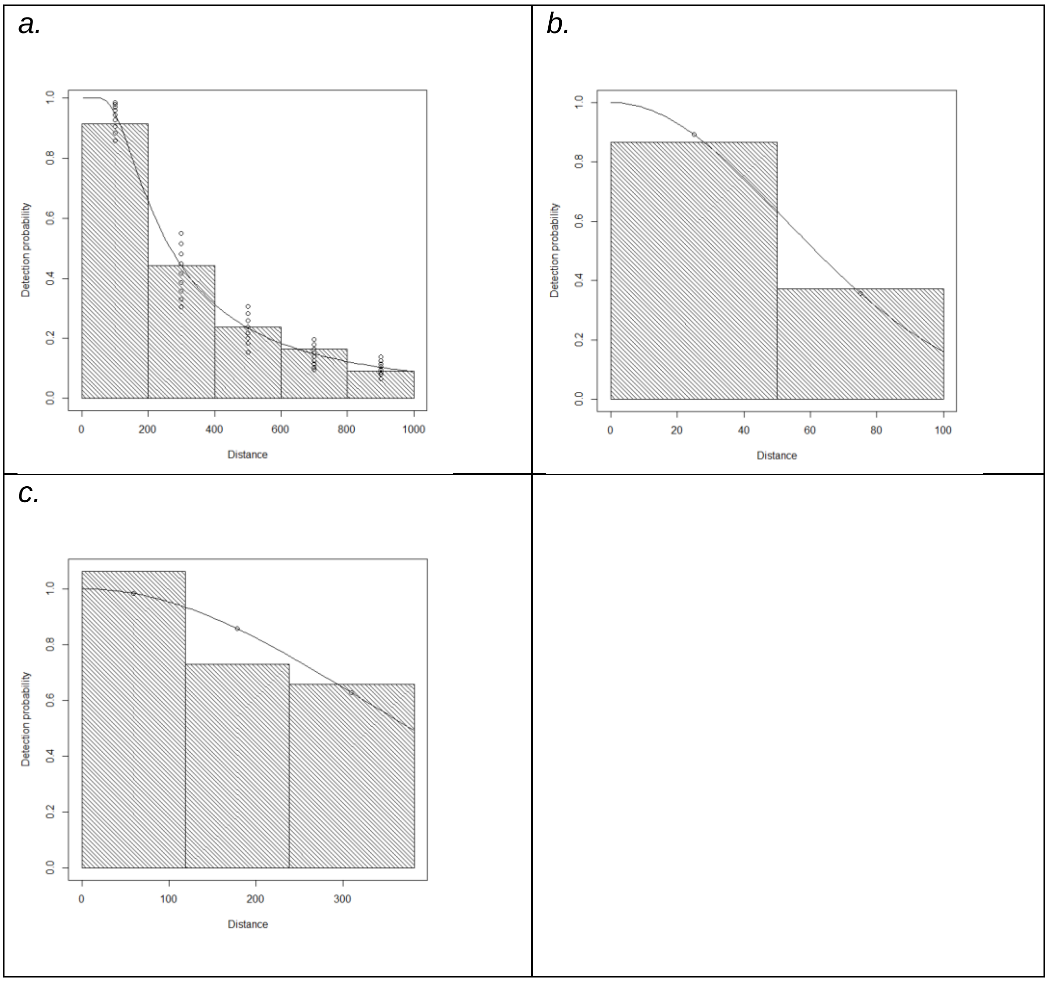

Figure 5. Detection functions for minke whales a. non-ESAS boat sightings, b. ESAS mode boat sightings, c. WWT aeroplane sightings with the new binning.

3.2.4 Dolphins

Figure 6 gives the detection functions for the single platform data. There were 3970 non-ESAS boat sightings over the range 0 – 800 m. The detection function with the lowest AIC was half normal with *VesselType, Seastate, Size* and *Species* as additional variables (no goodness of fit test possible, Figure 6a). In the case of the ESAS boat surveys, there were 512 sightings of dolphin groups within the truncation distance of 100 m. The best fit detection function was half-normal with the additional covariates of *Species, SeaState* and *VesselType* (no goodness of fit test possible, Figure 6b). The university aerial surveys saw 38 groups of dolphins. The best fit detection function was a half-normal distance only model (goodness of fit: $\chi^2 = 0.04$, df = 1, $P = 0.837$, Figure 6c). In the case of the WWT aerial surveys, there was just one identified Atlantic white-sided dolphin pod sighting under the original binning arrangements. The associated detection probability was taken from the WWT survey data under the new binning arrangements. In the case of the new binning arrangements (n = 215), the best fit model was a hazard rate with no covariates (goodness of fit: $\chi^2 = 1.45$, df = 1, $P = 0.228$, Figure 6d). In the case of the SCANS and SCANS-II aerial data, a half-normal detection function with no additional covariates was fitted over the range 0 – 300 m to 60 sightings (goodness of fit: $\chi^2 = 5.61$, df = 6, $P = 0.468$, Figure 6e).

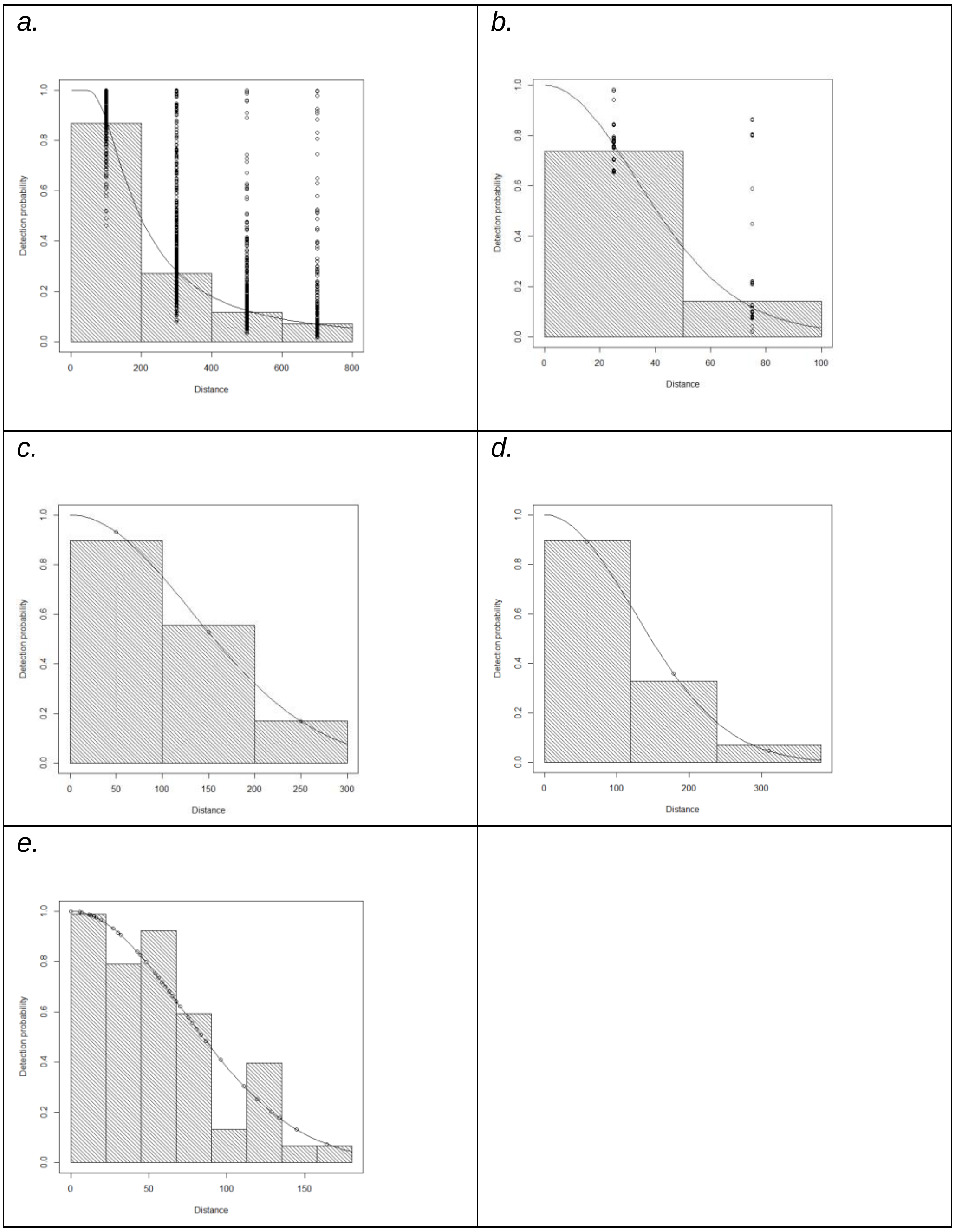

Figure 6. Detection functions for dolphins for a. non-ESAS boat sightings, b. ESAS mode boat sightings, c. Swansea University and University of Aberdeen aerial sightings d. WWT aeroplane sightings with the new binning and e) SCANS and SCANS-II aerial sightings.

3.3 Adjustment to Detectability

Table 7 summarises the adjustments made to every data set.

3.3.1 Detection on the Trackline (*g*(0))

Harbour porpoise

The double platform analysis of Cardigan Bay SWF boat data from 2008 contained 51 sightings of which 14 were duplicate sightings (seen by both platforms). The best fit mark-recapture component of a conditional detection function consisted of distance only (goodness of fit: $\chi^2 = 3.63$, df = 7, $P = 0.821$). The resultant best estimate of *g*(0) for *littleboats* was 0.431 (SE = 0.077), although for the Cardigan Bay data itself the estimate of *g*(0) was 0.676 (SE = 0.088) as the data were collected in independent-observer (IO) mode (Laake & Borchers 2004). The data were truncated at 800 m, as was the equivalent *bigboat* data (i.e. double platform data from SCANS/SCANS-II/CODA/Atlantic Array surveys). This data set had 1635 encounters with 931 groups seen by the tracker and 212 of these duplicate sightings seen by the primary. The mark-recapture component of the conditional detection function had *Size* and *SeaState* as well as distance as covariates (goodness of fit: $\chi^2 = 2.15$, df = 11, $P = 0.999$). The resultant *g*(0) for *bigboats* (0.306, SE=0.029) was used for *ferries* as well.

The aerial survey estimates from SCANS incorporated perception and availability bias for this species but no correction was available for the other aerial surveys.

Minke whales

No *g*(0) for *littleboats* was estimable from the 2008 double platform SWF survey, as no large cetaceans were seen. So *g*(0) for *littleboats* was assumed to be the same as the estimate for *bigboats* taken from the analysis of the double platform SCANS/SCANS-II/CODA/Atlantic Array surveys. In the latter case there were 268 encounters of which 148 were trials (i.e. seen by the trackers) of these 69 were seen by the primaries. The estimate of *g*(0) was 0.415 (SE = 0.012) from the mark-recapture component of the conditional detection function (goodness of fit: $\chi^2 = 2.85$, df = 5, $P = 0.723$).

No adjustments for aerial *g*(0) were made.

Dolphins

The estimate of *g*(0) for *littleboats* from the 2008 Sea Watch Cardigan Bay data was 0.670 (SE 0.193) based on only 11 trials in which four dolphin groups were seen by both observers (goodness of fit: $\chi^2 = 2.15$, df = 2, $P = 0.341$). This was conducted in IO mode so the above probability represents the combined probability of detection on the trackline for two independent observers. The *littleboat* estimate of *g*(0) was 0.470 (SE = 0.193). The bigboat *g*(0) estimate was 0.561 (SE = 0.035) from 198 trials with 111 duplicate sightings (goodness of fit: $\chi^2 = 3.67$, df = 10, $P = 0.961$).

No adjustments for aerial *g*(0) were made.

3.3.2 Availability at the Surface

Table 6. Mean surface and dive times of target species individuals.

Species	***Mean surface time (mins)***	***Mean dive time (mins)***
Harbour porpoise	0.065 (Otani 1998) 0.058 (Evans, P. pers. comm; also Evans 1971)	0.437 Otani (2000)
Minke whale	0.067 (Anderwald 2009) 0.044 (Gunnlaugsson 1989) 0.053 (Joyce *et al* 1989 off Svalbard)	1.311 (from Joyce et al 1989)
Bottlenose dolphin	0.058 (Evans, P. pers. comm)	1.867 (Lockyer & Morris 1986) 0.922 (Lockyer & Morris 1987) 0.43 (Mate et al 1995) 0.773 (Díaz López 2009) 1.043 (Díaz López *et al* 2008)
Common dolphin	0.058 (Evans, P. pers. comm)	1.0 (Evans, P. pers. comm)
Atlantic white-sided dolphin	0.058 (Evans, P. pers. comm)	

The sources for species availability at the surface are summarised in Table . Dolphin species not represented in Table 6 (Risso's dolphins and white beaked dolphins), were assumed to have the same characteristics as the other species, although dive times are likely to be longer for Risso's dolphin than the other dolphin species considered here (Wells *et al* 2009; PGH Evans *pers. comm.*). Final group availability was unique to a particular sighting, dependent on vessel speed and the presumed window of opportunity (see Section 2.3.2): harbour porpoise availability varied between 0.12 and 1 (mean: 0.49); minke whale between 0.04 and 1.00 (mean: 0.65); bottlenose dolphin between 0.06 and 1.0 (mean: 0.50); short-beaked common dolphin between 0.06 and 1.0 (mean: 0.84); Risso's dolphin between 0.06 and 1.0 (mean: 0.64); white-beaked dolphin between 0.06 and 1.0 (mean: 0.80) and white-sided dolphin between 0.11 and 1.0 (mean: 0.94). Table 8 shows the effect of the availability corrected on the mean $\widehat{N}$ for each survey.

Table 7. Perception (P) and availability (A) bias corrections made to detection probabilities by species (if species was present) and survey.

Data set	**Harbour porpoise**	**Minke whale**	**Bottlenose dolphin**	**Short-beaked common dolphin**	**Risso's dolphin**	**White-beaked dolphin**	**White-sided dolphin**
APEM	*A*	*A*	*A*	*A*	*A*	*A*	*A*
CBMWC	*P,A*	*P,A*	*P,A*	*P,A*	*P , A,*	*P,A*	*P,A*
Centrica	*P*	*P*	*P*	*P*	*P*	*P*	*P, A*
CODA	*P*	*P*	*P*	*P*	*P*	*P*	*P, A*
Cornish	*P*	*P*	*P*	*P*	*P*	*P*	*P, A*
RWEnPower	*P*	*P*	*P*	*P*	*P , A*	*P, A*	*P, A*
ESAS	*P, A, some P only*	*P*	*P, A, some P only*	*P, A, some P only*	*P , A, some P only*	*P, A, some P only*	*P, A some P only*
ESASaircraft	*A*	*A*	*A*	*A*	*A*	*A*	*A*
Forewind	*P*	*P*	*P*	*P*	*P*	*P*	*P, A*
HIDEF	*A*	*A*	*A*	*A*	*A*	*A*	*A*
HWDT	*P, A*	*P, A*	*P, A*	*P, A*	*P , A*	*P, A*	*P, A*
IFAW	*P*	*P*	*P*	*P*	*P*	*P*	*P, A*
IWDG	*P*	*P*	*P*	*P*	*P*	*P*	*P*
IWDG Ferry	*P*	*P*	*P*	*P*	*P*	*P*	*P*
MANW	*P, A*	*P, A*	*P, A*	*P, A*	*P, A*	*P, A*	*P,A*
MANX	*P, A*	*P, A*	*P, A*	*P, A*	*P, A*	*P, A*	*P,A*
Marinelife	*P, A*	*P, A*	*P, A*	*P, A*	*P, A*	*P, A*	*P, A*
Moray Offshore Renewables (MORL) boat	*P*	*P*	*P*	*P*	*P*	*P*	*P*
NORCET	*P*	*P*	*P*	*P*	*P*	*P*	*P*
North Wales Windfarm Environmental Assessment	*P*	*P*	*P*	*P*	*P*	*P*	*P*
Orca	*P*	*P*	*P*	*P*	*P*	*P*	*P*
Pembrokeshire Porpoise 2007	*P, A*	*P, A*	*P, A*	*P, A*	*P , A*	*P, A*	*P, A*
Pembrokeshire Porpoise 2008	*P, A*	*P, A*	*P, A*	*P, A*	*P , A*	*P, A*	*P, A*
MainstreamRP	*P, A*	*P, A*	*P, A*	*P, A*	*P , A*	*P, A*	*P, A*
RWE NPower (seabird)	*P, A*	*P, A*	*P, A*	*P, A*	*P , A*	*P, A*	*P, A*

Table 7. Cont. Perception (P) and availability (A) bias corrections made to detection probabilities by species (if assumed to be present) and survey.

Data set	**Harbour porpoise**	**Minke whale**	**Bottlenose dolphin**	**Short-beaked common dolphin**	**Risso's dolphin**	**White-beaked dolphin**	**White-sided dolphin**
SCANS II	*P*	*P*	*P*	*P*	*P*	*P*	*P, A*
SCANS II (air)	*P, A*	*A*	*A*	*A*	*A*	*A*	*A*
SCANS	*P*	*P*	*P*	*P*	*P*	*P*	*P, A*
SCANS (air)	*P, A*	*A*	*A*	*A*	*A*	*A*	*A*
Scottish Power Argyll boat	*P*	*P*	*P*	*P*	*P*	*P*	*P, A*
Sea Energy Renewables Inch Cape boat	*P*	*P*	*P*	*P*	*P*	*P*	*P, A*
Scottish Power & EAOW	*P*	*P*	*P*	*P*	*P*	*P*	*P, A*
Smartwind Hornsea	*P*	*P*	*P*	*P*	*P*	*P*	*P, A*
SWF	*P, A*	*P, A*	*P, A*	*P, A*	*P , A*	*A*	*P,A*
University of Aberdeen aerial	*A*	*A*	*A*	*A*	*A*	*A*	*A*
University of Aberdeen boat	*P, A*	*P, A*	*P, A*	*P, A*	*P , A*	*P, A*	*P, A*
University of Aberdeen boat SAC	*P, A*	*P, A*	*P, A*	*P, A*	*P , A*	*P, A*	*P, A*
University of Aberdeen/MORL/HiDef	*A*	*A*	*A*	*A*	*A*	*A*	*A*
University of Aberdeen/MORL/HiDef/Crown Estate	*A*	*A*	*A*	*A*	*A*	*A*	*A*
University of Aberdeen ferry	*P*	*P*	*P*	*P*	*P*	*P*	*P*
University of Swansea	*A*	*A*	*A*	*A*	*A*	*A*	*A*
WDCS	*P*	*P*	*P*	*P*	*P*	*P*	*P, A*
WWT	*A*	*A*	*A*	*A*	*A*	*A*	*A*

Table 8. Mean $\widehat{\mathbf{D}}$ per segment with and without an availability correction. Red indicates a change in mean value

Survey	Harbour porpoise		Minke whale		Bottlenose dolphin		Short-beaked common dolphin		Risso's dolphin		White-beaked dolphin		White-sided dolphin	
	Without correction	With correction	Without correction	With correction	Without correction	With correction	Without correction	With correction	Without correction	With correction	Without correction	With correction	Without correction	With correction
APEM	0.0545	0.4319	0	0	0	0	0	0	0	0	0	0	0	0
CBMWC	1.3029	1.8022	0.0008	0.0056	1.3641	2.5428	0	0	0	0	0	0	0	0
Centrica	0.5386	0.5386	0.0024	0.0024	0.0027	0.0027	0.0038	0.0038	0	0	0	0	0	0
CODA	0.9385	0.9385	0.1258	0.1258	0.2255	0.2255	0	0	0	0	0	0	0	0
Cornish	0.4102	0.4102	0	0	0.0377	0.0377	0.1629	0.1629	0	0	0	0	0	0
RWEnPower	0.1460	0.1460	0.0031	0.0031	0	0	3.9246	3.9246	0	0	0	0	0	0
ESAS	0.0578	0.0582	0.0111	0.0111	0.0032	0.0037	0.1187	0.1187	0.0012	0.0012	0.0491	0.0491	0.0146	0.0146
ESASaircraft	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Forewind	0.1863	0.1863	0.0118	0.0118	0	0	0	0	0	0	0.0115	0.0115	0	0
HIDEF	0.0289	0.2333	0	0	0	0	0	0	0	0	0.0015	0.0279	0	0
HWDT	1.8853	3.1587	0.0232	0.1149	0.0071	0.0093	0.0955	0.1202	0.0053	0.0083	0.0134	0.0202	0.0051	0.0054
IFAW	NA	NA	0	0	0	0	0	0	0	0	0	0	0	0
IWDG	0.0953	0.0953	0.0050	0.0050	0.0020	0.0020	0.3535	0.3535	0.0024	0.0024	0	0	0	0
IWDG Ferry	0.4830	0.4830	0.0035	0.0035	0.0008	0.0008	0.1990	0.1990	0.0016	0.0016	0	0	0.0003	0.0003
MANW	5.9866	10.4736	0	0	0	0	0	0	0	0	0	0	0	0
MANX	1.7673	3.6226	0.0770	0.4113	0	0	0	0	0.1451	0.3905	0	0	0	0
Marinelife	0.2675	0.296	0.0024	0.0032	0.0151	0.0167	0.0864	0.0882	0.001	0.001	0.0028	0.0035	0.0001	0.001
Moray Offshore Renewables boat	2.4145	2.4145	0.0204	0.0204	0.0012	0.0012	0.0125	0.0125	0.0004	0.0004	0.0024	0.0024	0	0
NORCET	0.2765	0.2765	0.0077	0.0077	0.0468	0.0468	0.0085	0.0085	0.0010	0.0010	0.0205	0.0205	0.0101	0.0101

Survey	Harbour porpoise		Minke whale		Bottlenose dolphin		Short-beaked common dolphin		Risso's dolphin		White-beaked dolphin		White-sided dolphin	
	Without correction	With correction	Without correction	With correction	Without correction	With correction	Without correction	With correction	Without correction	With correction	Without correction	With correction	Without correction	With correction
Orca	0.2298	0.2298	0.0041	0.0041	0.0094	0.0094	0.0355	0.0355	0.0008	0.0008	0.0036	0.0036	0.0002	0.0002
Pembrokeshire Porpoise 2007	1.4476	1.9702	0.0023	0.0212	0	0	0.1209	0.1411	0	0	0	0	0	0
Pembrokeshire Porpoise 2008	2.8370	4.2128	0.0013	0.0144	0.0096	0.0123	0.1344	0.1396	0	0	0	0	0	0
MainstreamRP	0.0850	0.0850	0	0	0	0	0	0	0	0	0	0	0	0
University of Aberdeen SAC	0.5770	1.2023	0.0206	0.1159	0.1743	0.5381	0	0	0	0	0	0	0	0
RWE NPower (seabird)	0.3116	0.3116	0.0037	0.0037	0	0	0.4336	0.4336	0	0	0	0	0	0
SCANS II	1.7328	1.7328	0.0255	0.0255	0.0174	0.0174	0.0525	0.0525	0.0016	0.0016	0.0297	0.0297	0.0557	0.0557
SCANS II (air)	-	0.1522	0.0037	0.0715	0.0033	0.0488	0.0717	0.2342	0.0051	0.0235	0	0	0	0
SCANS	1.5756	1.5756	0.0318	0.0318	0.0038	0.0038	0.0160	0.0160	0.0044	0.0044	0.0459	0.0459	0.0024	0.0024
SCANS (air)	-	0.2675	0.0015	0.0356	0	0	0	0	0	0	0	0	0	0
Scottish Power Argyll boat	0.5634	0.5634	0.0263	0.0263	0	0	0.3853	0.3853	0	0	0.0008	0.0008	0	0
Sea energy renewables Inch Cape boat	0.1760	0.1760	0	0	0	0	0	0	0	0	0	0	0	0

Survey	Harbour porpoise		Minke whale		Bottlenose dolphin		Short-beaked common dolphin		Risso’s dolphin		White-beaked dolphin		White-sided dolphin	
	Without correction	With correction	Without correction	With correction	Without correction	With correction	Without correction	With correction	Without correction	With correction	Without correction	With correction	Without correction	With correction
Scottish Power & EAOW	0.5148	0.5148	0	0	0.0051	0.0051	0	0	0.0013	0.0013	0.0079	0.0079	0	0
Smartwind Hornsea	0.5633	0.5633	0.0017	0.0017	0	0	0	0	0	0	0.0048	0.0048	0	0
SWF	0.9102	1.512	0.0135	0.0611	0.2564	0.4492	0.2331	0.3099	0.0099	0.0145	0.0130	0.0188	0.0170	0.0181
SWF double platform	1.2858	2.4008	0	0	0.2934	0.6112	0	0	0	0	0	0	0	0
University of Aberdeen aerial	0.1760	0.8361	0.0001	0.0026	0.0092	0.0581	0.0044	0.0194	0	0	0.0003	0.0040	0	0
University of Aberdeen boat	1.1755	1.1755	0.0633	0.0633	0	0	0	0	0	0	0	0	0	0
University of Aberdeen/MORL/HiDef	0.2425	1.4321	0	0	0	0	0.0024	0.0432	0	0	0	0	0	0
University of Aberdeen/MORL/HiDef/Crown Estate	0.1613	1.2344	0.0128	0.3196	0	0	0	0	0	0	0	0	0	0
University of Aberdeen ferry	1.0044	1.0044	0.0181	0.0181	0.0051	0.0051	0.0553	0.0553	0.0017	0.0017	0.0020	0.0020	0	0
University of Swansea	0.0056	0.0345	0	0	0.0082	0.0411	0.0280	0.1441	0	0	0	0	0	0
WDCS	0.1261	0.1261	0	0	0	0	1.4703	1.4703	0	0	0	0	0	0
WWT	0.1322	0.8370	0.0002	0.0044	0.0007	0.0062	0.0047	0.0171	0.0002	0.0011	0.0020	0.0161	<0.0001	0.0002

3.4 Realized Effort

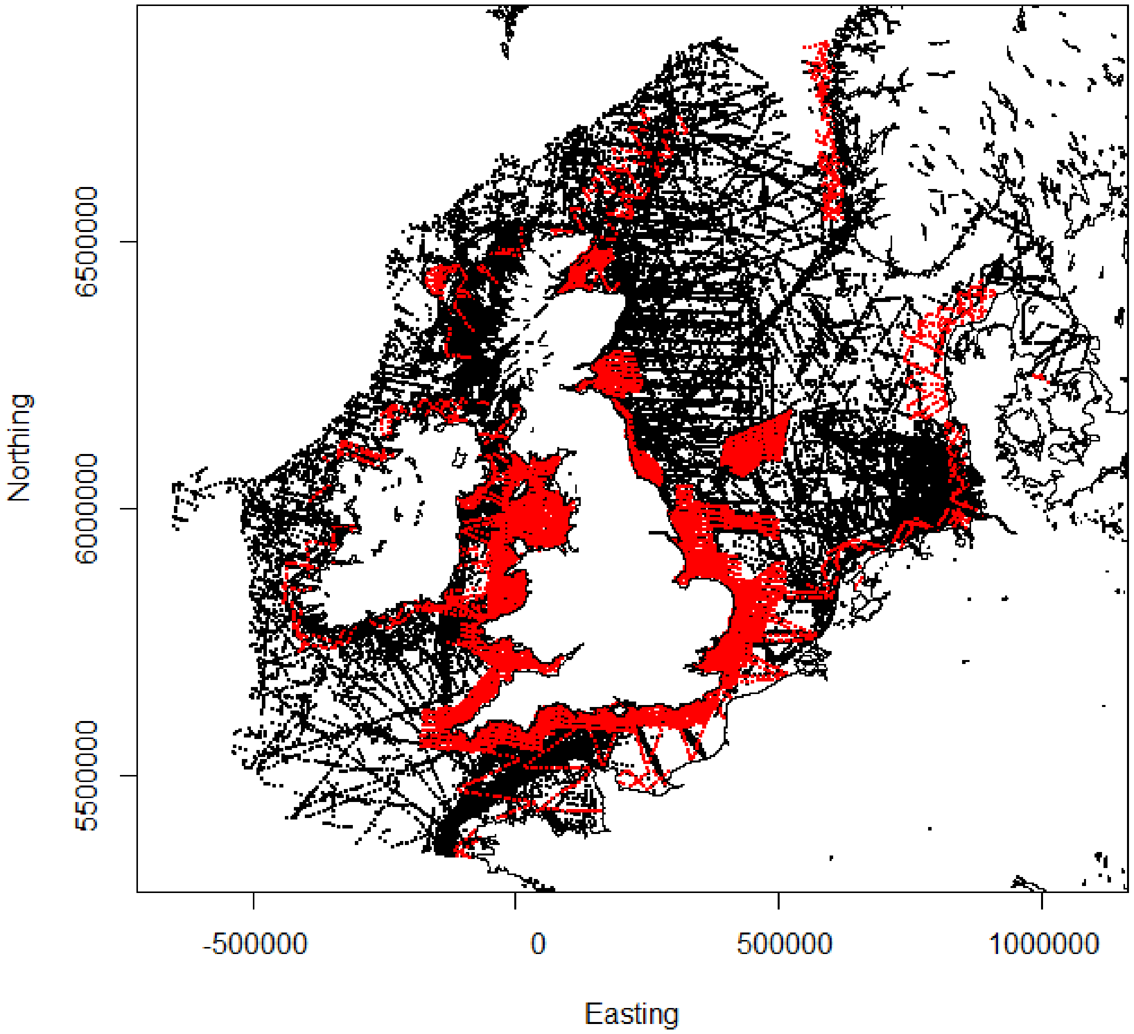

Figure 7. Realized effort in JCP Phase III by platform type. Each point represents the midpoint of a segment of effort from the period 1994 – 2010: black is boat effort and red is aeroplane effort.

Realized survey effort by vessel type is given in Figure 7, and Figures 8 and 9 give the quantity of realized survey effort per 5 × 5 km cell over the region of interest by year and by season respectively. There is confounding of location (and hence depth) and survey vessel type, with ferries traversing the same region repeatedly and aerial effort primarily in coastal regions. There is greater effort in more recent years and, unsurprisingly, during the summer. There is relatively little effort in the western and north-western approaches despite the offshore CODA survey in 2007. In more recent years survey effort has been concentrated in regions of potential offshore development interest (e.g. Figure 8q, r).

The annual and seasonal breakdown of effort frequency since 1994 is given in Figure 10.

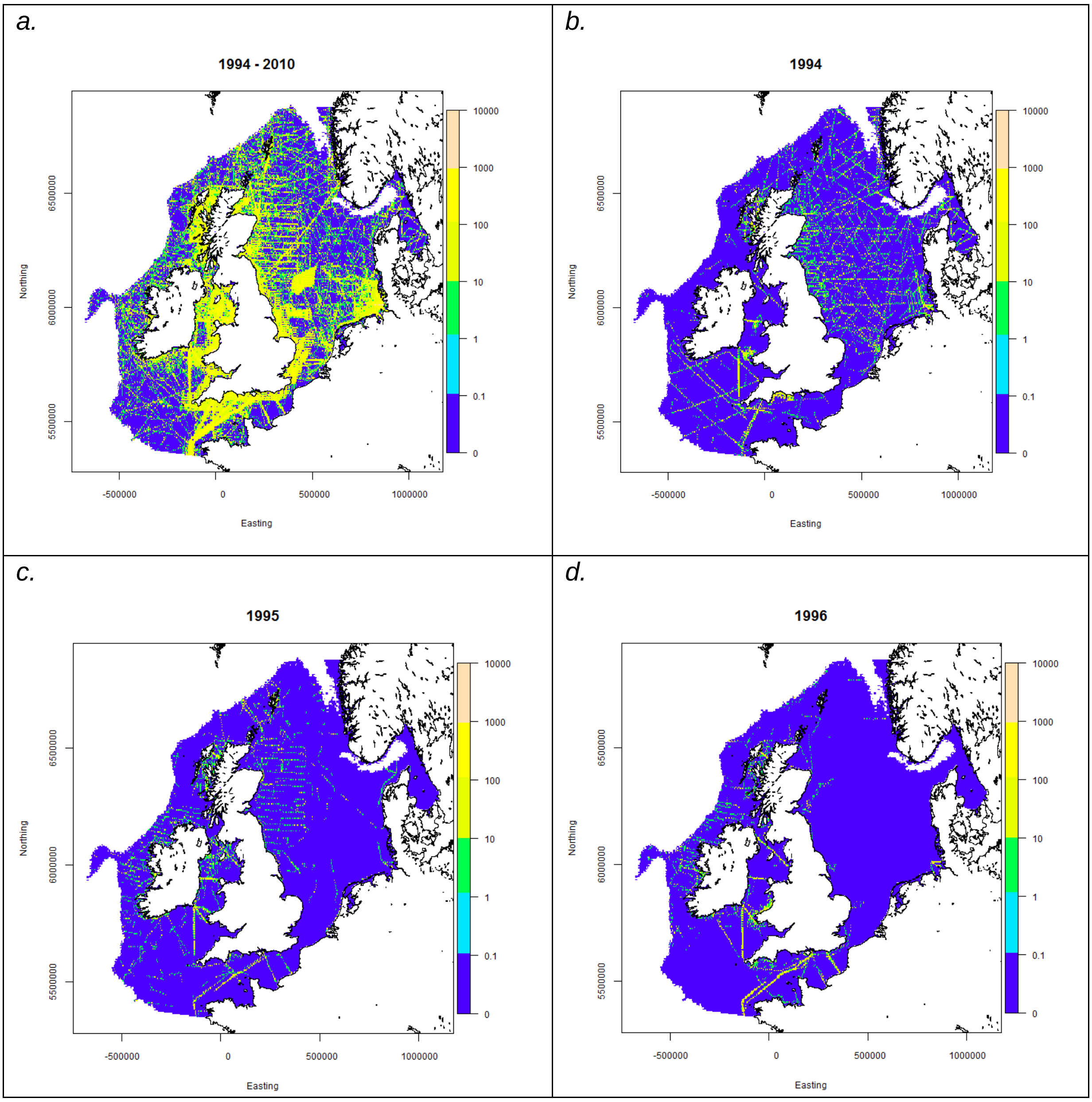

Figure 8. Realized effort over the Phase III area: a. all years 1994 - 2010, b. 1994, c. 1995 and d. 1996. Colours represent effort as distance travelled (km) per grid cell over the appropriate period in conditions of sea state 4 or less (with the addition of some aerial effort at sea state 6 or less). Each cell is 5 by 5 km.

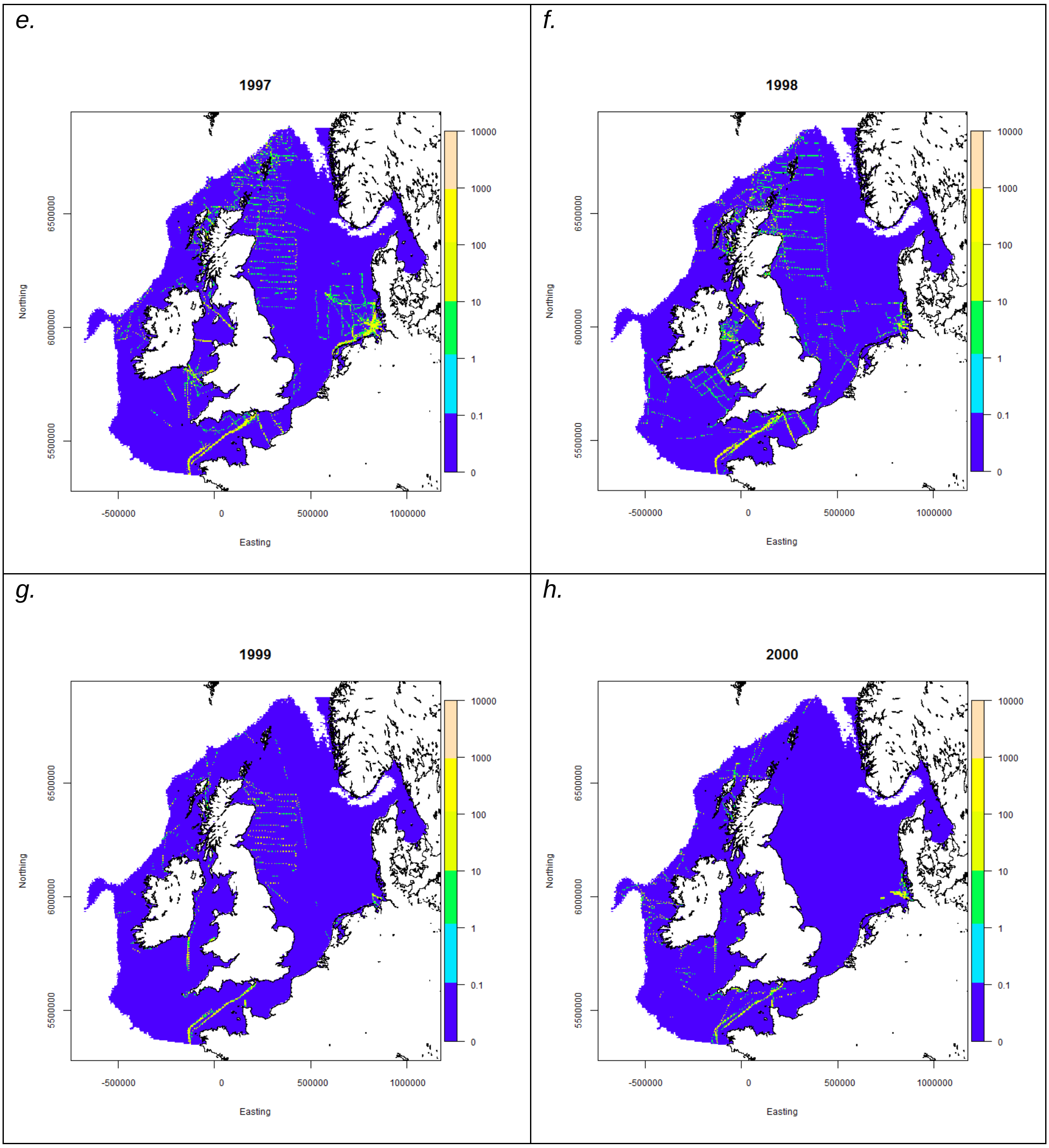

Figure 8 Continued. Realized effort over the Phase III area: e. 1997, f. 1998, g. 1999 and h. 2000. Colours represent effort as distance travelled (km) per grid cell over the appropriate period in conditions of sea state 4 or less (with the addition of some aerial effort at sea state 6 or less). Each cell is 5 by 5 km.

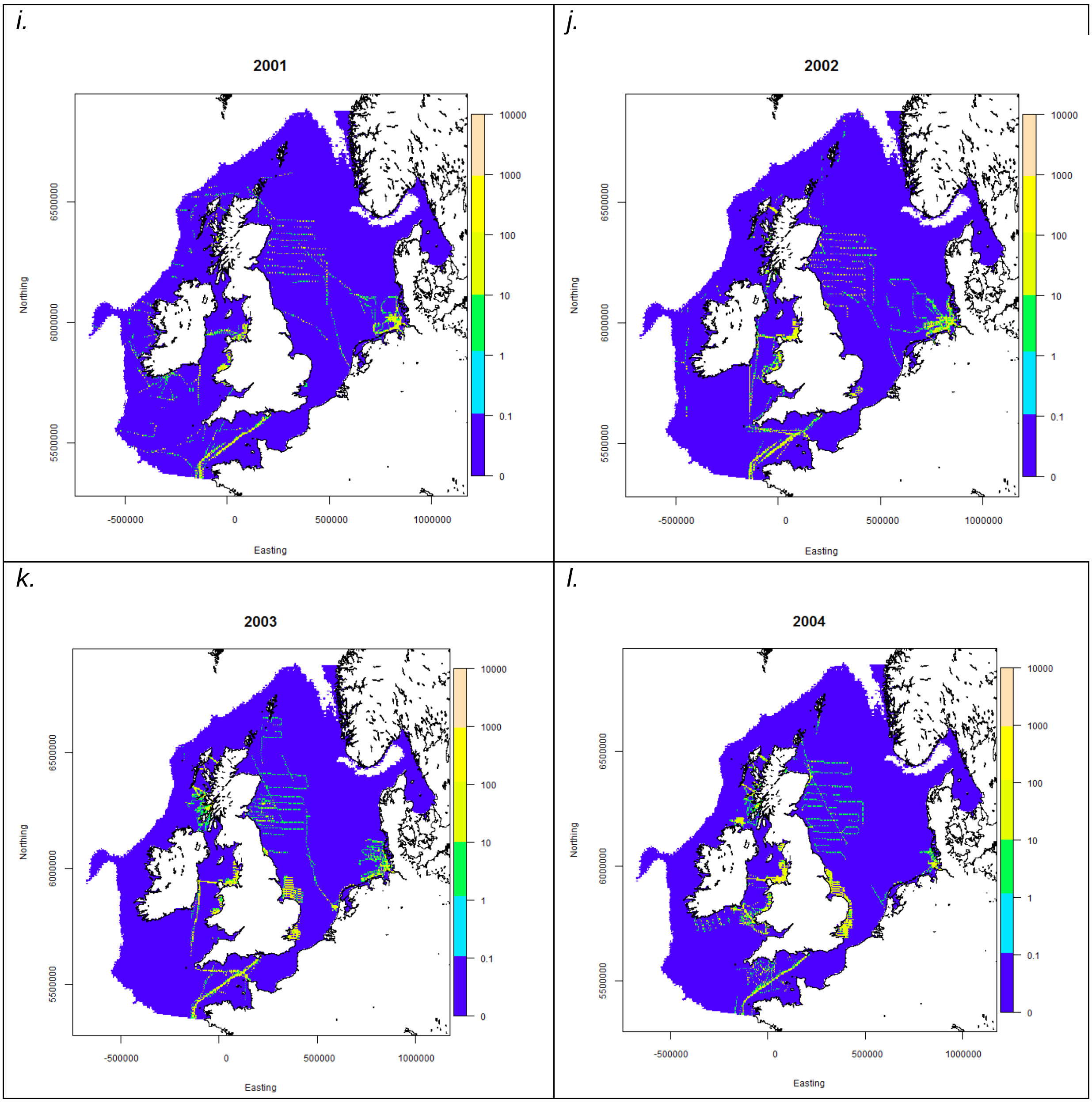

Figure 8 Continued. Realized effort over the Phase III area: i. 2001, j. 2002, k. 2003, and l. 2004. Colours represent effort as distance travelled (km) per grid cell over the appropriate period in conditions of sea state 4 or less (with the addition of some aerial effort at sea state 6 or less). Each cell is 5 by 5 km.

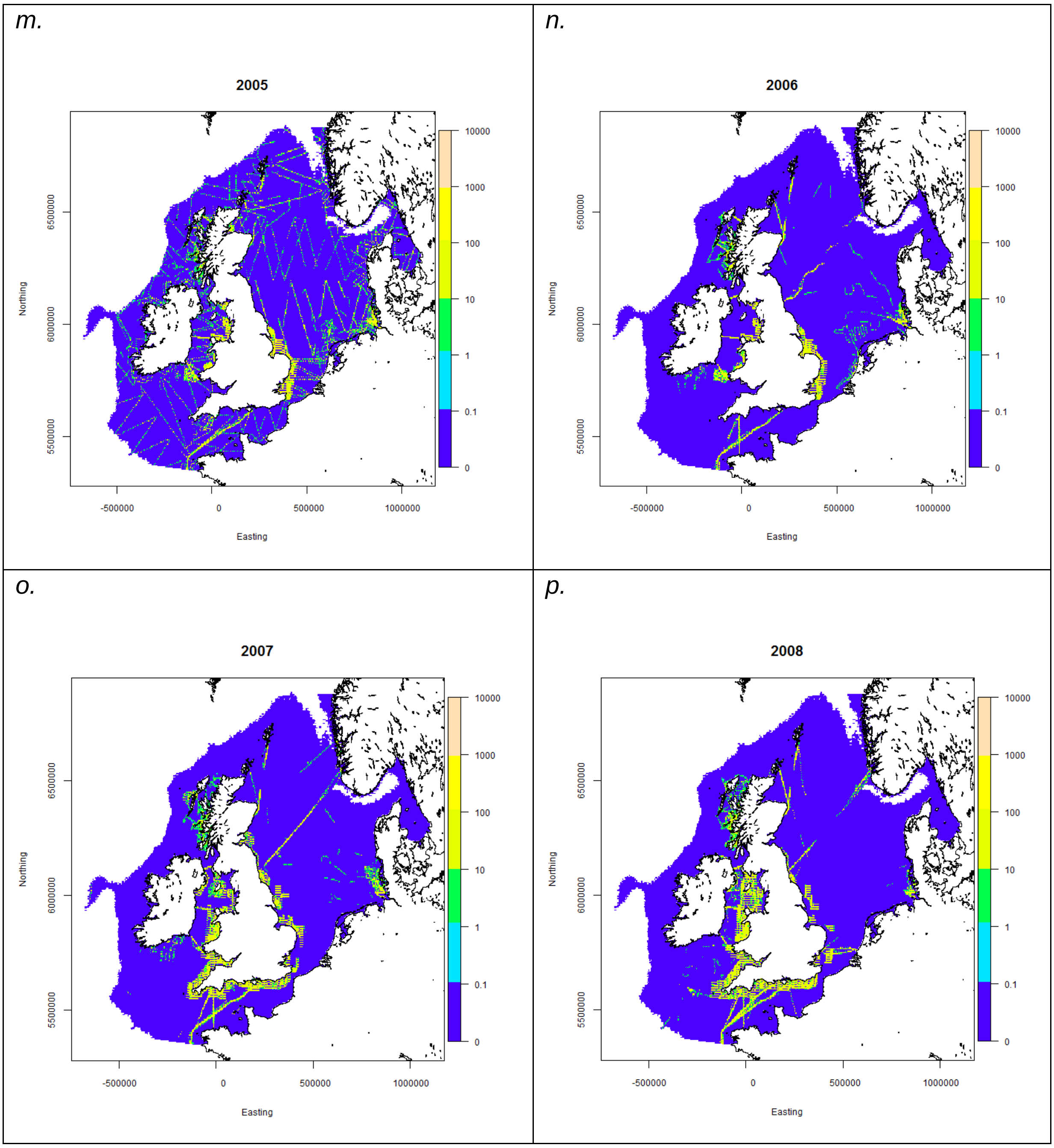

Figure 8 Continued. Realized effort over the Phase III area: m. 2005, n. 2006, o. 2007, and p. 2008. Colours represent effort as distance travelled (km) per grid cell over the appropriate period in conditions of sea state 4 or less (with the addition of some aerial effort at sea state 6 or less). Each cell is 5 by 5 km.

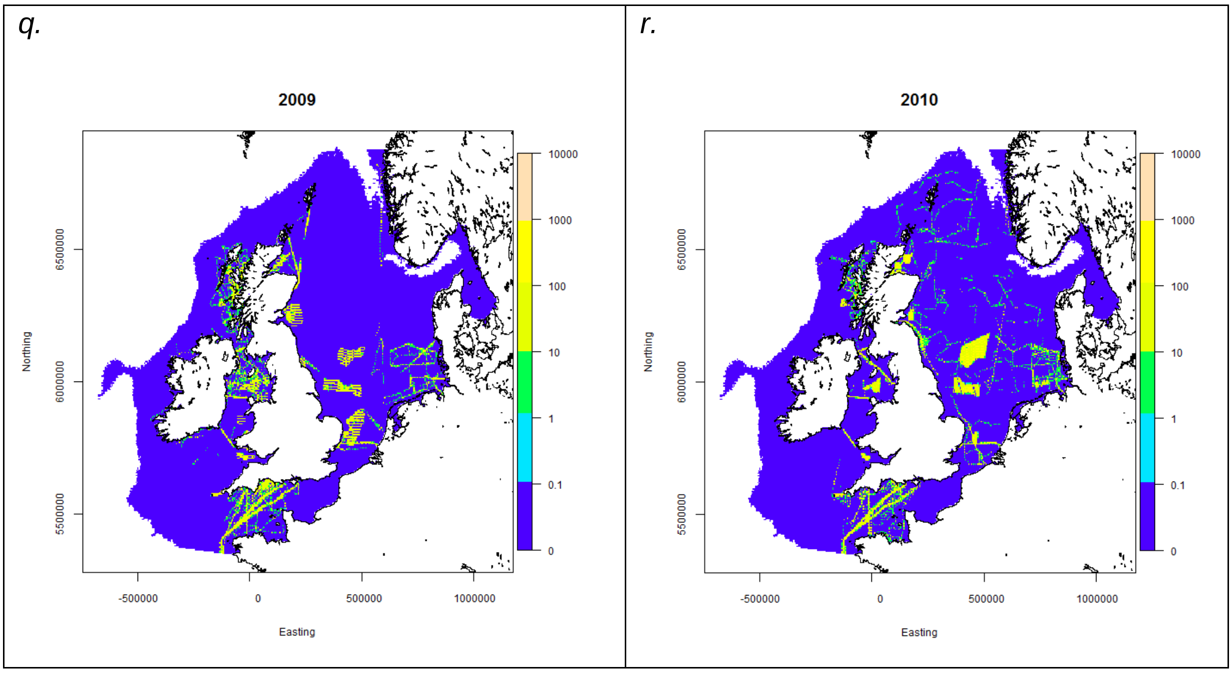

Figure 8. Continued. Realized effort over the Phase III area: a. 2009, b. 2010. Colours represent effort as distance travelled (km) per grid cell over the appropriate period in conditions of sea state 4 or less (with the addition of some aerial effort at sea state 6 or less). Each cell is 5 by 5 km.

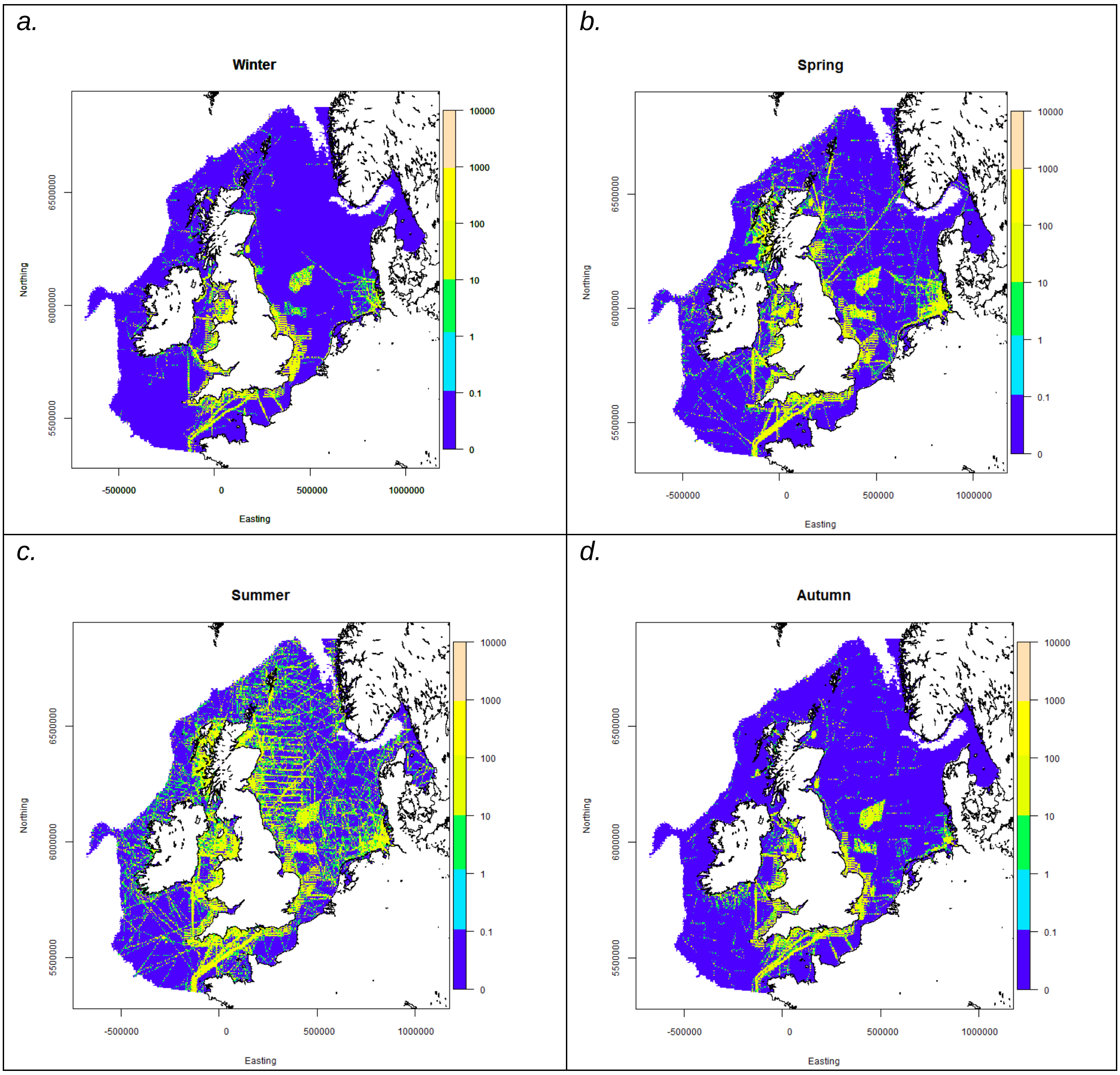

Figure 9. Realized effort over the Phase III area by Season. a. Winter (Jan. – Mar.), b. Spring (Apr. – Jun.), c. Summer (Jul. – Sep.) and d. Autumn (Oct. – Dec.). Colours represent effort as distance travelled (km) per grid cell over the relevant season 1994 – 2010 in conditions of sea state 4 or less (with the addition of some aerial effort at sea state 6 or less). Each cell is 5 by 5 km.

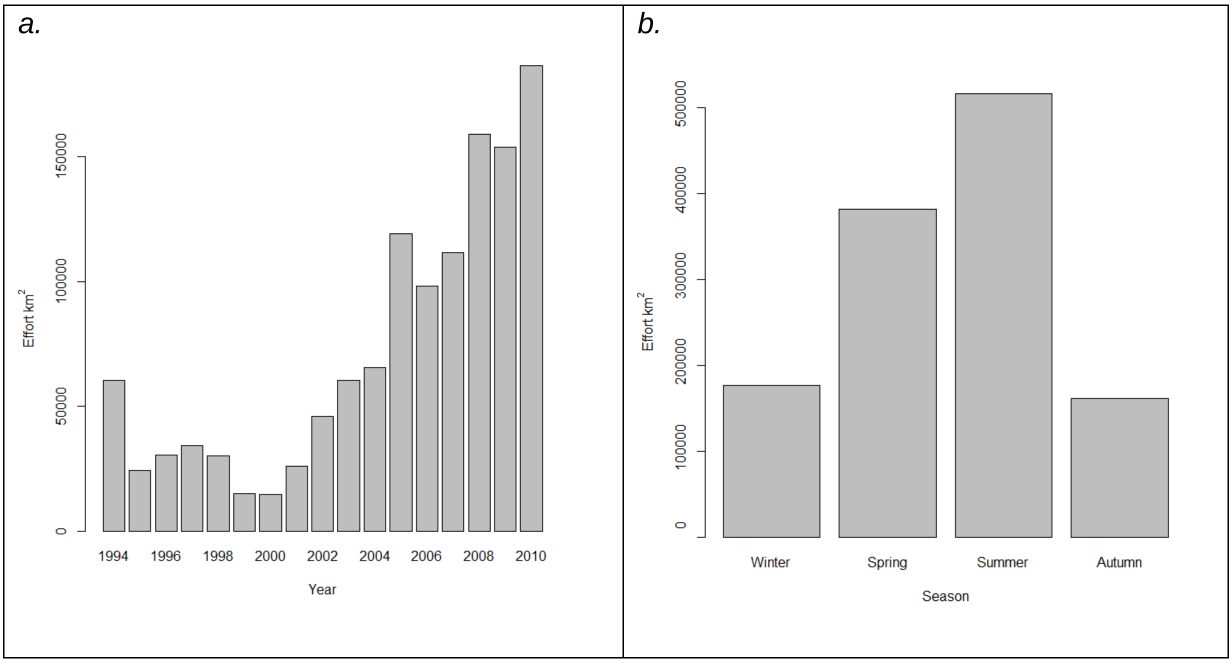

Figure 10. Aspects of the JCP Phase III effort. a. Realized effort by year across all surveys, b. Realized effort by season across all surveys (length km traversed).

3.5 Density Surface Modelling

3.5.1 Model Selection

Summaries of the GEE models after model selection are given in Table 9. Only in the case of harbour porpoise was a model chosen with an interaction of space and year. For particularly data sparse species (RD, WBD and WSD) the spatial smooth was restricted to a maximum of 6 degrees of freedom in order to prevent overly complex models from being selected, and so prevent the correspondingly high uncertainty in model predictions that would otherwise arise.

Table 9. Density surface models by species. Species codes: HP = harbour porpoise, MW = minke whale, BND = bottlenose dolphin, CD = short-beaked common dolphin, RD= Risso's dolphin, WBD = white-beaked dolphin, WSD= Atlantic white-sided dolphin. bs(x,5) indicates a b-spline with 5 degrees of freedom. cc indicates a cubic cyclic spline. s indicates a radial basis using the CReSS method.

Species	***%Zero segments***	***Model***
HP	*88.6*	*bs(Year,5) + bs(Depth,5) + cc(Dayofyear, 5) + bs(Slope,5) + bs(SST,5) + s(Easting, Northing,12) + s(Easting, Northing, Year, 60)*
MW	*99.1*	*bs(Year,3)+ cc(Dayofyear, 5)+ bs(SST,5) + bs(Slope, 5) + bs(Depth, 5)+ s(Easting, Northing, 11)*
BND	*99.1*	*bs(Year(5) + bs(Slope, 5) + bs(Depth, 5) + bs(SST, 5) + cc(Dayofyear, 5) + s(Easting, Northing,12)*
CD	*98.8*	*bs(Year, 5) + bs(Depth,5) + bs(SST,5) + bs(Slope, 5) + cc(Dayofyear, 5) + s(Easting, Northing,12)*
RD	*99.9*	*bs(Year, 5) + cc(Dayofyear,5)+ s(Easting, Northing, 6)*
WBD	*99.6*	*bs(Year, 4) + bs(Depth,5) + cc(Dayofyear,5) + s(Easting, Northing, 6)*
WSD*	*99.9*	*bs(Year,5) + bs(Depth,5) + cc(Dayofyear, 5) + s(Easting, Northing, 6)*

**Highest estimated* $\hat{D}_c$ *reduced to observed* D_c *to facilitate fitting*

3.5.2 Harbour Porpoise

Analysis of the 88734 segments for harbour porpoise revealed a moderate proportion of non-zero segments (*n* = 10098). Mean $\hat{D}_c$ was 0.8738 (SE: 0.022) (ranging from 0.174 in 1998 to 1.444 in 2009) animals/km^2 across the entire region of interest. The final chosen model consisted of single smooths of all the available predictors plus an interaction of the 2D spatial smooth with *Year* (Table 9).

Harbour porpoise densities fluctuated throughout the year (Table 10, Figures 11, A4.4-A4.6). A density surface map for harbour porpoise for the summer of 2010 is given in Table 10, Figures 11, A4.4-A4.6 with $\hat{D}_c$ from 2008 – 2010 to indicate the data being drawn upon with this model with an interaction. $\hat{D}_c$ are supplied as mean values over a 53 by 60 grid with each grid cell representing 900 km^2 in contrast to the prediction cells which are 25 km^2. Some consistent patterns in harbour porpoise density were found (Appendix Figures A4.1 – A4.6). Harbour porpoise periodically occurred in coastal waters all around the British Isles with the exception of the English Channel where they are never recorded,

except on the border with the Celtic Sea. Varying areas of higher density for harbour porpoise were seen in the North Sea, the Celtic Sea, the Hebrides and the west coast of Ireland (1994 – 2000, see Figure A4.1) with predicted densities sometimes greater than three animals/km^2.

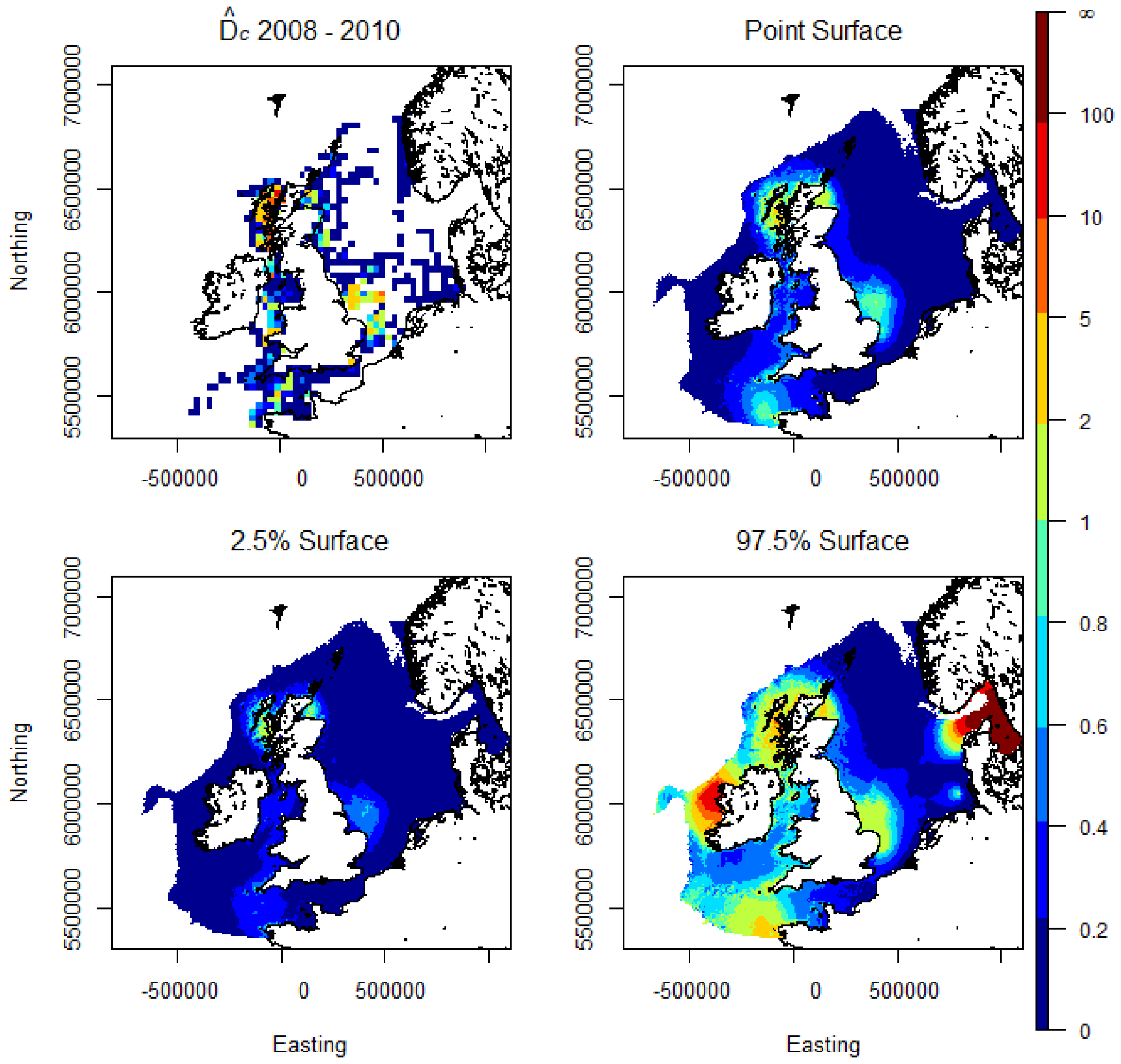

Figure 11. Predicted harbour porpoise densities (animals/km^2) for summer 2010. Top left: input densities $\hat{D}_c$ summers 2008 – 2010 to show data being drawn upon for this prediction with a temporal interaction. Top right: point estimate of cell densities. Bottom left: lower (2.5%) confidence limit on cell densities. Bottom right: upper (97.5%) confidence limit on cell densities (animals/km^2). Note that the top left plot exaggerates the spatial coverage of the relevant effort.

Table 10. Estimated abundances of harbour porpoise in 2010. Please refer to Appendices 6 and 7 for an explanation of how these figures should be scaled to use in the context of agreed Management Unit reference populations.

	Winter			***Spring***			***Summer***			***Autumn***		
	Point	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***
Shallow OSPAR2	154700	129300	7.96E+24*	82300	50500	1.78E+30*	101900	87000	2.40E+27*	60600	49000	3.39E+23*
OSPAR3	127900	97700	473600	111400	66100	871400	127600	90400	503400	84600	58300	355600
Truncated mATL	281900	237200	2.13E+18*	192900	122000	3.58E+10*	229000	188500	1.06E+10*	144700	108600	5.2E+09*
Truncated EEZ	234500	195700	318900	147000	88400	292300	180100	141800	250700	110200	85100	166500
Moray Firth	13500	7400	27100	8100	5200	16200	9000	5800	13500	5300	3200	9500
Firth of Forth	7000	5200	11800	3500	1900	6600	4400	2900	6800	2500	1600	3600
Atlantic Array	8500	5100	12800	7100	4300	12700	8600	5600	13300	6000	4100	9900
Islay	1300	300	4100	800	200	3200	1100	400	3300	600	200	1800
Solway Firth	1200	600	2300	600	300	1200	800	400	1400	500	300	900
North	4900	2700	10600	3200	1500	7300	3500	1700	6100	2100	900	4800
East Orkney	1900	1200	3400	1200	600	2600	1300	800	2000	800	400	1500
Lewis	1400	700	3000	1100	500	2600	1400	500	2600	800	400	1900
Kyle Rhea	400	200	900	300	100	800	300	200	700	200	100	400
Sound of Islay	500	200	1100	300	100	600	400	200	800	200	100	400
Argyll Array	3100	1400	6600	2200	900	4600	2900	1700	6800	1800	1000	3700
Hastings	300	100	700	200	100	400	200	100	600	200	100	400
IOW	900	400	1800	600	200	1700	800	400	1600	600	200	1200
Dogger Bank	5700	3800	9700	2200	1300	4400	3100	2200	5600	1600	1100	2300
South Dogger Bank	18400	12500	26300	7000	4000	13600	9700	6700	13100	5000	3700	7400
Norfolk Bank	13700	7000	26200	5300	2600	15600	7100	3600	12700	4000	1800	8500
Irish Sea	4600	2500	8100	2300	1400	4100	3200	2100	5400	2000	1000	3500
Strangford Lough	500	300	1000	200	100	500	300	200	600	200	100	400
Shetlands	100	0	400	100	0	400	100	0	200	0	0	200

**Caused by great uncertainty in the estimates for the Skagerrak and Kattegat.*

3.5.3 Minke Whale

Analysis of the 131448 segments for minke whale revealed a large number of zero segments with only 1152 non-zero segments. Mean $\hat{D}_c$ was 0.022 (SE = 0.001) (ranging from 0.008 in 2010 to 0.046 in 2001) animals/km^2 across the entire region of interest. The chosen model consisted of single smooths of all the available predictors and a 2D spatial smooth (Table 9).

Minke whale abundances are given in Table 11. A density surface map for minke whale in the summer of 2010 is given in Figure 12. Some consistent patterns in minke whale distributions were found. Density was predicted as highest around the northern coast of Great Britain, from Yorkshire to the Kintyre peninsula, with changing high density regions in the north-western North Sea (Appendix figures A4.7 – A4.12 $\hat{D}_c$ plots). Minke whales are occasionally predicted in the western English Channel.

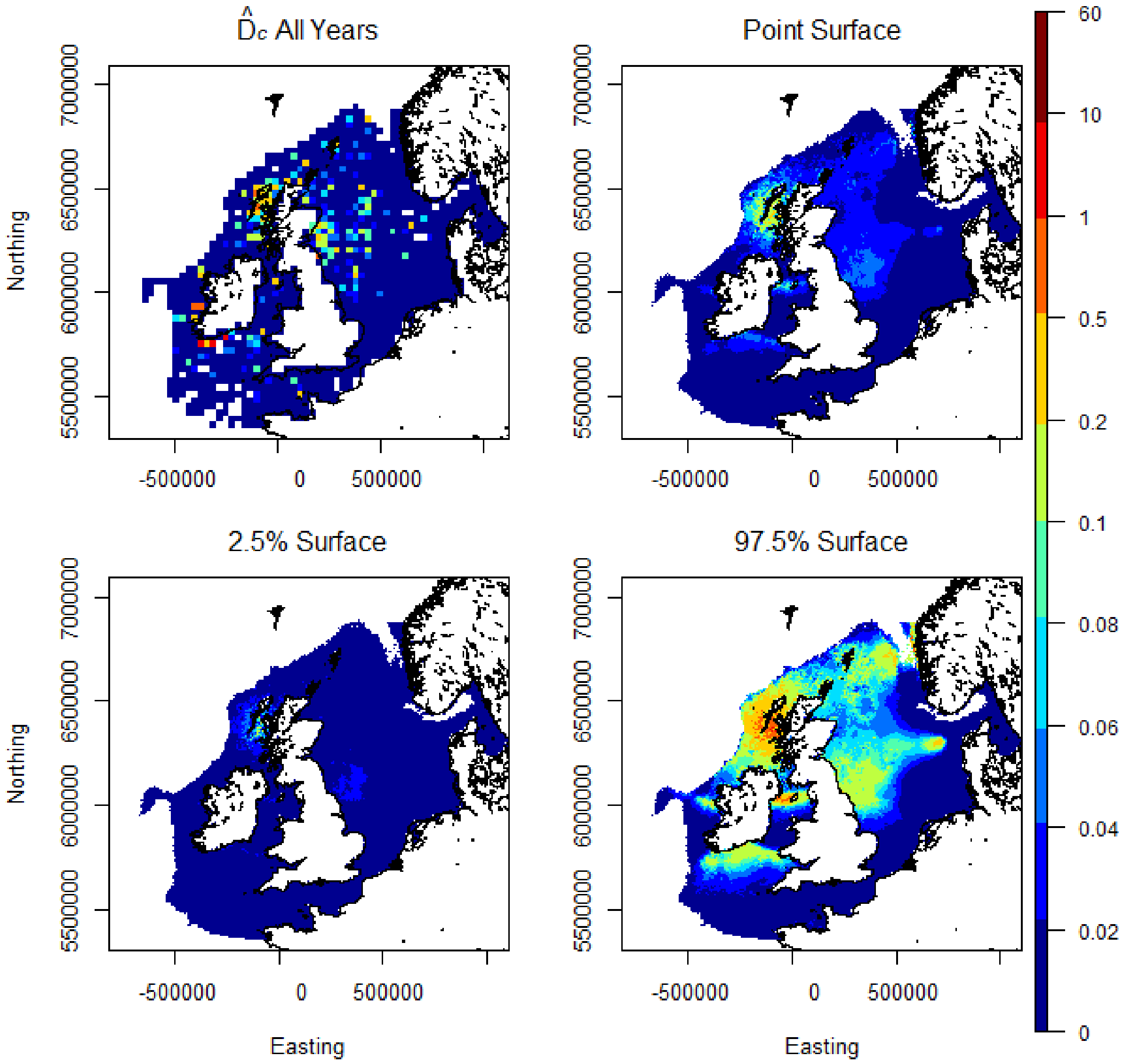

Figure 12. Predicted minke whale densities for summer 2010. Top left: input densities $\hat{D}_c$ for all summers. Top right: point estimate of cell densities. Bottom left: lower (2.5%) confidence limit on cell densities. Bottom right: upper (97.5%) confidence limit on cell densities (animals/km^2). Note that the top left plot exaggerates the spatial coverage of the relevant effort.

Table 11. Estimated abundances of minke whale in 2010. Please refer to Appendices 6 and 7 for an explanation of how these figures should be scaled to use in the context of agreed Management Unit reference populations.

	Winter			***Spring***			***Summer***			***Autumn***		
	Point	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***
Shallow *OSPAR2*	520	150	4370	1470	130	12860	8730	4430	23260	420	80	1810
OSPAR3	1610	720	8100	2810	460	21440	7770	4030	28840	930	230	4520
Truncated *mATL*	2140	970	12440	4160	600	30820	15230	8070	48770	1320	330	6180
Truncated EEZ	1590	640	9590	2970	340	22440	12980	6690	40070	1020	250	4510
Moray Firth	20	0	130	30	0	260	210	80	540	20	0	60
Firth of Forth	20	0	150	60	0	480	360	140	990	20	0	60
Atlantic Array	40	10	210	110	10	830	280	120	950	50	10	200
Islay	10	0	70	20	0	190	100	40	400	10	0	40
Solway Firth	0	0	10	10	0	40	30	10	110	0	0	30
North	30	10	160	30	0	260	170	70	550	10	0	80
East Orkney	10	0	70	10	0	100	80	30	270	10	0	30
Lewis	20	10	110	20	0	200	100	40	360	10	0	50
Kyle Rhea	0	0	10	0	0	20	10	0	40	0	0	10
Sound of Islay	0	0	10	0	0	30	10	10	60	0	0	10
Argyll Array	50	20	270	90	10	690	320	140	1240	30	10	150
Hastings	0	0	0	0	0	0	0	0	0	0	0	0
IOW	0	0	0	0	0	20	0	0	20	0	0	10
Dogger Bank	10	0	200	100	10	940	530	240	1400	20	0	70
South Dogger Bank	0	0	100	70	10	650	310	170	1000	20	0	60
Norfolk Bank	0	0	10	10	0	80	30	10	120	0	0	10
Irish Sea	10	0	60	40	10	170	190	80	620	20	0	180
Strangford Lough	0	0	10	10	0	30	30	10	140	0	0	40
Shetlands	10	0	40	10	0	60	40	10	150	0	0	10

3.5.4 Bottlenose Dolphin

Analysis of the 131448 segments for bottlenose dolphins revealed a large number of zero segments with only 1154 non-zero segments with a mean $\hat{D}_c$ of 0.067, SE = 0.004 (densities ranged from 0.004 in 1994 to 0.129 in 2008) animals/km^2 across the entire region of interest. The final chosen model consisted of single smooths of all the available predictors and a 2D spatial smooth (Table 9).

Predicted bottlenose dolphin numbers are given in Table 12. A density surface map for bottlenose in the summer of 2010 is given in Figure 13. Some consistent patterns in bottlenose dolphin predicted density were found across the different time periods (Appendix figures A4.13 – A4.18). The animals were essentially coastal, with particular consistent regions of high density in Cardigan Bay, the Moray Firth and the west coast of Ireland. However, they are also occasionally found in other regions, e.g. the Bristol Channel, off the coast of Fife and the Channel Islands. The predicted high density in the Moray Firth is not clear in Figure 13, so a close-up of the northern British Isles is supplied in Figure 14.

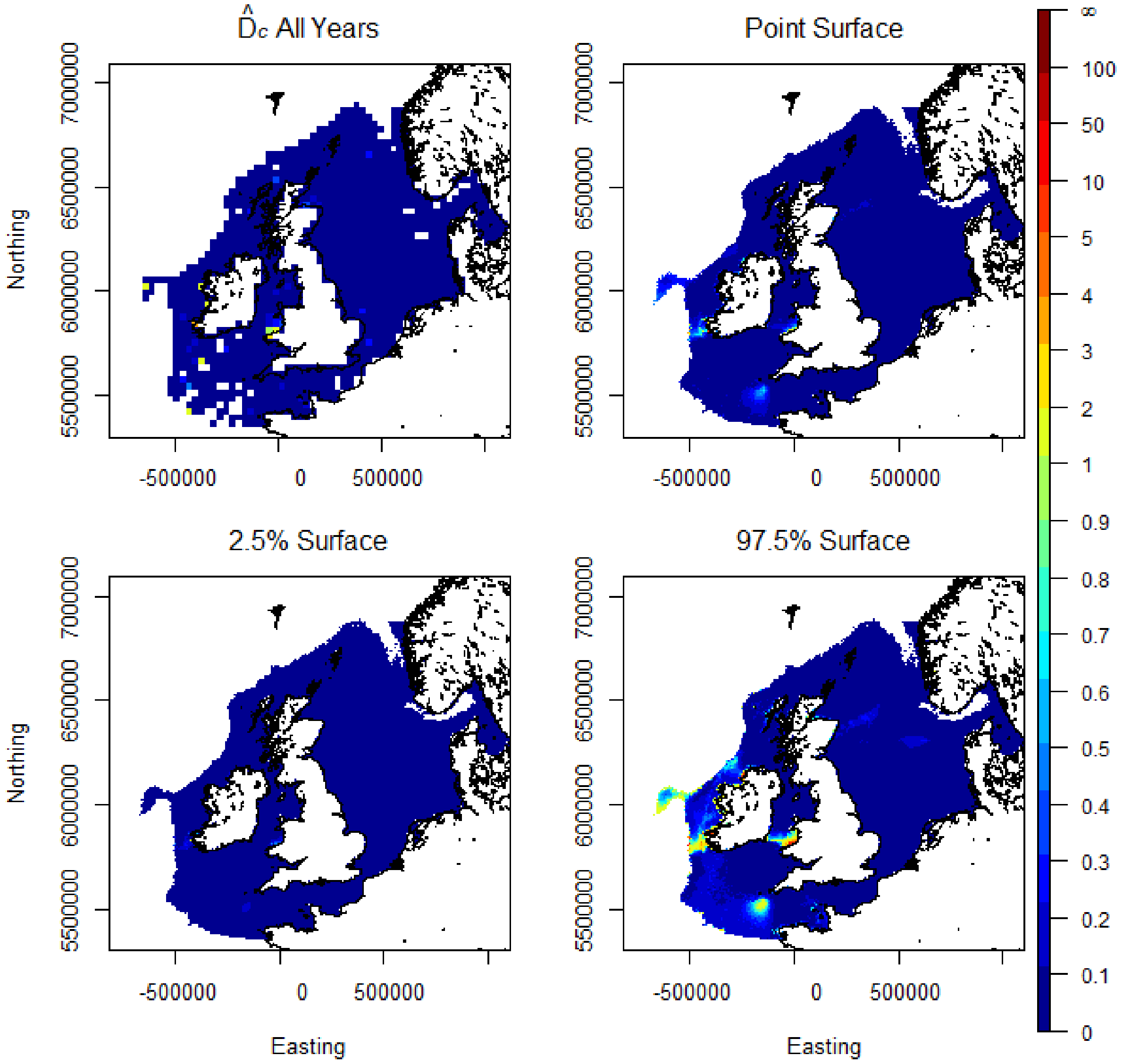

Figure 13. Predicted bottlenose dolphin densities for summer 2010. Top left: input densities $\hat{D}_c$ summer all years. Top right: point estimate of cell densities. Bottom left: lower (2.5%) confidence limit on cell densities. Bottom right: upper (97.5%) confidence limit on cell densities (animals/km^2). Note that the top left plot exaggerates the spatial coverage of the relevant effort.

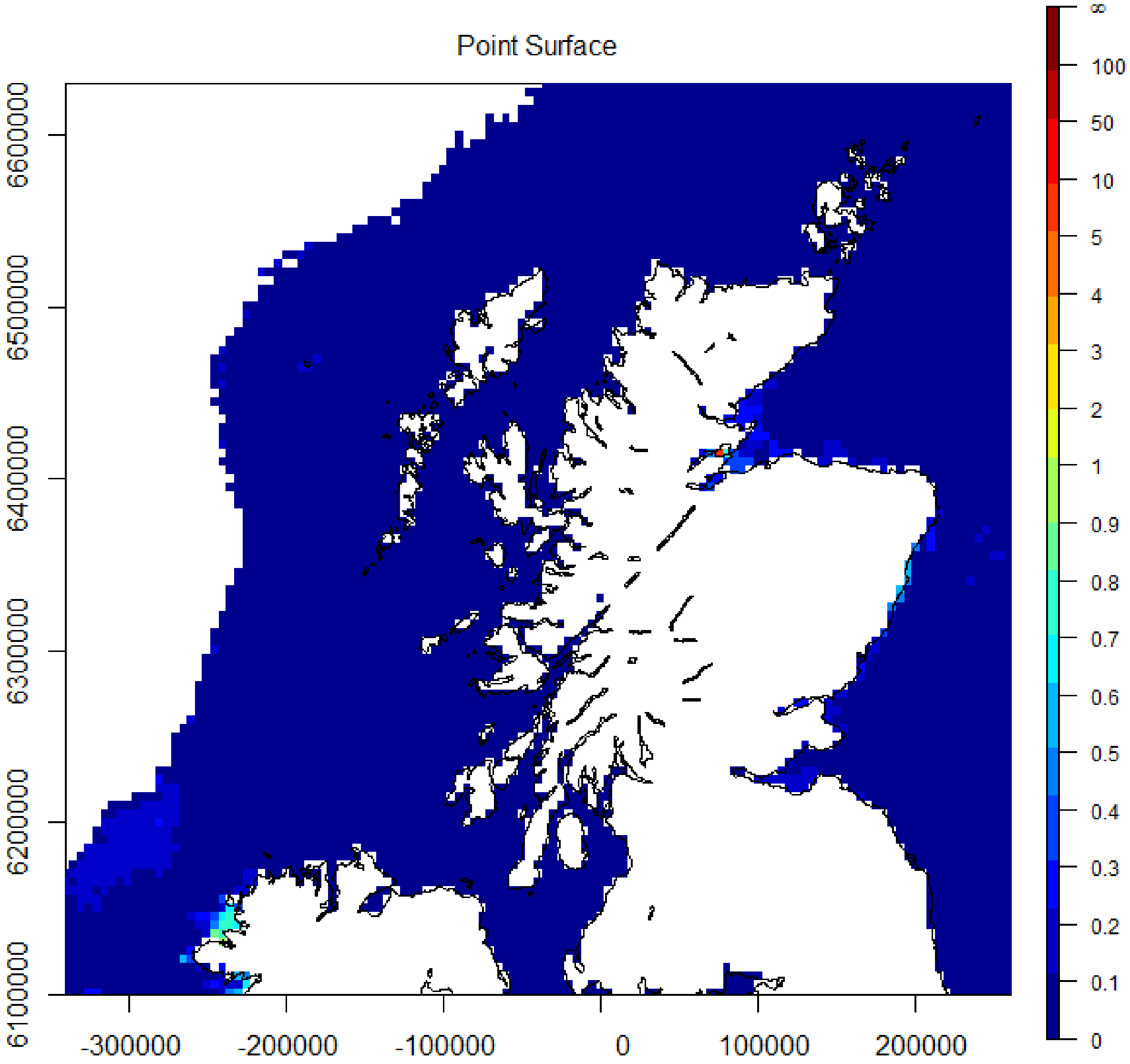

Figure 14. Close up of predicted bottlenose dolphin densities for the northern British Isles in summer 2010.

Table 12. Estimated abundances of bottlenose dolphin in 2010. Please refer to Appendices 6 and 7 for an explanation of how these figures should be scaled to use in the context of agreed Management Unit reference populations.

	Winter			***Spring***			***Summer***			***Autumn***		
	Point	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***
Shallow OSPAR2	2550	1210	1.16E+10	5520	1690	18410	5840	2830	12970	2370	1180	4160
OSPAR3	11780	5830	37260	18200	7810	60590	20920	8850	68000	7660	3110	21350
Truncated mATL	16510	9060	45312900	26450	11200	86540	29950	14200	88250	11170	4910	27650
Truncated EEZ	6380	3180	17850	10690	3560	36200	11710	5320	31720	4570	2160	10930
Moray Firth	170	60	330	250	60	780	230	80	450	110	40	190
Firth of Forth	230	90	450	460	130	1340	430	190	780	190	80	290
Atlantic Array	70	20	330	100	30	390	120	40	490	40	20	170
Islay	0	0	20	0	0	20	0	0	20	0	0	10
Solway Firth	10	0	20	10	0	50	10	0	50	10	0	20
North	10	0	30	20	0	50	20	10	30	10	0	10
East Orkney	20	10	50	30	10	110	30	10	50	10	10	20
Lewis	10	10	40	20	10	60	20	10	50	10	0	20
Kyle Rhea	10	0	30	10	0	30	10	0	40	0	0	10
Sound of Islay	0	0	10	0	0	10	0	0	10	0	0	10
Argyll Array	10	0	60	20	10	60	20	10	80	10	0	30
Hastings	0	0	10	0	0	10	10	0	10	0	0	10
IOW	30	10	90	40	10	130	50	20	130	20	10	50
Dogger Bank	0	0	590	80	20	310	100	30	320	30	10	80
South Dogger Bank	0	0	240	30	10	110	30	10	100	10	0	30
Norfolk Bank	0	0	120	20	0	50	20	0	60	10	0	20
Irish Sea	10	0	60	30	10	110	30	10	100	10	0	30
Strangford Lough	0	0	10	0	0	20	0	0	10	0	0	0
Shetlands	10	0	20	10	0	40	10	0	30	10	0	10

**Caused by great uncertainty in the estimates off Ireland.*

3.5.5 Short-beaked Common Dolphin

Analysis of the 131448 segments for short-beaked common dolphin revealed only 1468 non-zero segments with a mean $\hat{D}_c$ of 0.117 (SE = 0.009) (ranging from 0.038 in 2003 to 0.361 in 1995) animals/km^2 across the entire region of interest. The final chosen model consisted of single smooths of all the available predictors and a 2D spatial smooth (Table 9).

A density surface map for short-beaked common dolphin in the summer of 2010 is given in Figure 15, although the highest predicted point estimate is for autumn (Table 13). Short-beaked common dolphins are predicted to occur predominantly in the south-west of the prediction area, to the west of Ireland and the Hebrides (Appendix figures A4.19 – A4.24). They seldom occur in the North Sea.

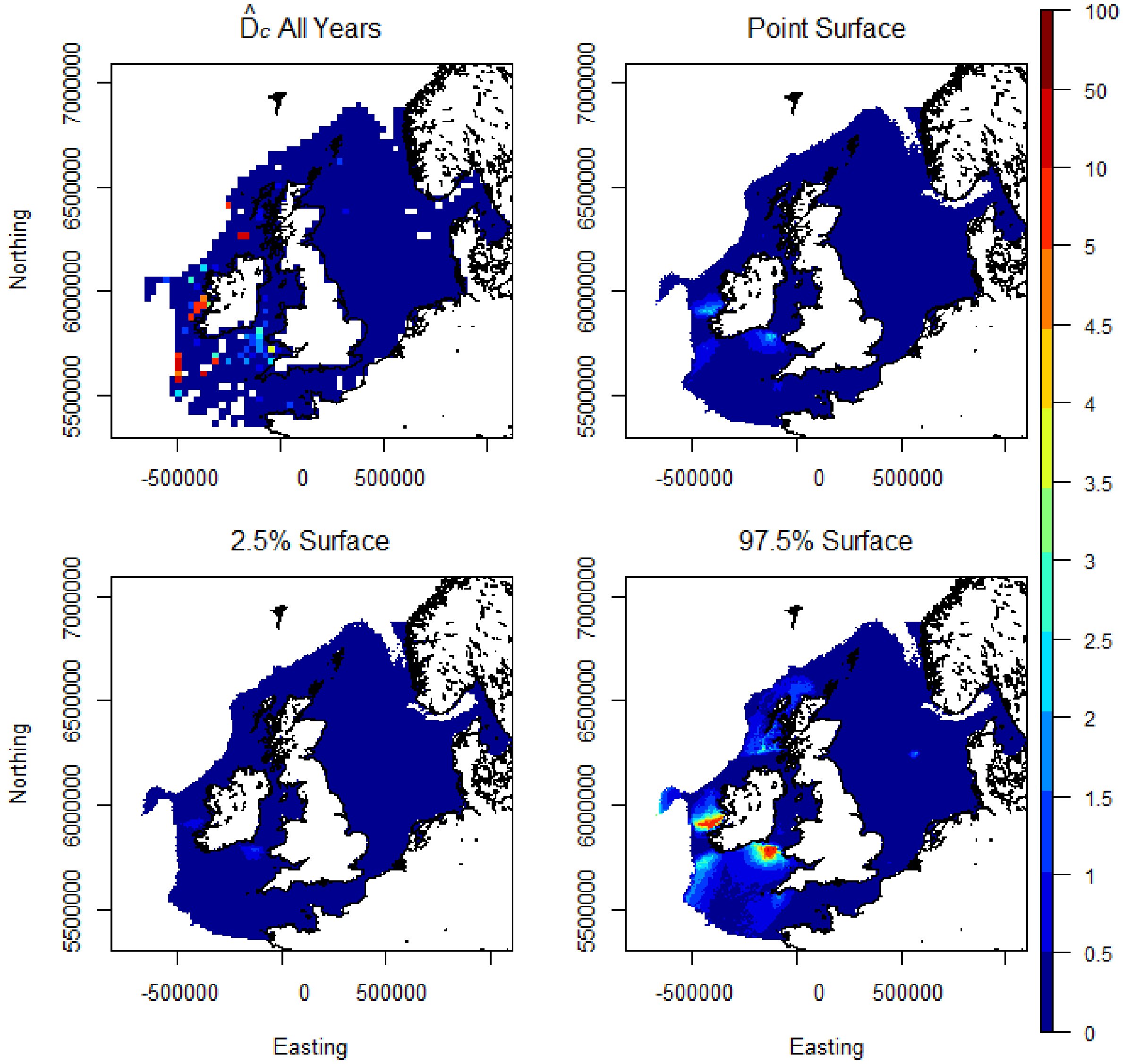

Figure 15. Predicted short-beaked common dolphin densities for summer 2010. Top left: input densities $\hat{D}_c$ summer all years. Top right: point estimate of cell densities. Bottom left: lower (2.5%) confidence limit on cell densities. Bottom right: upper (97.5%) confidence limit on cell densities (animals/km^2). Note that the top left plot exaggerates the spatial coverage of the relevant effort.

Table 13. Estimated abundances of short-beaked common dolphin in 2010. Please refer to Appendices 6 and 7 for an explanation of how these figures should be scaled to use in the context of agreed Management Unit reference populations.

	Winter			***Spring***			***Summer***			***Autumn***		
	Point	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***
Shallow OSPAR2	2800	1570	11740	3720	1710	8860	5440	3010	16100	18880	9760	40710
OSPAR3	65020	31600	147530	71610	34220	163220	90140	47330	236170	363860	182670	807800
Truncated mATL	71640	34650	162290	78030	37610	176270	98870	52230	255130	396220	198040	872300
Truncated EEZ	21790	10940	54100	29030	13520	70210	39370	20770	111670	151930	77260	341090
Moray Firth	10	0	50	30	10	90	90	40	300	200	80	570
Firth of Forth	10	0	40	20	10	90	60	20	220	130	40	450
Atlantic Array	2430	1260	5840	6260	2720	16260	7990	4160	25100	33120	16540	79450
Islay	10	0	40	30	10	100	50	20	250	180	60	690
Solway Firth	0	0	0	0	0	0	0	0	0	0	0	10
North	80	30	390	140	40	500	400	150	1820	1010	380	3320
East Orkney	0	0	10	0	0	20	20	10	60	40	10	110
Lewis	70	30	260	90	40	290	200	90	860	590	250	1690
Kyle Rhea	0	0	10	0	0	10	0	0	30	20	10	70
Sound of Islay	0	0	0	0	0	10	10	0	30	20	10	90
Argyll Array	250	110	860	470	200	1790	730	320	3640	2650	1120	9280
Hastings	0	0	0	0	0	0	0	0	0	0	0	0
IOW	0	0	0	0	0	10	10	0	20	20	10	50
Dogger Bank	10	0	60	10	0	30	10	0	60	20	0	110
South Dogger Bank	0	0	10	0	0	10	0	0	10	10	0	40
Norfolk Bank	10	0	40	10	0	40	20	10	50	60	30	180
Irish Sea	10	0	50	50	20	160	80	30	260	310	110	860
Strangford Lough	0	0	20	10	0	50	20	10	110	80	20	350
Shetlands	0	0	0	0	0	0	0	0	0	0	0	0

3.5.6 Risso's Dolphin

Analysis of the 131448 segments for Risso's dolphin revealed a very large proportion of zero segments, with only 155 non-zero segments. The mean $\hat{D}_c$ was 0.004 (SE<0.001) (ranging from 0 in 1998 to 0.013 in 1999) animals/km^2 across the entire region of interest. The final chosen model consisted of single smooths of *Year*, *Dayofyear* and *Depth* and a 2D spatial smooth (Table 9).

A predicted density surface map for Risso's dolphin in the summer of 2010 is given in Figure 16. Risso's dolphins are detected infrequently in the Phase III region, but they are predicted fairly consistently off the Hebrides, between Anglesey and the Isle of Man, off county Kerry, off county Wexford and in the western English Channel (Appendix figures A4.25 – A4.30). They seldom occur in the North Sea (Table 14).

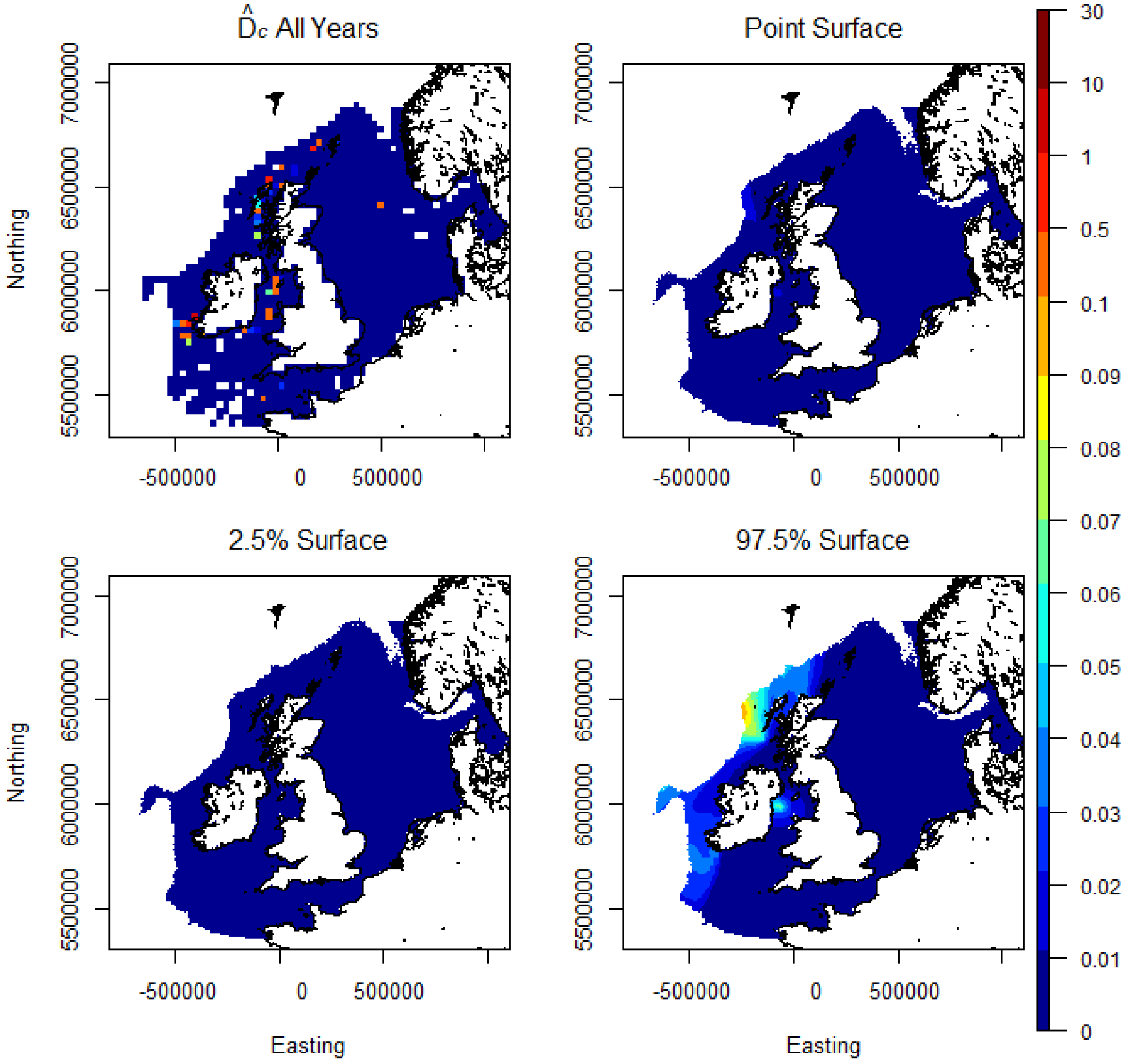

Figure 16. Predicted Risso's dolphin densities for summer 2010. Top left: input densities $\hat{D}_c$ summer all years. Top right: point estimate of cell densities.

Bottom left: lower (2.5%) confidence limit on cell densities. Bottom right: upper (97.5%) confidence limit on cell densities (animals/km^2). Note that the top left plot exaggerates the spatial coverage of the relevant effort.

Table 14. Estimated abundances of Risso's dolphins in 2010. Please refer to Appendices 6 and 7 for an explanation of how these figures should be scaled to use in the context of agreed Management Unit reference populations.

	Winter			***Spring***			***Summer***			***Autumn***		
	Point	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***	***Point***	***2.5%***	***97.5%***
Shallow OSPAR2	20	0	80	260	0	1720	130	0	970	10	0	50
OSPAR3	130	0	480	1730	0	10940	890	10	5420	60	0	310
Truncated mATL	160	0	630	2190	0	14280	1120	10	7140	70	0	420
Truncated EEZ	90	0	370	1290	0	8140	660	10	4100	40	0	230
Moray Firth	0	0	0	0	0	40	0	0	10	0	0	0
Firth of Forth	0	0	0	10	0	50	0	0	20	0	0	0
Atlantic Array	0	0	0	0	0	90	0	0	40	0	0	0
Islay	0	0	0	0	0	0	0	0	0	0	0	0
Solway Firth	0	0	0	10	0	20	0	0	10	0	0	0
North	0	0	10	30	0	150	10	0	90	0	0	10
East Orkney	0	0	0	0	0	20	0	0	10	0	0	0
Lewis	0	0	0	20	0	110	10	0	60	0	0	0
Kyle Rhea	0	0	0	0	0	10	0	0	0	0	0	0
Sound of Islay	0	0	0	0	0	0	0	0	0	0	0	0
Argyll Array	0	0	10	20	0	170	10	0	70	0	0	0
Hastings	0	0	0	0	0	0	0	0	0	0	0	0
IOW	0	0	0	0	0	0	0	0	0	0	0	0
Dogger Bank	0	0	0	0	0	20	0	0	10	0	0	0
South Dogger Bank	0	0	0	0	0	10	0	0	10	0	0	0
Norfolk Bank	0	0	0	0	0	0	0	0	0	0	0	0
Irish Sea	0	0	10	70	0	280	30	0	160	0	0	10
Strangford Lough	0	0	0	0	0	20	0	0	10	0	0	0
Shetlands	0	0	0	0	0	10	0	0	0	0	0	0

3.5.7 White-beaked Dolphin

Only small numbers of white-beaked dolphins were seen, with only 549 out of the 131448 available segments having the animals present. The mean $\hat{D}_c$ was 0.021 (SE = 0.002) (ranging from 0.004 in 2000 to 0.053 in 1997) animals/km^2 across the entire region of interest. The final chosen model consisted of single smooths of *Year*, *Slope, Dayofyear* and *Depth* and a 2D spatial smooth (Table 9).

Predicted densities of animals are typically low, but are highest off the Hebrides and the northern North Sea (Figure 17). The predicted densities for the Celtic Sea, Irish Sea, English Channel are typically close to zero (Table 15), Appendix figures A4.31 – A4.36).

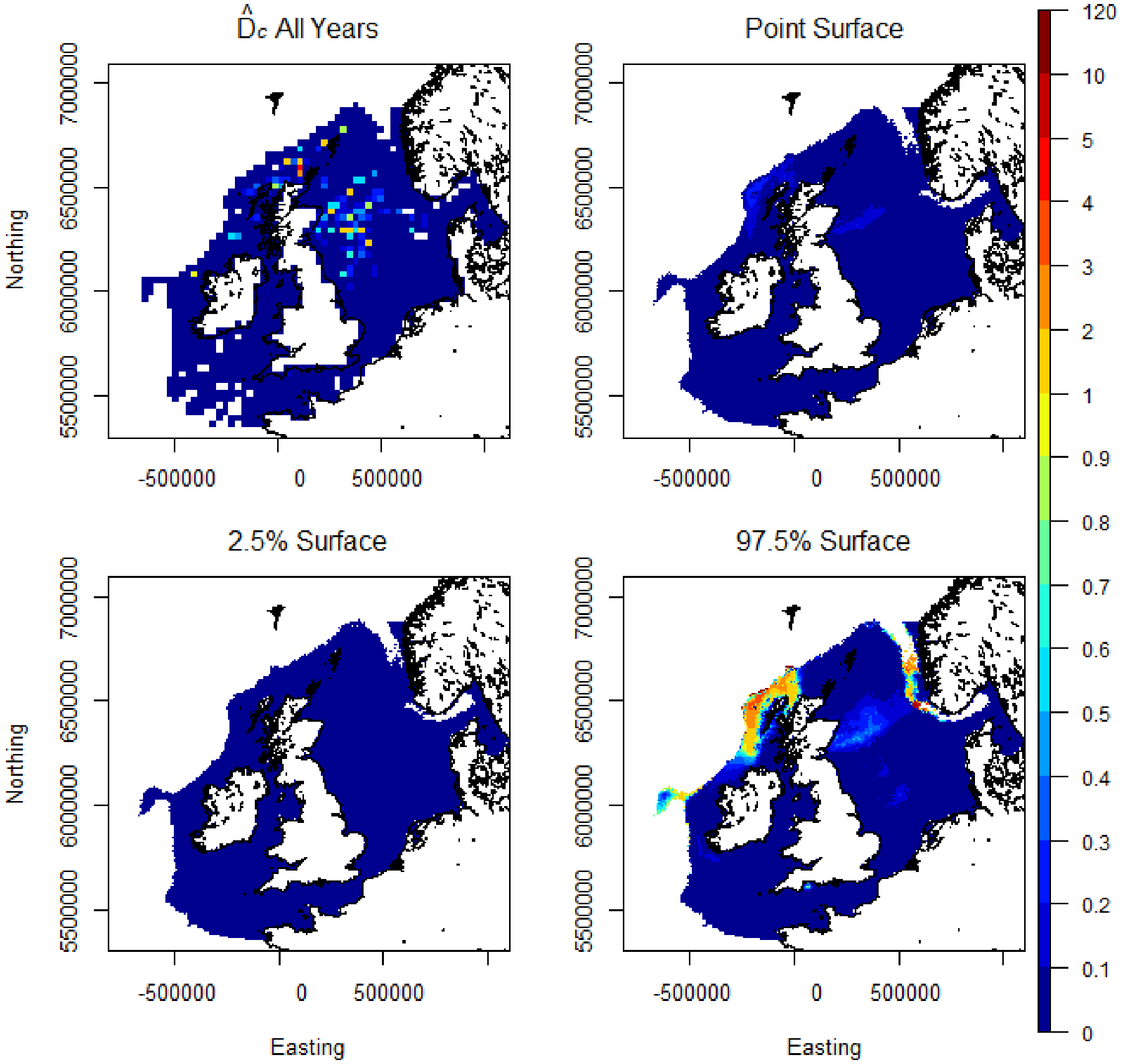

Figure 17. Predicted white-beaked dolphin densities for summer 2010. Top left: input densities $\hat{D}_c$ summer all years. Top right: point estimate of cell densities. Bottom left: lower (2.5%) confidence limit on cell densities. Bottom right: upper (97.5%) confidence limit on cell densities (animals/km^2). Note that the top left plot exaggerates the spatial coverage of the relevant effort.

Table 15. Estimated abundances of white-beaked dolphin in 2010. Please refer to Appendices 6 and 7 for an explanation of how these figures should be scaled to use in the context of agreed Management Unit reference populations.

	Winter			**Spring**			**Summer**			**Autumn**		
	Point	**2.5%**	**97.5%**	**Point**	**2.5%**	**97.5%**	**Point**	**2.5%**	**97.5%**	**Point**	**2.5%**	**97.5%**
Shallow *OSPAR2*	5320	2530	15670	22670	11000	86070	9310	5200	47440	7020	3430	16280
OSPAR3	3660	1080	35800	15610	5300	143220	6410	2210	114050	4830	1920	28520
Truncated *mATL*	8250	3480	86120	35200	15420	352070	14450	6910	225600	10900	5440	58880
EEZ	7210	2970	74220	30750	13430	283430	12620	5760	197880	9520	4730	47480
Moray Firth	40	20	110	180	80	400	70	40	200	60	20	120
Firth of Forth	410	170	1110	1760	620	4530	720	360	1840	540	220	1130
Atlantic Array	0	0	0	0	0	20	0	0	10	0	0	10
Islay	0	0	10	10	0	30	0	0	20	0	0	10
Solway Firth	0	0	0	0	0	0	0	0	0	0	0	0
North	10	10	40	50	30	140	20	10	80	20	10	40
East Orkney	0	0	10	10	10	40	10	0	20	0	0	10
Lewis	80	20	740	340	120	2780	140	40	1730	100	40	570
Kyle Rhea	0	0	10	0	0	30	0	0	20	0	0	0
Sound of Islay	0	0	0	0	0	0	0	0	0	0	0	0
Argyll Array	20	10	120	80	30	570	30	10	360	20	10	100
Hastings	0	0	0	0	0	0	0	0	0	0	0	0
IOW	0	0	0	0	0	10	0	0	0	0	0	0
Dogger Bank	290	130	650	1230	500	2830	510	290	1190	380	180	730
South Dogger Bank	170	80	380	710	290	1790	290	170	610	220	90	420
Norfolk Bank	20	10	40	70	30	220	30	20	60	20	10	50
Irish Sea	0	0	0	0	0	10	0	0	10	0	0	0
Strangford Lough	0	0	0	0	0	0	0	0	0	0	0	0
Shetlands	0	0	10	10	0	30	0	0	10	0	0	10

3.5.8 Atlantic White-sided Dolphin

Only small numbers of white-sided dolphins were seen, with only 71 out of the 131448 available segments having the animals present. The mean $\hat{D}_c$ was thus 0.006 (SE= 0.002) (ranging from 0 in 2001, 2004 and 2009 to 0.037 in 1997) animals/km^2 across the entire region of interest.

The final chosen model consisted of single smooths of *Year*, *Dayofyear* and *Depth* and a 2D spatial smooth (Table 9).

Predicted densities of animals are very low (Figure 26, Figures A4.37 – A4.39, Table 16). White-sided dolphins are most often predicted in the northwest approaches and north of Britain.

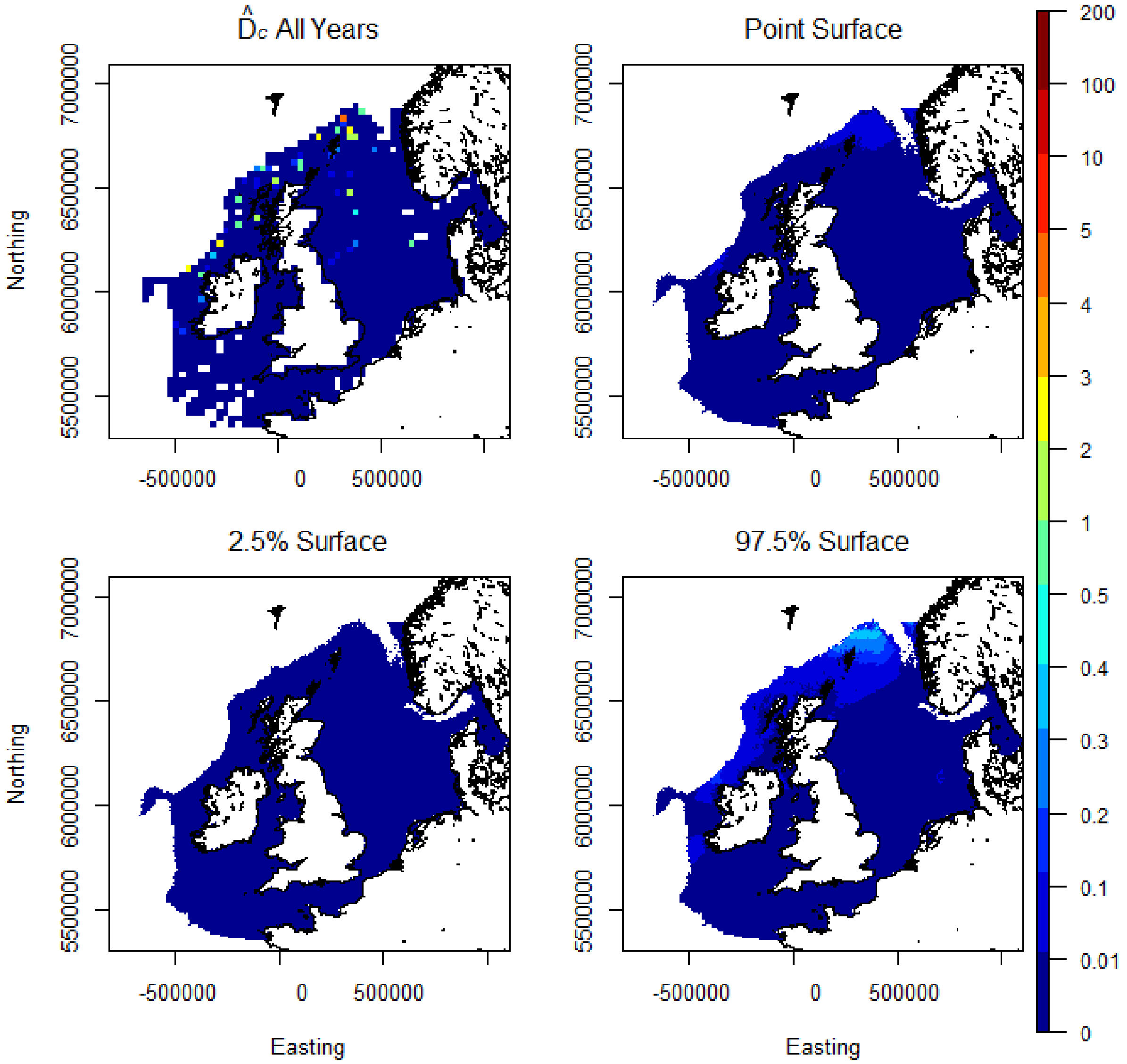

Figure 18. Predicted white-sided dolphins for summer 2010. Top left: mean $\widehat{D}_c$ summers all years. Top right: predicted densities summer 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

Table 16. Estimated abundances of white-sided dolphin in summer 2010. Geographic areas are as given in the text and do not necessarily refer to the regions as the terms are commonly used. Please refer to Appendices 6 and 7 for an explanation of how these figures should be scaled to use in the context of agreed Management Unit reference populations.

	Winter			**Spring**			**Summer**			**Autumn**		
	Point	**2.5%**	**97.5%**	**Point**	**2.5%**	**97.5%**	**Point**	**2.5%**	**97.5%**	**Point**	**2.5%**	**97.5%**
Shallow *OSPAR2*	80	0	1080	740	100	5170	1170	130	10810	490	30	3750
OSPAR3	20	0	260	230	30	1540	360	30	2750	150	10	820
Truncated *mATL*	90	0	1230	900	130	5920	1410	160	12700	590	40	4300
EEZ	80	0	1070	770	100	5060	1210	140	11390	500	30	3790
Moray Firth	0	0	0	0	0	10	0	0	20	0	0	10
Firth of Forth	0	0	0	0	0	20	0	0	30	0	0	20
Atlantic Array	0	0	0	0	0	0	0	0	0	0	0	0
Islay	0	0	0	0	0	0	0	0	0	0	0	0
Solway Firth	0	0	0	0	0	0	0	0	0	0	0	0
North	0	0	10	0	0	30	10	0	70	0	0	20
East Orkney	0	0	0	0	0	0	0	0	10	0	0	0
Lewis	0	0	0	0	0	10	0	0	20	0	0	0
Kyle Rhea	0	0	0	0	0	0	0	0	0	0	0	0
Sound of Islay	0	0	0	0	0	0	0	0	0	0	0	0
Argyll Array	0	0	0	0	0	10	0	0	20	0	0	10
Hastings	0	0	0	0	0	0	0	0	0	0	0	0
IOW	0	0	0	0	0	0	0	0	0	0	0	0
Dogger Bank	0	0	0	0	0	0	0	0	0	0	0	0
South Dogger Bank	0	0	0	0	0	0	0	0	0	0	0	0
Norfolk Bank	0	0	0	0	0	0	0	0	0	0	0	0
Irish Sea	0	0	0	0	0	0	0	0	0	0	0	0
Strangford Lough	0	0	0	0	0	0	0	0	0	0	0	0
Shetlands	0	0	0	0	0	20	0	0	50	0	0	10

3.6 Temporal Trends

Predicted yearly abundance estimates for each species in summer within the four large regions (OSPAR2, OSPAR3, mATL and EEZ) are shown in Figures 19 and 21 to 26, together with 95% pointwise confidence limits (i.e. the lower 2.5th and upper 97.5th quantile at each time point of the 500 bootstrap smooths). Also shown are the average abundances in each of the three reporting periods and, for reporting periods 1 and 2, 95% pairwise confidence intervals on the difference between the average abundances for these periods and the average abundance for period 3. The interpretation of these intervals is as follows: if the interval for period 1 or 2 does not contain the average abundance for period 3 then there has been a statistically significant population change between the relevant periods. Similar information is contained in Table 17, which shows the estimated annual population change for each region of interest, together with 95% confidence limits on the change. If these confidence limits do not contain 1.0 then there has been a statistically significant population change.

Another way to express the uncertainty on population change estimates is the coefficient of variation (CV) of the population change indices (the Δs). These are given in Table 18. Note that these are not CVs of the annual abundance estimates; rather they are CVs for the ratio of average population abundances. As such, they should be expected to be much smaller than CVs on the abundance estimates, and indeed they are mostly in the range 0.02 to 0.10. Narrower CVs correspond, in general, with narrower pairwise confidence intervals on Δ. Comparing the CVs between time periods, the population change between time period 1 and 3 was, in general, measured more precisely than that between periods 2 and 3. Comparing between species, CVs were generally higher for harbour porpoise, where a more complex model was fit, than for the other species.

Harbour porpoise was the only species for which a spatio-temporal interaction model was retained during model selection (Section 3.5.1). For this species, there were some regional differences in estimated trend, as illustrated in Figure 20 for four of the smaller regions (note that the y-axis on this figure is on the log scale, to better illustrate the pattern), and also shown in Table 17. Confidence intervals were in general wide, both on the estimated population abundances, and on the population change estimates. Few of the population changes were statistically significant, although there were estimated increases in 8 regions.

For the other species, spatio-temporal smooths were not selected, and hence the fitted spatial density surface does not change over time. It is still possible to observe regional variation in trend if sea surface temperature was a selected covariate (as it was in many of the models, Section 3.5.1), but in practice the estimated trends were identical to two decimal places in all regions. Therefore, only trends for the four large regions are shown in Table 17. In any case, without a spatio-temporal interaction it is debatable how valid trend estimates are at the smaller regional scale –(see Discussion).

Minke whale appeared to show a generally declining trend, although there was weak evidence of an increase around year 2000 (Figure 21). The average annual population change between reporting periods 1 and 3 was 0.91 (i.e. a decline of 9% per year) with 95%CI 0.84 - 0.99 (Table 18), and hence was significantly different from no change.

Bottlenose dolphin (Figure 22) and common dolphin (Figure 23) showed no significant population changes (Table 17), although the latter appeared to have some population oscillations on an approximately decadal time scale.

Risso’s dolphin showed significant declines between reporting periods 1 and 3 (average annual change 0.88 (i.e. a decline of 12% per year) with 95% CI approx. 0.73 - 0.95

depending on the region, Table 17), and although a monotonic decline is not incompatible with the confidence intervals on population change, it is also possible that the underlying trend was non-monotonic with a particularly large peak around 1997 (Figure 24).

White-beaked dolphin showed no significant population trends; confidence intervals on the whole were wide (Figure 25).

White-sided dolphin showed significant declines between reporting periods 1 and 3 (annual population change 0.80 with 95% CI 0.73 - 0.90) and 2 and 3 (0.71 with 95% CI approx.. 0.51 - 0.95 depending on the region, Table 17). The former may be partly due to a peak in density around 1997, but there is considerable uncertainty on this peak (Figure 26).

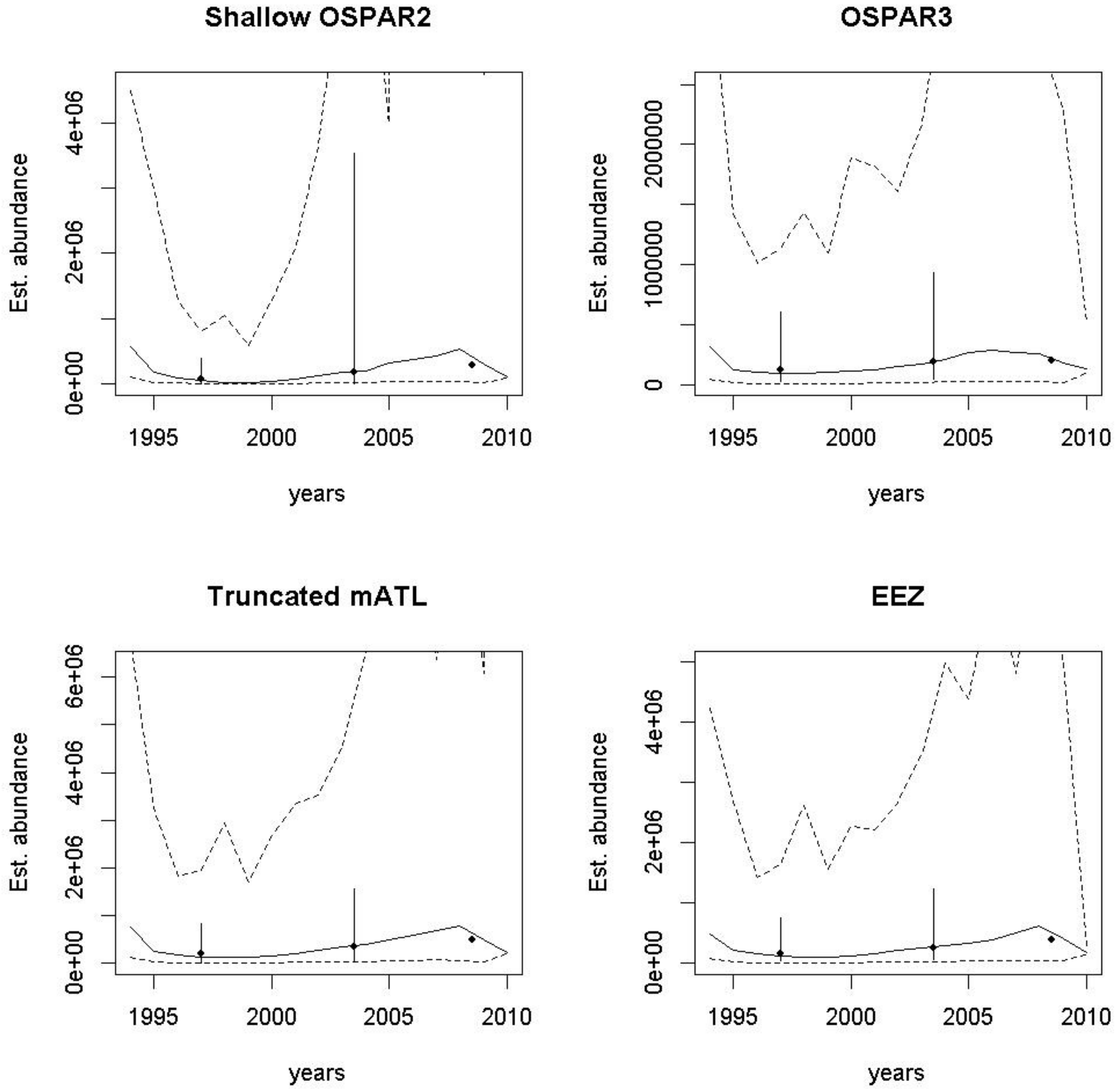

Figure 19. Harbour porpoise predicted abundances (solid black line) and pointwise 95%CI (dashed black line) in four regions. Also shown are geometric mean abundances in each of three reporting period (dot at reporting period midpoints), and 95% pairwise confidence interval (vertical line), comparing reporting periods 1 and 2 with reporting period 3. If the confidence interval from an earlier reporting period does not overlap the mean abundance from reporting period 3, the difference is statistically significant at the 5% level, and is shown in red. For harbour porpoise, there are no significant trends.

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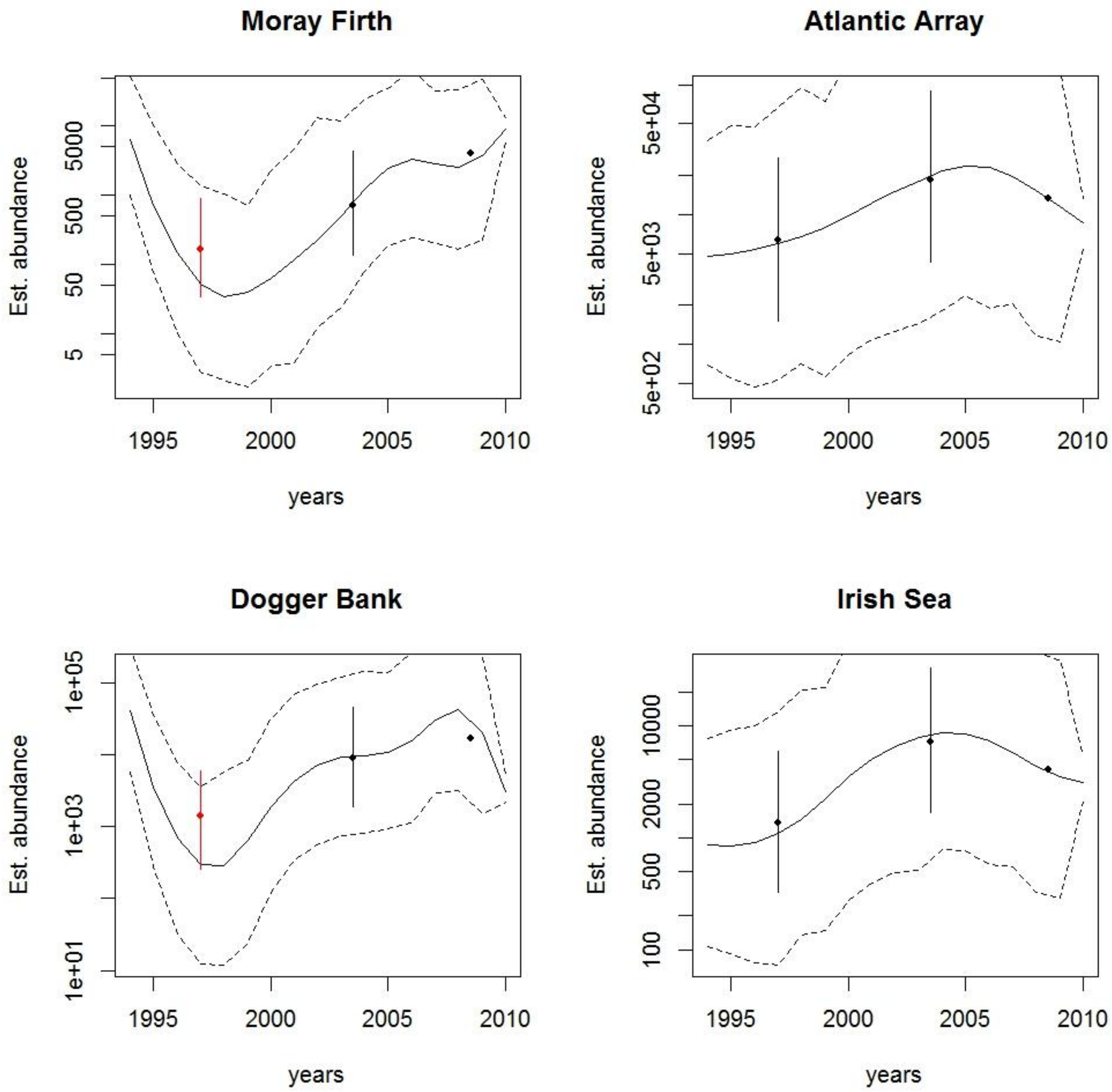

Figure 20. Harbour porpoise predicted abundances and pointwise 95% CI in four of the smaller regions of interest. Note that the y-axis is on the log scale, to better illustrate the differences in temporal pattern allowed by this spatio-temporal interaction model. See previous figure for an explanation of the symbols.

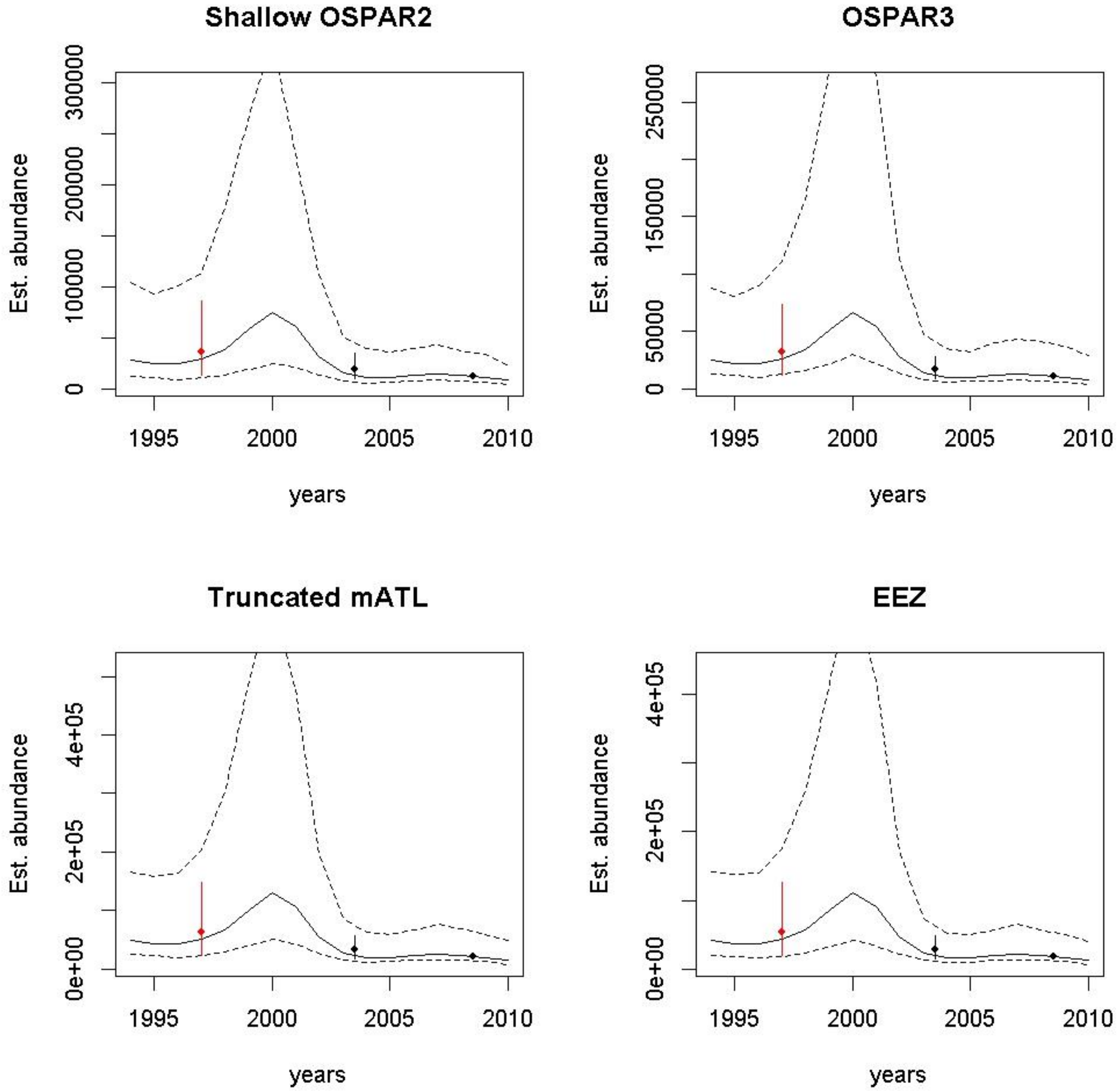

Figure 21. Minke whale predicted abundances (solid black line) and pointwise 95% CI (dashed black line) in four regions. Also shown are geometric mean abundances in each of three reporting period (dot at reporting period midpoints), and 95% pairwise confidence interval (vertical line), comparing reporting periods 1 and 2 with reporting period 3. If the confidence interval from an earlier reporting period does not overlap the mean abundance from reporting period 3, the difference is statistically significant at the 5% level, and is shown in red. For minke whale, there is estimated to have been a significant decline between the first and third reporting period.

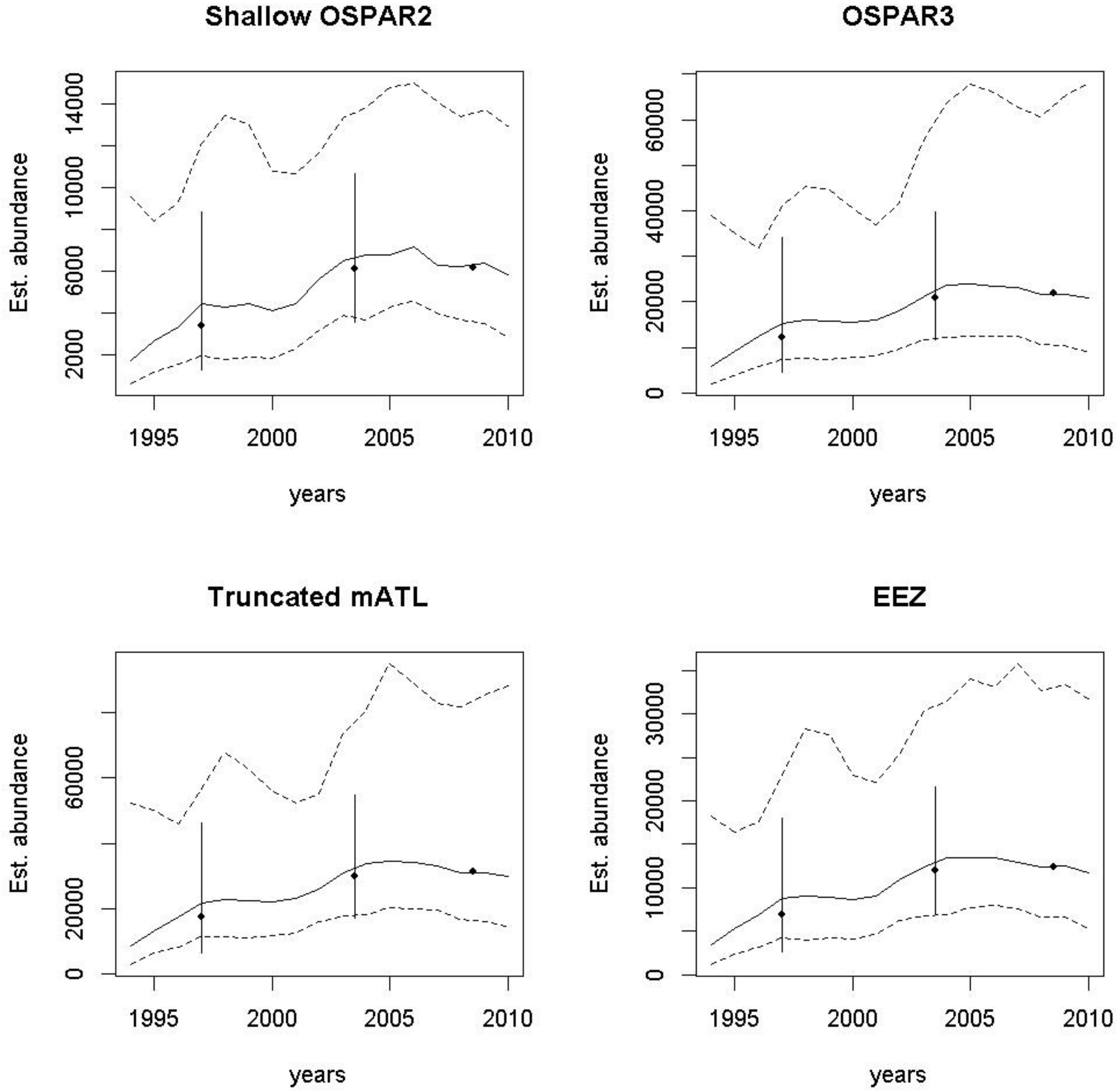

Figure 22. Bottlenose dolphin predicted abundances (solid black line) and pointwise 95% CI (dashed black line) in four regions. Also shown are geometric mean abundances in each of three reporting period (dot at reporting period midpoints), and 95% pairwise confidence interval (vertical line), comparing reporting periods 1 and 2 with reporting period 3. If the confidence interval from an earlier reporting period does not overlap the mean abundance from reporting period 3, the difference is statistically significant at the 5% level, and is shown in red. For bottlenose dolphin, there are no significant trends.

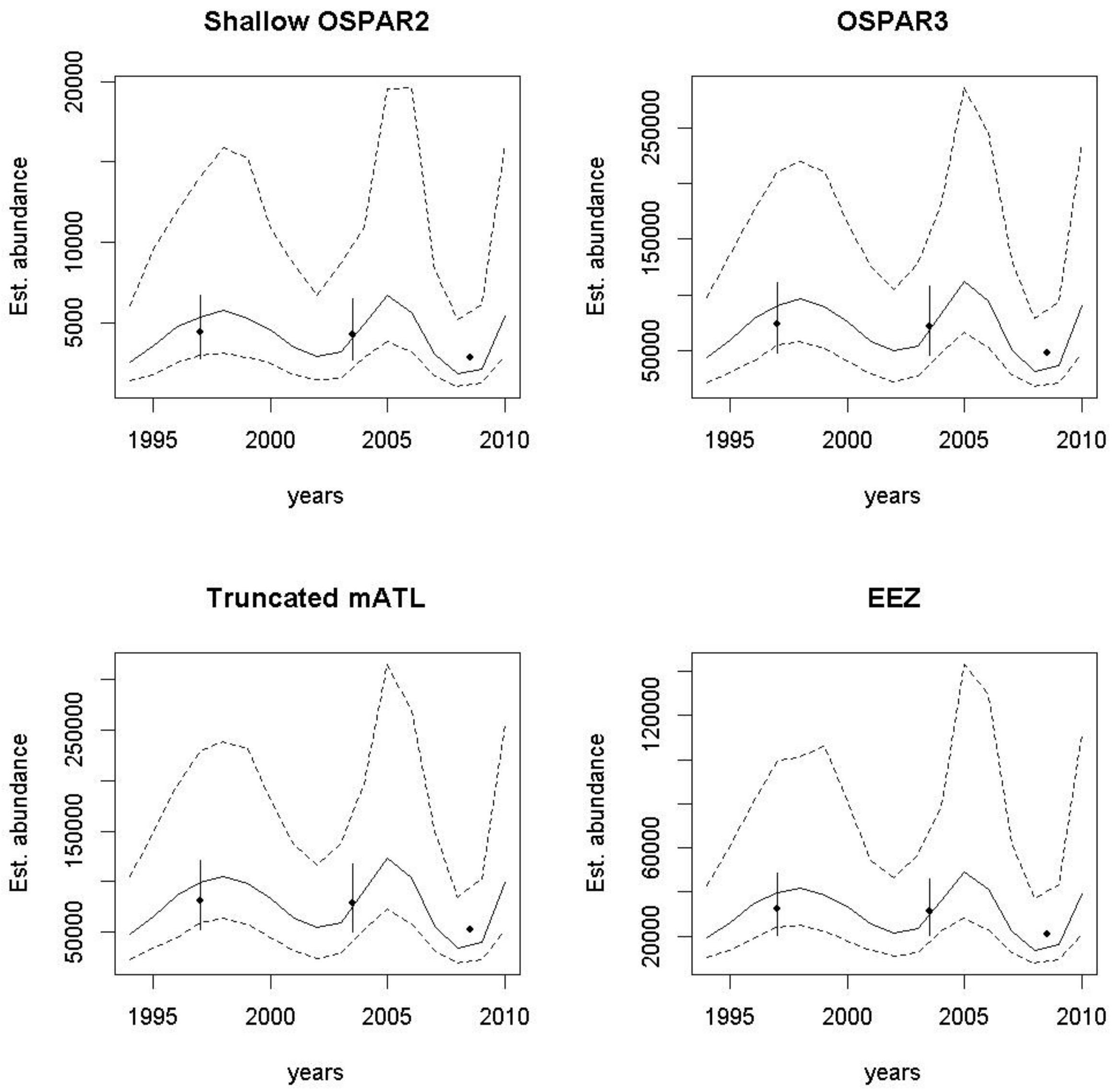

Figure 23. Short-beaked common dolphin predicted abundances (solid black line) and pointwise 95% CI (dashed black line) in four regions. Also shown are geometric mean abundances in each of three reporting period (dot at reporting period midpoints), and 95% pairwise confidence interval (vertical line), comparing reporting periods 1 and 2 with reporting period 3. If the confidence interval from an earlier reporting period does not overlap the mean abundance from reporting period 3, the difference is statistically significant at the 5% level, and is shown in red. For short-beaked common dolphin, there are no significant trends.

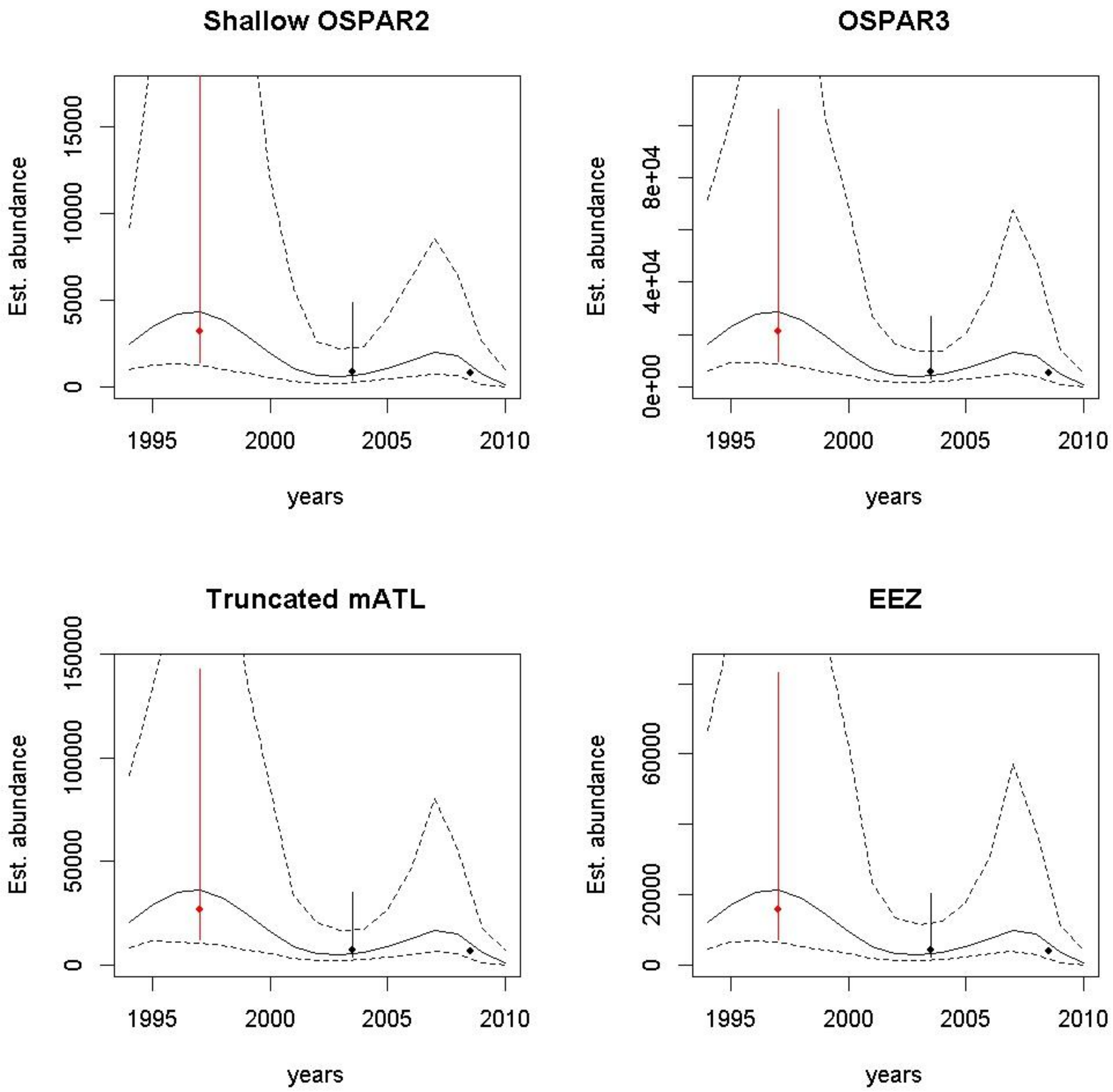

Figure 24. Risso's dolphin predicted abundances (solid black line) and pointwise 95% CI (dashed black line) in four regions. Also shown are geometric mean abundances in each of three reporting period (dot at reporting period midpoints), and 95% pairwise confidence interval (vertical line), comparing reporting periods 1 and 2 with reporting period 3. If the confidence interval from an earlier reporting period does not overlap the mean abundance from reporting period 3, the difference is statistically significant at the 5% level, and are shown in red. For Risso's dolphin, there is estimated to have been a significant decline between the first and third reporting period.

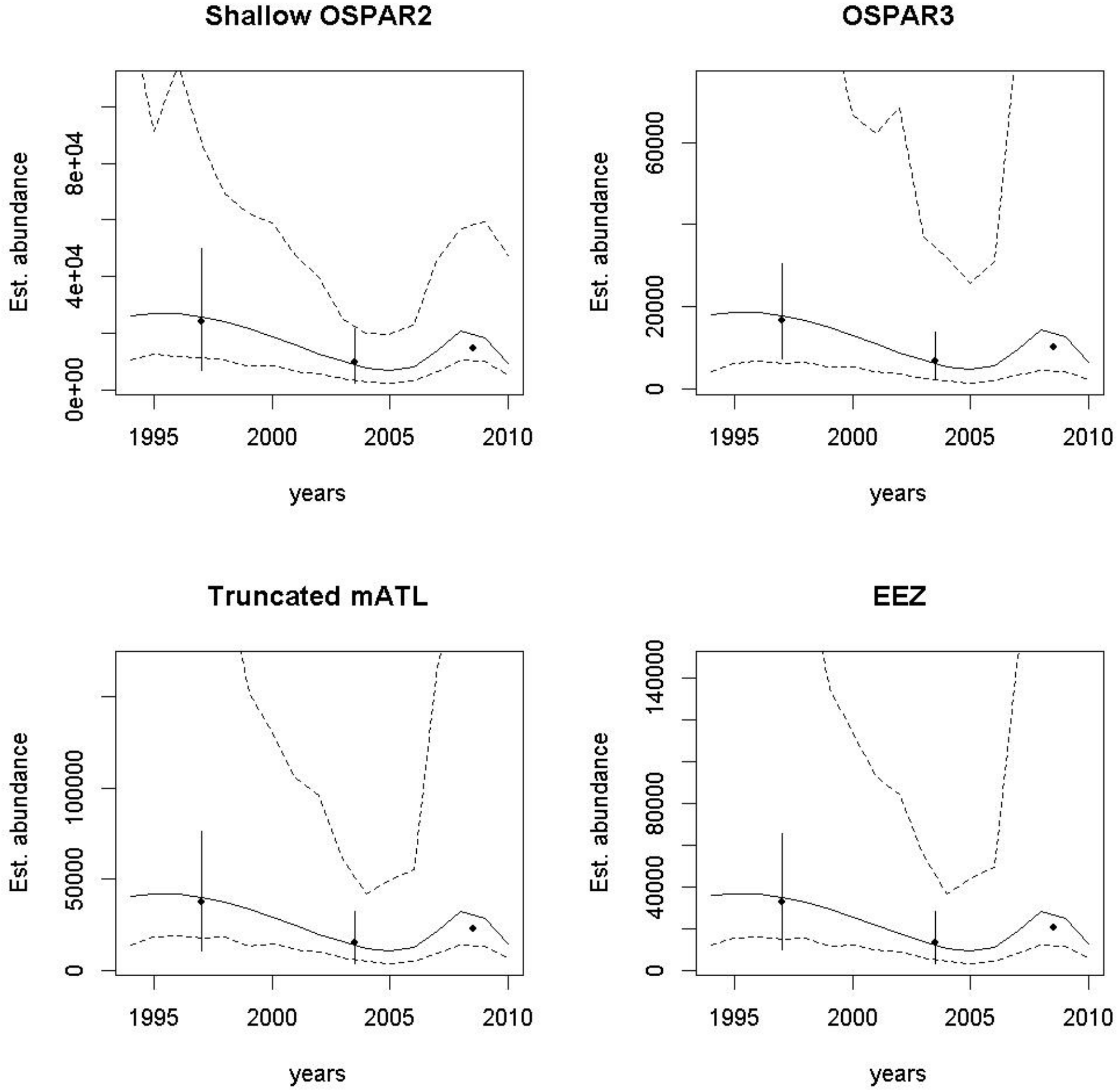

Figure 25. White-beaked dolphin predicted abundances (solid black line) and pointwise 95% CI (dashed black line) in four regions. Also shown are geometric mean abundances in each of three reporting period (dot at reporting period midpoints), and 95% pairwise confidence interval (vertical line), comparing reporting periods 1 and 2 with reporting period 3. If the confidence interval from an earlier reporting period does not overlap the mean abundance from reporting period 3, the difference is statistically significant at the 5% level, and are shown in red. For white-beaked dolphin, there are no significant trends.

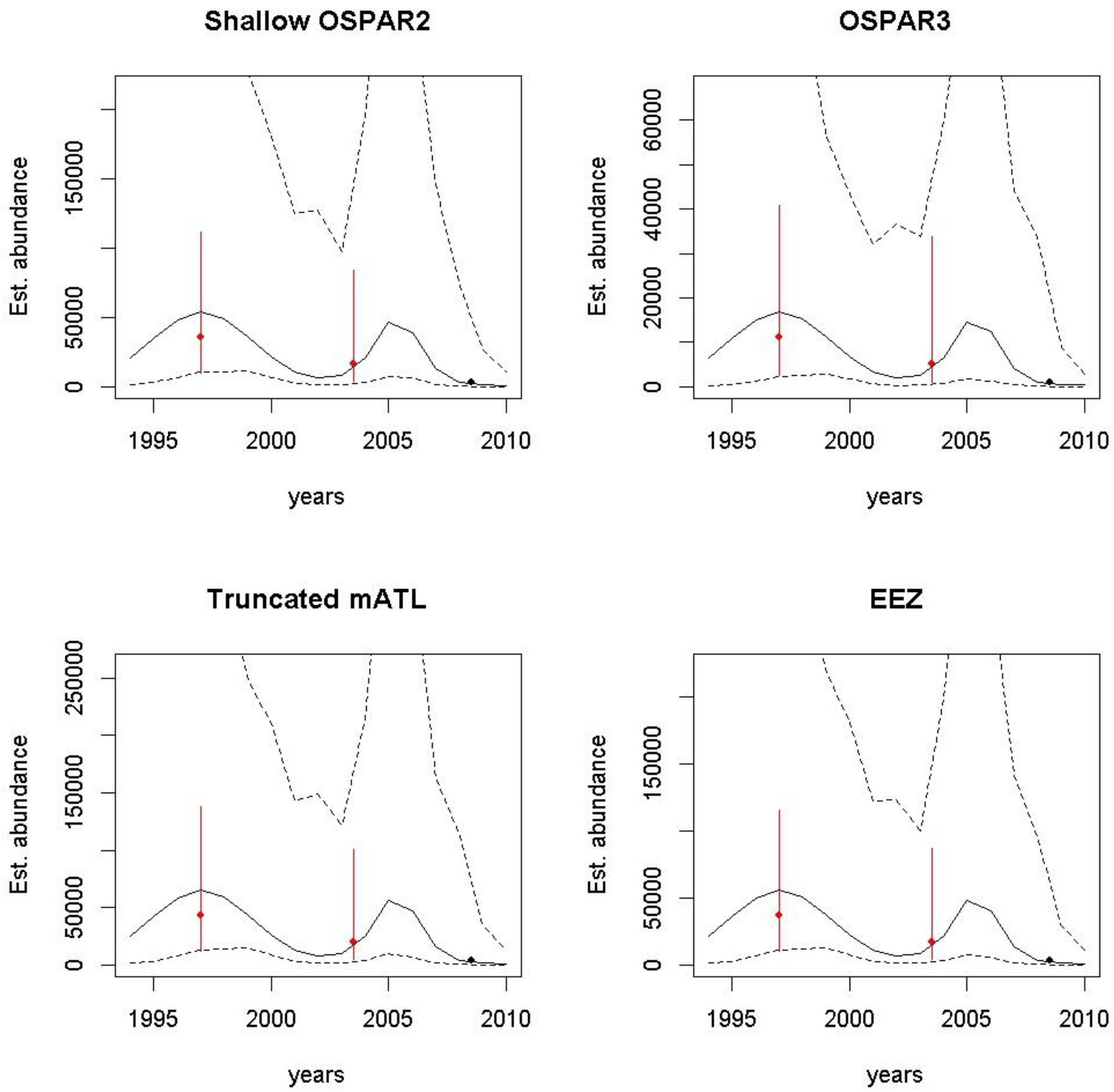

Figure 26. White-sided dolphin predicted abundances (solid black line) and pointwise 95% CI (dashed black line) in four regions. Also shown are geometric mean abundances in each of three reporting period (dot at reporting period midpoints), and 95% pairwise confidence interval (vertical line), comparing reporting periods 1 and 2 with reporting period 3. If the confidence interval from an earlier reporting period does not overlap the mean abundance from reporting period 3, the difference is statistically significant at the 5% level, and are shown in red. For white-sided dolphin, abundance is estimated to have declined between the first and third reporting period, and the second and third.

Table 17. Estimated trend (average annual population change) for seven species over multiple regions and time periods. Δ_1 is average annual change between reporting period 1 (1994-2000) and 3 (2007-2010); Δ_2 is average annual change between reporting period 2 (2001-2006) and 3. Values greater than 1.0 are an increase – e.g. 1.14 is an increase of 14% per year. Values in brackets are 95% confidence intervals; where these do not include 1.0, and hence there is a statistically significant trend at the 5% level, the entry is coloured red. For harbour porpoise there was variation in trend among region (because a spatio-temporal interaction model was selected) and so estimates are given for each of the 23 regions of interest; for the other species trends are almost identical in all regions (because only main effects models were selected) and hence only the trends for the larger areas are shown.

Harbour porpoise		
Region	Δ_1 (95% CI)	Δ_2 (95% CI)
Shallow OSPAR2	1.14 (0.97, 3.29)	1.11 (0.61, 13.03)
OSPAR3	1.04 (0.91, 1.19)	1.01 (0.74, 1.36)
Truncated mATL	1.09 (0.95, 1.41)	1.07 (0.79, 1.95)
Truncated EEZ	1.08 (0.94, 1.22)	1.08 (0.79, 1.43)
Moray Firth	1.32 (1.14, 1.51)	1.41 (0.98, 1.97)
Firth of Forth	1.14 (1.01, 1.31)	1.13 (0.82, 1.51)
Atlantic Array	1.07 (0.94, 1.21)	0.93 (0.68, 1.25)
Islay	1.20 (1.04, 1.37)	1.07 (0.77, 1.42)
SolwayFirth	1.12 (0.99, 1.28)	0.91 (0.66, 1.23)
North	1.14 (1.00, 1.29)	1.15 (0.82, 1.50)
East Orkney	1.14 (0.99, 1.29)	1.20 (0.86, 1.60)
Lewis	1.04 (0.91, 1.17)	1.08 (0.79, 1.44)
Kyle Rhea	0.97 (0.84, 1.11)	1.11 (0.81, 1.51)
Sound of Islay	1.14 (1.00, 1.29)	1.05 (0.78, 1.38)
Argyll Array	1.13 (0.98, 1.28)	1.10 (0.80, 1.46)
Hastings	1.31 (1.14, 1.53)	1.02 (0.73, 1.34)
IOW	1.13 (0.99, 1.29)	1.07 (0.76, 1.44)
Dogger Bank	1.24 (1.10, 1.44)	1.14 (0.82, 1.55)
South Dogger Bank	1.42 (1.23, 1.68)	1.32 (0.96, 1.79)
Norfolk Bank	1.99 (1.54, 2.58)	1.12 (0.77, 1.56)
Irish Sea	1.10 (0.97, 1.25)	0.89 (0.66, 1.19)
Strangford Lough	1.15 (1.01, 1.31)	0.92 (0.67, 1.25)
Shetlands	0.97 (0.84, 1.11)	1.07 (0.76, 1.45)
Minke whale		
Region	Δ_1 (95% CI)	Δ_2 (95% CI)
Shallow OSPAR2	0.91 (0.84, 0.99)	0.91 (0.81, 1.03)
OSPAR3	0.91 (0.84, 0.99)	0.91 (0.82, 1.03)
Truncated mATL	0.91 (0.84, 0.99)	0.91 (0.81, 1.03)
Truncated EEZ	0.91 (0.84, 0.99)	0.91 (0.82, 1.03)
Bottlenose dolphin		
Region	Δ_1 (95% CI)	Δ_2 (95% CI)
Shallow OSPAR2	1.05 (0.97, 1.14)	1.00 (0.90, 1.12)
OSPAR3	1.05 (0.96, 1.14)	1.01 (0.89, 1.13)
Truncated mATL	1.05 (0.97, 1.15)	1.01 (0.89, 1.13)
Truncated EEZ	1.05 (0.97, 1.15)	1.01 (0.89, 1.12)

Common dolphin		
Region	Δ_1 (95% CI)	Δ_2 (95% CI)
Shallow OSPAR2	0.96 (0.93, 1.00)	0.92 (0.85, 1.01)
OSPAR3	0.96 (0.93, 1.00)	0.92 (0.85, 1.01)
Truncated mATL	0.96 (0.93, 1.00)	0.92 (0.85, 1.01)
Truncated EEZ	0.96 (0.93, 1.00)	0.92 (0.85, 1.01)
Risso's dolphin		
Region	Δ_1 (95% CI)	Δ_2 (95% CI)
Shallow *OSPAR2*	0.88 (0.73, 0.95)	0.97 (0.69, 1.13)
OSPAR3	0.88 (0.77, 0.95)	0.97 (0.71, 1.12)
Truncated mATL	0.88 (0.76, 0.95)	0.97 (0.71, 1.12)
Truncated EEZ	0.88 (0.76, 0.95)	0.97 (0.71, 1.12)
White-beaked dolphin		
Region	Δ_1 (95% CI)	Δ_2 (95% CI)
Shallow *OSPAR2*	0.96 (0.90, 1.07)	1.09 (0.93, 1.45)
OSPAR3	0.96 (0.91, 1.03)	1.09 (0.94, 1.36)
Truncated mATL	0.96 (0.90, 1.07)	1.09 (0.94, 1.46)
Truncated EEZ	0.96 (0.90, 1.06)	1.09 (0.93, 1.44)
White-sided dolphin		
Region	Δ_1 (95% CI)	Δ_2 (95% CI)
Shallow *OSPAR2*	0.81 (0.73, 0.90)	0.71 (0.51, 0.95)
OSPAR3	0.81 (0.72, 0.92)	0.71 (0.49, 0.96)
Truncated mATL	0.81 (0.73, 0.90)	0.71 (0.51, 0.95)
Truncated EEZ	0.81 (0.73, 0.90)	0.71 (0.51, 0.94)

3.7 Power to Detect Trends

The estimated declines detectable with a power of 0.8 (assuming α-level of 0.05 and a two-tailed test) are given by species, time period and region in Table 18. For harbour porpoise, where a spatio-temporal interaction model was fit, values are given for all 23 regions of interest. For the other species, where only spatial and temporal main effects terms were fit (i.e. no interaction), the detectable population decline was very similar for all regions, and only the values for the four large regions are shown. Detectable declines are shown in two ways. First is the annual population change detectable with the specified level of power – for example, for minke whale in the region `Shallow OSPAR2', the detectable average annual change between time periods 1 and 3, Δ_1, is 0.90, equivalent to a decline of (1-0.90)*100=10% per year. Second is the total population change that would occur between the midpoint of time period 1 and the midpoint of time period 3 (i.e. 11.5 years) given the annual detectable change. In the above example this is 0.90^11.5=0.31. In other words, given that the detectable decline between reporting periods 1 and 3 for this species is 10% per year, this implies the population would need to decline to 31% of its original size between reporting periods 1 and 3 for the change to be detectable with a power of 0.8.

In general, the detectable annual population decline is closer to 1.0 (i.e. no change) for the comparison between reporting periods 1 and 3 than between reporting periods 2 and

3. This is because the CVs on population change are, in general, smaller for the first reporting period, as noted earlier. However, the total population change detectable did not show a consistent pattern: in some species a smaller total change was detectable over the longer time interval between reporting period 1 and 3 compared with the shorter time between reporting periods 2 and 3; in other species it was the other way round.

The detectable population decline was farthest from 1.0 for harbour porpoise, with detectable annual population changes of around 0.6 - 0.8 (i.e. 40 - 20% decline per year) being required in most cases. Detectable change was particularly far from 1.0 for the Shallow OSPAR2 region, but this was due to the great uncertainty associated with the estimates for the eastern North Sea. Putting these figures in the context of total population change between reporting periods, populations of harbour porpoise would have to decline to approximately 10% of their original size or less for this to be detectable with 80% power.

Of the other species, the species for which change was most likely to be detected was common dolphin, with detectable annual population changes of 0.94 (6% decline per year) between reporting periods 1 and 3 and approximately 0.87 (13% decline per year) between reporting periods 1 and 3. These both correspond with a total population change of approximately 0.5 – i.e. a halving of population size being detectable with 80% power. For most other species, total detectable change ranged between this value and the value for harbour porpoise. One exception was white-sided dolphin in reporting period 2, where detectable population change was extreme due to some outliers in the bootstrap re-samples causing the estimated CV to be high.

Table 18. Estimated coefficient of variation (CV) on average annual rate of change Δ, and detectable rate of change Δ (i.e. largest rate of population change below 1.0 that is detectable with a power of 0.8 and α-level of 0.05) for seven species over multiple regions and time periods. Δ_1 is average annual change between reporting period 1 (1994-2000) and 3 (2007-2010); Δ_2 is average annual change between reporting period 2 (2001-2006) and 3. Also shown is the detectable population change – i.e. the total population change implied by an average annual change of Δ over 11.5 years (Δ_1) and 5 years (Δ_2) respectively.

	Δ_1			Δ_2		
	Harbour porpoise					
Region	CV(Δ_1)	detectable(Δ_1)	detectable pop. change	CV(Δ_2)	detectable(Δ_2)	detectable pop. change
Shallow *OSPAR2*	0.71	0.17	0.00	7.12	0.01	0.00
OSPAR3	0.07	0.82	0.10	0.17	0.62	0.09
Truncated *mATL*	0.12	0.72	0.02	0.35	0.39	0.01
Truncated EEZ	0.07	0.82	0.10	0.16	0.64	0.11
Moray Firth	0.09	0.77	0.05	0.20	0.57	0.06
Firth of Forth	0.08	0.80	0.08	0.18	0.61	0.09
Atlantic Array	0.07	0.82	0.11	0.16	0.64	0.11
Islay	0.07	0.81	0.09	0.17	0.63	0.10
SolwayFirth	0.07	0.81	0.09	0.16	0.63	0.10
North	0.07	0.81	0.09	0.17	0.62	0.09
East Orkney	0.08	0.81	0.08	0.17	0.61	0.09
Lewis	0.07	0.82	0.10	0.16	0.63	0.10
Kyle Rhea	0.07	0.81	0.09	0.16	0.63	0.10
Sound of Islay	0.07	0.81	0.09	0.16	0.64	0.11
Argyll Array	0.07	0.81	0.10	0.16	0.64	0.10
Hastings	0.11	0.74	0.03	0.18	0.61	0.09
IOW	0.09	0.77	0.05	0.18	0.61	0.09
Dogger Bank	0.08	0.80	0.07	0.17	0.62	0.09
South Dogger Bank	0.11	0.74	0.03	0.18	0.61	0.09
Norfolk Bank	0.19	0.58	0.00	0.19	0.58	0.07
Irish Sea	0.07	0.81	0.09	0.17	0.63	0.10
Strangford Lough	0.07	0.81	0.09	0.16	0.63	0.10
Shetlands	0.07	0.81	0.09	0.18	0.61	0.08

	Minke whale					
Region	CV(Δ_1)	detectable(Δ_1)	detectable pop. change	CV(Δ_2)	detectable(Δ_2)	detectable pop. change
Shallow *OSPAR2*	0.04	0.90	0.31	0.06	0.85	0.43
OSPAR3	0.04	0.91	0.32	0.06	0.85	0.44
Truncated *mATL*	0.04	0.90	0.32	0.06	0.85	0.44
Truncated EEZ	0.04	0.90	0.32	0.06	0.85	0.44
	Bottlenose dolphin					
Region	CV(Δ_1)	detectable(Δ_1)	detectable pop. change	CV(Δ_2)	detectable(Δ_2)	detectable pop. change
Shallow OSPAR2	0.04	0.89	0.25	0.06	0.86	0.46
OSPAR3	0.04	0.89	0.25	0.06	0.84	0.42
Truncated mATL	0.04	0.89	0.26	0.06	0.85	0.44
Truncated EEZ	0.04	0.89	0.26	0.06	0.85	0.45
	Common dolphin					
Region	CV(Δ_1)	detectable(Δ_1)	detectable pop. change	CV(Δ_2)	detectable(Δ_2)	detectable pop. change
Shallow OSPAR2	0.02	0.94	0.52	0.05	0.88	0.51
OSPAR3	0.02	0.94	0.52	0.05	0.87	0.51
Truncated mATL	0.02	0.94	0.48	0.05	0.86	0.48
Truncated EEZ	0.02	0.94	0.50	0.05	0.87	0.49
	Risso’s dolphin					
Region	CV(Δ_1)	detectable(Δ_1)	detectable pop. change	CV(Δ_2)	detectable(Δ_2)	detectable pop.change
Shallow OSPAR2	0.07	0.82	0.11	0.12	0.71	0.18
OSPAR3	0.07	0.83	0.12	0.12	0.71	0.18
Truncated mATL	0.07	0.83	0.12	0.12	0.71	0.18
Truncated EEZ	0.07	0.83	0.12	0.12	0.71	0.18

	White-beaked dolphin					
Region	CV(Δ_1)	detectable(Δ_1)	detectable pop. change	CV(Δ_2)	detectable(Δ_2)	detectable pop. change
Shallow OSPAR2	0.04	0.88	0.24	0.14	0.68	0.15
OSPAR3	0.03	0.91	0.34	0.10	0.75	0.24
Truncated mATL	0.05	0.88	0.23	0.14	0.68	0.15
Truncated EEZ	0.04	0.88	0.24	0.14	0.68	0.15
	White-sided dolphin					
Region	CV(Δ_1)	detectable(Δ_1)	detectable pop. change	CV(Δ_2)	detectable(Δ_2)	detectable pop. change
Shallow OSPAR2	0.06	0.85	0.15	0.72	0.16	0.00
OSPAR3	0.06	0.84	0.13	0.72	0.16	0.00
Truncated mATL	0.06	0.85	0.15	0.70	0.17	0.00
Truncated EEZ	0.06	0.85	0.15	0.64	0.19	0.00

4 Discussion

4.1 Effort Coverage

The spatio-temporal analysis is more comprehensive than has previously been attempted for this area, although for example Paxton & Thomas (2010), Paxton *et al* (2012), Baines & Evans (2009, 2012), Hammond (2006, SCANS-II), Reid *et al* (2003), Evans & Wang (2003), Hammond *et al* (2002, SCANS), Weir *et al* (2001) and Northridge *et al* (1995a,b) covered large fractions of the region considered here. Also there have been a variety of analyses of smaller areas (e.g. Weir *et al* 2009, Macleod *et al* 2004). Many of those data used in these previous analyses have been incorporated here.

Effort coverage varies considerably (Figure 8 – 10) both spatially and temporally with much more effort since 2000. There has also been a shift in the distribution of effort, with much more survey effort concentrated in regions of developer interest (cf. Figure 8a,b,c vs. Figure 8d).

Winter (Jan. - Mar.) and autumn (Oct. - Dec.) have meagre coverage away from the ferry routes compared to summer, compromising the ability to make accurate and precise estimates of abundance for this time period. There is relatively little effort in the northern North Sea and the Celtic Sea, and there were no data available for some small regions in the eastern English Channel and to the south of the Hebrides (Figure 8a). It may be that with collaboration with more international partners better coverage of the eastern North Sea can be achieved.

As in previous JCP data resource studies (Paxton *et al* 2012, Paxton & Thomas 2010), only survey data or systematically collected data were used from the available data. There are data defined as `casual watches’, etc., in the available datasets, and data from boats where the height of the observer above sea level was not known or not recorded. Some of this type of data may be usable if more information is found and effort is clearly delineated. Nevertheless, there remains an important distinction in the surveys used: some are formal surveys designed for cetacean abundance estimation, while others are opportunistic placement of observers on board available vessels. It is possible that platform of opportunity data based solely in areas of high cetacean abundance could bias the results, although the models should be quite robust against this if nearby collected effort was unbiased in terms of expected cetacean density.

4.2 Adjustments for Detectability

The detection functions considered the influence of covariates on the scale of the function, not the shape. Other variables that could be considered in the future (albeit without the possibility of comparison with all previous surveys) could include glare, visibility, observer ID and experience (with experience, observers can increase their probability of detecting harbour porpoise, Laake *et al* 1997).

The estimates of perception bias were, unsurprisingly because of the data overlap, comparable with the published figures for the individual surveys. For example, $g(0)$ estimates from SCANS-II boat surveys (Hammond *et al* in press) were comparable to the figures found here (e.g. harbour porpoise: JCP 0.31 (CI: 0.25 – 0.36) SCANS-II 0.22 (0.16 – 0.28), minke whale JCP 0.42 (CI: 0.17 – 0.66), SCANS-II 0.55 (CI: 0.23 - 0.86), dolphins: JCP 0.56 (CI: 0.49 – 0.63), SCANS-II (common dolphin only): 0.57 (CI: 0.40 - 0.74)).

The estimates of availability bias used were based on values taken from the literature and expert opinion. The possible effect of mis-specifying availability is discussed further in section 4.4.

A fundamental limitation of the JCP data resource is that the data come from a very wide variety of surveys, some with different field protocols. In adjusting for detection differences over space and time, we have assumed that differences can be accounted for by modelling detection functions using observed distances and other measured covariates such as ship type, and by taking general account of perception and availability bias. If field methods differ in ways not related to the covariates used, or if they have evolved over time (e.g. through better observer training, or more rigorous and standardized enforcement of observation protocols) then this will cause patterns in the observations that are not related to changes in animal density, and are not accounted for by our analyses.

4.3 Density Surface Modelling

The density surface or abundance modelling method undertaken here was different to that of Paxton & Thomas (2010), but similar to Paxton *et al* (2012) where leakage in predictions around islands and headlands was controlled using the methods developed in Scott-Hayward *et al* (2011). Temporal correlations in the model residuals were accommodated using generalized estimating equations (GEEs). Whilst predictors of biological interest (*Depth, Slope, SST*) did occur in the final models and these environmental covariates entered the model first, the final model always contained these predictors along with a 2D spatial smooth and so no effort has been made to interpret the models biologically. Model selection without 2D spatial smooths would allow more useful models in terms of explaining the biology of the animals. Similarly, no effort has been made to investigate evidence of interspecies interactions from the JCP data resource in cases where such interactions have been hypothesised (e.g. white-beaked dolphins vs. short-beaked common dolphin, Weir *et al* 2009)

4.4 Estimates of Abundance

The models generally produced realistic estimates of density and hence abundance. Occasionally regions of high density were predicted in areas where there was little effort (for example, the predicted high density region of bottlenose dolphin on the north-west coast of Ireland in 2010, Figure 15). The point estimates of abundance are similar to those from previous studies where comparisons can be made (see individual species accounts). Occasionally, the areas of highest predicted density are unreasonably high (e.g. short-beaked common dolphins off the south-west coast of Ireland in the autumn of 2010). Almost always these ‘hotspots’ are in places and times of low effort where high densities were actually detected with very little survey effort, or where there is no effort and some extrapolation has taken place. There are possible biases in the data too. For example short-beaked common dolphins and white-beaked dolphins are attracted to boats (Hammond *et al in press.,* Evans pers. comm., Würsig 2009, Cañadas *et al* 2004, Palka & Hammond 2001). From smaller boats, animals can be first seen only after they have already approached the vessel, distorting the detection function leading to a lower estimate of the probability of detection (Evans pers. comm.). It would be possible to correct for this only for surveys that were working in trial mode with trackers observing some distance away.

Over- or under-estimation of availability will also bias predicted densities. The availability calculation requires a number of inputs: diving and surfacing times and the “window of opportunity" i.e. the time the animals are exposed to the observer related to the forward observing distance and the speed of the vessel/plane. Uncertainty in the dive times is considered in the calculation but uncertainty in the surfacing times for dolphins was unknown so the results may be sensitive to the choice of surfacing time. If the window of opportunity is underestimated (i.e. too small a distance is considered) then this will lower the estimated availability, inflating the corrected estimates of density. Likewise the window of opportunity can be overestimated. Without formal experiments, estimation of this critical window is subject to error. We asked some data suppliers for their estimates of how far forward they are searching, but perceived distances and actual searched distances may be different. Animals in a pod were assumed to surface independently of one another (i.e. with an asynchronous diving pattern). Therefore the availability of large pods could be overestimated, since synchronously-diving animals provide fewer opportunities for the pod to be observed.

We found some seasonal patterns in abundance. Species may be undergoing seasonal migrations; however it is also possible that such a seasonal difference may be an artefact of unmodelled seasonal variation in detectability and/or availability, leading to lower detectability in winter. The data were not sufficient for us to investigate seasonally-varying detection probability.

4.5 Uncertainty in the Estimates

There was considerable uncertainty in some of the geo-referenced estimates (see the surfaces in Appendix 4). Even reasonable point estimates of density generated unreasonably high upper bound bootstrap estimates of density (e.g. harbour porpoise in the summer 2010,Figure 11), with densities in excess of 10000 animals/km^2 in some regions. Variance in the quasi-Poisson models used was proportional to the mean, so all other things being equal, large point estimates will necessarily lead to wide confidence intervals. Extrapolation into regions of low or zero effort also lead to great uncertainty in the estimates.

Despite the wide confidence intervals, some aspects of the uncertainty in the analysis have not been incorporated into the final estimates. There is no model selection uncertainty (Burnham and Anderson 2002) at either the detection function fitting or spatial modelling stage. Future work could incorporate this, leading to wider confidence intervals.

4.6 Appropriate Resolutions for Inference

For all species except harbour porpoise, the selected models did not allow for changes in spatial distribution over time (except for the potential effect of long-term changes in the spatial pattern of sea surface temperature). This has two consequences. First, inferences about spatial pattern are essentially averages over the entire time period (1994-2010), and therefore may not be accurate for any particular time period within this interval. For example, there may be particular interest in the most recent spatial distribution of animals (i.e. for 2010). The models fitted here cannot be expected to produce unbiased estimates of this distribution if there have been significant changes during the time interval analysed. Second, inferences about temporal trends are essentially averages over the entire spatial region, and may well not be accurate for any particular sub-region. The population changes reported here give a large-scale context but, for example, cannot be assumed to hold for any of the 19 smaller regions of conservation or developer interest.

Even with these caveats, it is germane to ask what spatial scale the time-averaged estimates may be reliably interpreted on, and what temporal scale the spatially-averaged estimates may be used. The modelling methods used here are based on spatial and temporal smoothing of the adjusted counts. Hence the estimates will only be reliable over larger areas of space and time. One possible approach for addressing this would be to perform a detailed examination of the residuals from the density surface models to determine at what scales they are, on average, zero. For example, at small spatial scales we expect local fluctuations in density (e.g. localized hotspots) to be smoothed over, and hence we expect to see systematic over- or under-prediction and non-zero averaged residuals. At larger spatial scales, these fluctuations will average out. We have undertaken a preliminary analysis along these lines for 2 species, Harbour porpoise and Risso's dolphin, to investigate the relationship between averaged residuals and prediction area. Results, given in Appendix 5, indicate that predictions over scales in the order of 500-1000 km^2, give average residuals close to zero (so unbiased estimates), and average absolute residuals in the order of 60-40% of the predicted abundance for Harbour porpoise and 100-60% for Risso's dolphin. At smaller scales, estimates can be biased and absolute residuals relatively large – i.e., inferences unreliable. We therefore suggest that 500-1000 km^2 is the minimum area for which predictions may be useful. Further, these analyses were only possible in areas where there were data (since they are based on data residuals); inferences will be less reliable in areas far from data values. We did not perform a similar analysis to address the question of what time scale the spatially-averaged residuals may be reliably interpreted on, but our judgement is that inferences on the temporal scale of a decade or greater may be appropriate.

For harbour porpoise, a spatio-temporal interaction model was selected for inference. Hence, unlike the other species, it is feasible that local estimated trends may be accurate, as may be spatial patterns for subsets of the time series (such as more recently). We suggest that appropriate scales for spatio-temporal inference are again around 500-1000 km^2 and a decade, but unlike for the other species both subsets may be taken at once.

One further use of the outputs of this analysis is the identification of regions of persistent predicted high density (especially if models with a spatio-temporal interaction are used) which may be of use in the designation of regions of conservation interest. How this relates to the resolution issue is currently unclear.

4.7 Species-level Estimates of Density and Abundance

4.7.1 Harbour porpoise

The patterns found here are similar to those of Baines & Evans (2009, 2012), Paxton & Thomas (2010) and Paxton *et al* (2011): the high density region of the Hebrides are still present, but the relative importance of the high density band between south Wales and Dublin has decreased. In addition the new analysis reveals a large area of harbour porpoise high density off the coast of East Anglia, although density patterns fluctuate through time with different areas utilised for a few years at a time. This is in agreement with the SCANS and SCANS-II synoptic surveys, which showed a shift in the core North Sea area between 1994 and 2005 from north to south (Hammond *et al* in press, Hammond *et al* 2002). Similarly, other studies considered together have shown that harbour porpoise numbers fluctuate in the southern North Sea (e.g. Thomsen *et al* 2006, Camphuysen 2004). However, the current analysis suggests that there is no consistent directional trend in harbour porpoise densities in the North Sea at least over the course of the period 1994 - 2010. Rather, densities increase off the coast of East Anglia and then the modal density area moves to the north. This may, however, reflect increased sampling effort in the Dogger Bank, which, prior to the mid-2000s, was rarely surveyed. Prior to 1994, there is evidence that harbour porpoise were generally further north in the North Sea (Northridge *et al* 1995, Evans 1990) from surveys, with some more equivocal evidence from stranding data that harbour porpoise were fewer in number in the southern North Sea coast in the 1970-1980s (e.g. Haelters *et al* 2011). The current paucity of eastern North Sea/Skagerrak data in the data resource meant there was considerable uncertainty in the eastern North Sea estimates (Figure 11) so no conclusions about densities there can really be drawn.

Estimated numbers from the SCANS survey region of 1994 (not including the Baltic blocks) using the JCP model were of 653100 (110800 – 5369700 confidence intervals influenced by the uncertainty in Danish waters), compared to 329200 (166400 – 651200) calculated from the actual SCANS paper (i.e. Table 4 in Hammond *et al* 2002). A direct comparison with SCANS-II cannot be given as two blocks partially lie outside the JCP Phase 3 area, but the nearest estimate from the JCP (the whole region) for summer 2005 is 501900 (CI: 53600 – 6678100 confidence intervals influenced by the uncertainty in Danish waters) compared to a SCANS-II estimate of 375400 (CI: 256300 – 549700) (from the design based estimates of SCANS-II, Hammond *et al* in press).

4.7.2 Minke whale

The current analysis and available $\widehat{D}$ suggests that minke whale densities fluctuate across the north-western European shelf although regions with persistent high density include the south-west coast of Ireland, Isle of Man, the Hebrides and off the coast of north-west England and western Scotland, although the centre of this last region moves considerably. This is mostly similar distribution of densities to that described for the period 1979 – 1995 by Northridge *et al* (1995) and by Reid *et al* (2003).

The fluctuations in the North Sea were not wholly captured by the chosen model. Models with interactions that captured the spatio-temporal patterns in density more effectively led to implausibly wide confidence intervals in the bootstrap, and so there was a trade off in terms of model choice between accuracy and precision. The predicted number of animals from SCANS was 8400 (3600 – 60600) (from Table 6 in Hammond *et al* 2002) and by closest approximation in SCANS-II (Hammond *et al* in press), 18960 (9800 – 36680). The nearest equivalent JCP figures are 33900 (CI: 16360 – 115110) and 19950 (12690 – 58950).

Minke whales are primarily summer visitors to the north-west European continental shelf, with numbers increasing dramatically in the middle months of the year (Table 11). This is in agreement with analyses that have been made of components of the JCP data resource (e.g. Irish Sea: Baines and Evans 2009, 2012; Inner Hebrides: Macleod *et al* 2004; North-western Approaches: Weir *et al* 2001). Minke whales do persist on the shelf in winter but at low numbers so the existence of an all year around resident population of shelf minke whales cannot be refuted.

4.7.3 Bottlenose dolphin

Unsurprisingly, the models predicted geographically discrete coastal populations of this species (e.g. Figures 14 & 15). Density was estimated to be higher in spring and summer (Table 12). If numbers do fluctuate seasonally, it is not wholly clear where the animals go. Stockin *et al* (2006) reported Moray Firth dolphins off the coast of Aberdeen but such minor movements would not cause a reduction in abundance across the whole shelf. The models did predict a persistent population of dolphins off Aberdeen as detected by long term primarily shore-based observations from the east Aberdeenshire coast (Anderwald *et al* 2010)

The locations of higher density agree mostly with the incoming data, with the exception of a region off the north of Ireland (Figure 14). The fluctuations in densities were not wholly captured by the chosen model. Models with interactions that captured the density more effectively led to implausibly wide confidence intervals in the bootstrap. So once more there was a trade off in terms of model choice between accuracy and precision.

The conventional SCANS-II estimate for bottlenose dolphin abundance has substantial numbers predicted for a block (P) that is only partially covered by the JCP therefore it cannot really be compared to a result here. SCANS contained no estimates of abundance for *Tursiops truncatus*.

4.7.4 Short-beaked common dolphin

This species primarily occurs on the south-western side of the British Isles and Celtic Sea, with non-negligible predicted densities in the northwest approaches (Figure 16) as found by other studies (e.g. Baines and Evans 2009, 2012). Again there was an accuracy/precision trade off. The conventional SCANS-II estimate for common dolphin abundance has substantial numbers predicted for a block (P) that is only partially covered by the JCP therefore it cannot really be compared to the result here.

4.7.5 Risso’s dolphin

This species had the lowest estimated density of all those considered here (Section 3.5.6), although local relative abundance can be high off the west coast of Ireland, the northern Irish Sea and the Hebrides. Similar results for the Irish Sea were found by Baines & Evans (2009) and by Weir *et al* (2001) for the Hebrides. Predicted numbers are given in Table 14. No other figures for the entire JCP area are available by way of comparison. N.B. In revision, it was discovered that not all sightings of this species had been supplied by WDCS (de Boer pers. comm.) hence our estimates of the abundance of this species in the English Channel will be an underestimate.

4.7.6 White-beaked dolphin

This species mainly occurs in the northwest Approaches and the central North Sea (Figure 17, Figures A4.32 – A4.36). Similar patterns were observed by Northridge *et al* (1995) and Weir *et al* (2001) from ESAS data collected from 1979 – 1991 and 1979 – 1998 respectively (so only partially overlapping with the data included here). Reid *et al* (2003) reach much the same conclusion. There was negligible evidence of white-beaked dolphin presence in the southernmost North Sea although strandings occurred in most years on the coast of the Netherlands and Belgium up until at least 2005 (van der Meij *et al* 2006, Kinze *et al* 1997). Predicted numbers peaked in the prediction area in spring, in contrast to Canning *et al*’s (2008) conclusions based on data from the north-east of Scotland, where strandings peaked in August.

The predicted number of animals from SCANS was 7900 (CI: 3000 – 27600) (Hammond *et al* 2002) and by closest approximation from SCANS-II (Hammond *et al* in press), 16500 (CI: 9200 – 29600). The nearest equivalent JCP figures are 28770 (CI: 11730 – 161730) and 10540 (CI: 3960 – 49530900) respectively. However, SCANS estimated a further 11800 *Lagenorhynchus* sp. which would presumably mostly be *L. albirostris*. N.B. In revision, it was discovered that not all sightings of this species had been supplied by WDCS (de Boer pers. comm.) hence our estimates of the abundance of this species in the English Channel will be an underestimate.

4.7.7 Atlantic white-sided dolphin

As far as we are aware, the estimates as part of the various iterations of the JCP data resource analyses represent the first time the abundance of this specific species (as opposed to *Lagenorhynchus* as a genus) has been estimated across the European continental shelf although Macleod (2004) provides estimates for the shelf edge. The available data suggests the species is a fairly consistent member of the cetacean fauna of the shelf albeit at very low abundance on the shelf edge (Figure 18, Figures A4.37 – A4.42). This is in agreement with the observations of Reid *et al* (2003), Weir *et al* (2001) and Northridge *et al* (1995a,b). Presumably as a shelf edge species, only some of its range was covered in the prediction region considered here. N.B. In revision it was discovered that not all sightings of this species had been supplied by WDCS (de Boer pers. comm.) hence our estimates of the abundance of this species in the English Channel will be an underestimate.

4.8 Trend Estimation and Power Analysis

The trend estimates and subsequent power analysis well illustrate the trade-off between model complexity and precision. For harbour porpoise, the data were able to support a more complex and realistic model with spatio-temporal interactions, allowing temporal trend to vary over space; however this resulted in relatively high uncertainty in these predictions and therefore low power. It was estimated that populations would need to decline to 10% or less of their starting value between reporting periods for this to be detectable with an assumed power of 0.8. For the other species, spatio-temporal interaction models were not chosen during model selection, and precision of the trend estimates was generally better. A spatially- and temporally-varying covariate, sea surface temperature, was included in some models, but this did not cause the trend estimates to vary greatly over space. This implies that long-term trends in sea surface temperature are similar over the study area, at least at the time of year when the predictions were made for trend estimation (mid-summer). All other species, therefore, had effectively a single trend estimate over the whole area. While this does not then admit regional trend estimation (see below), the precision in some cases was quite good. For common dolphins, the best example, a halving of the population between reporting periods could be detected with power of 0.8. For the other species, a percentage remaining of between 50% and 10% was estimated to be detectable (except for one reporting period with white-sided dolphin where precision was extremely poor). In approximate descending order of precision, averaging the results across the two reporting periods and rounding to the nearest 5%, the detectable population remaining was approximately 50% for common dolphin, 35% for minke whale, 35% for bottlenose dolphin, 20% for white-beaked dolphin, 15% for Risso’s dolphin, and 5% for white-sided dolphin.

In the power analysis undertaken here, the focus was on the per year population rate of change that would be detectable with a given level of power (0.8) and α-level (0.05). The problem may be formulated in other ways: for example, one may ask: “”What is the power to detect a given rate of population change that is considered to be biologically significant?” As an illustration, the time between reporting periods has historically been seven years; it may be decided that a suitable target is to be able to detect a population halving or doubling over this time frame, which corresponds to an annual rate of change of 0.91 for the decline or 1.10 for the increase. For a fixed α-level and rate of population change (the “effect size”, Δ), power depends on the coefficient of variation in population change (CV(Δ)), as shown in Table 19. Taking minke whales as an example, where $CV(\Delta_2)$ was 0.06, the power would be 0.37.

Table 19. Power to detect a halving or doubling of population size between two reporting periods for a given coefficient of variation on the estimated average annual population change (CV(Δ)), assuming an α-level of 0.05 and a two-tailed z-test of the null hypothesis that population change is 1.0 (i.e. no change).

CV(Δ)	Power
0.01	1.00
0.02	1.00
0.03	0.91
0.04	0.70
0.05	0.51
0.06	0.37
0.07	0.29
0.08	0.24
0.09	0.19
0.10	0.16

Throughout this document, for hypothesis tests and power analysis we have assumed an α-level of 0.05 is used in determining statistical significance. Given that we have also used a target power of 0.8, it may be argued that an α-level of 0.2 would be more appropriate, thereby balancing the frequency of type I and II errors. Using a higher α-level would make smaller population changes detectable.

In determining the coefficient of variation to include in a power analysis for trends, two types of uncertainty can be incorporated: uncertainty due to not knowing the true population size at each time point (observation error) and uncertainty due to the population size itself being a random variable, the outcome of random processes of environment and demography (process error). Observation error should always be included in a power analysis, but whether process error should be depends on the inferences that are to be drawn (Thomas *et al* 2004). If process error is not included, the inferences are about the power to detect trends in the time period and location analysed, assuming the same environmental and demographic process would repeat themselves if the surveying and analysis were repeated. If process error is included, the inferences are more general, and can be extended to areas and times that are like that analysed in that they have similar random environmental and demographic processes – as in, for example, power to detect future declines. In the analyses performed here, only observation error was accounted for, by using a bootstrap procedure to generate new datasets with similar levels of observation error; hence inferences on power should refer only to the area analysed and the period 1994-2010. Predicting power to detect future population changes requires incorporation of the additional process error uncertainty. See Thomas *et al* (2004) for further discussion of this issue.

For species with effectively a single trend estimated for the whole area, it is possible that there is biologically significant spatial variation in trend, and this will not be captured in the model. Hence producing trend estimates for the smaller areas of developer and conservation interest for these species from this analysis is not tenable.

Some statistically significant trends were detected. Harbour porpoise were estimated to have increased between the 1st and 3rd reporting periods (i.e. 1994-2000 vs 2007-2010) in 6 regions: Sound of Islay, Hastings, Dogger and South Dogger Banks, Norfolk Bank and Strangford Lough. Given that harbour porpoise is a widely distributed and highly mobile species, is is not clear whether these changes have any biological significance. Minke whales, Risso's dolphin and white-sided dolphin were estimated to have declined on average over the whole area during the same period, and this decline was also statistically significant with white-sided dolphins for the comparison of 2nd and 3rd reporting periods. The latter two results are based on very few positive sightings, but nonetheless all three species deserve further investigation.

4.9 Conclusions and Future Work

The Joint Cetacean Protocol data resource is a very large and rich dataset. We analysed data from 38 data sources, consisting in the current analysis of approximately 1.09 million kilometres of effort over 17 years. However, this effort was very uneven, with large areas of the study region receiving little or no effort, particularly in some seasons and years (Figure 8). Further, almost all data sources came from restricted regions of space or time, in many cases with little overlap between sources.

Our first task was to convert the diverse count data into estimates of density on the surveyed tracklines, by accounting for detectability. This was largely successful, although we have concerns about the corrections made for availability bias, which were based on little information. We also face a fundamental limitation of the JCP data in that we have to assume observations of the same species coming from the same vessel types and under similar survey conditions have the same detection probability, while in practice this may differ between data sources, and even within data sources if field protocols or practice have changed substantially over time. There is not enough spatio-temporal overlap between data sources to use differences in encounter rates between sources to correct for any differences – this must instead be done on the basis of detectability modelling.

Our next task was to fit density surface models. In order to allow for changes in distribution over time we included in the candidate model set models with spatio-temporal interactions – i.e. allowing for changes in spatial distribution over time. These models fitted the observed data (estimated density per survey segment) moderately well, and produced spatial patterns that were largely realistic. However, in all cases except harbour porpoise, the resulting uncertainty in predictions was untenable – illustrating the well-known trade-off between accuracy and precision. For all species except harbour porpoise, our final models did not include the interaction term, meaning that some changes apparent in the data were not represented in our estimates (e.g. minke whale concentrations in the North Sea).

The effort that went into this analysis was considerable. One important question, therefore, is how often it should be repeated. If the statistical methods were fixed, then given that these methods are based on large-scale spatial and temporal smooths, there seems little benefit from repeating the analysis more frequently than once per reporting period (i.e. approximately every 6-7 years). However, several factors mean it may be worthwhile to undertake analysis effort more frequently. First, the analysis methods are not fixed: statistical research in the area of spatio-temporal modelling is rapidly developing. Second, more good-quality data are potentially available from within the time period and region analysed. Incorporation of this, especially dedicated line-transect surveys, would be valuable (see Section 4.1 for a description of the principal gaps in coverage in the current analysis). This is particularly true for times or places not well covered at present. Third, it may be possible to include data collected from other types of survey such as passive acoustic surveys, etc. Fourth, improved methods for dealing with availability bias are under active development; it would also be very helpful if more data on both perception and availability bias were used in analyses such as those we performed here. For all of these reasons, an on-going analysis effort may be profitable.

Our planned scope of inference was larger both in space and time than our final scope. It proved infeasible to make estimates prior to 1994, due to questions about the comparability of the data, and for areas of water deeper than 300m due to lack of information. If it is desired to estimate spatial or temporal patterns for, e.g. the entire UK

EEZ, then more data from deeper waters will be required. Also, given that one of the major drivers for the JCP is EU Habitats Directive reporting requirements, under which all European countries have similar needs for information, then it may be that pooling information across countries will lead to stronger inferences for all.

Given the diverse nature of the input data, and the patchiness of the spatio-temporal coverage, it is clear that inferences from the Joint Cetacean Protocol data resource will always rely strongly on modelling. Such results are vulnerable to failure of model assumptions, model mis-specification, and other issues. Therefore, for robust inferences about population abundance and trend, it seems prudent that the platform-of-opportunity data component of the JCP be complemented by periodic, large-scale designed surveys that are designed to produce reliable snapshots of abundance at the desired spatial scales.

5 References

Anderwald, P., Evans, P. G. H., Canning, C., Hepworth, K., Innes, M., Macdonald, P., Sim, I., Stockin, K. & Weir, C. 2010. *Cetaceans of the East Grampian Region*. Sea Watch Foundation, Aberdeen. 68pp.

Anderwald, P., Evans, P. G. H., Dyer, R,. Dale, A., Wright, P. J. & Hoelzel, A. R. 2012. Spatial scale and Environmental Determinants in Minke Whale Habitat Use and Foraging. *Marine Ecology Progress Series* **450,** 259 – 274.

Baines, M. E. & Evans, P. G. H. 2009. *Atlas of the Marine Mammals of Wales.* CCW Marine Monitoring Report **No. 68.** 84 pp.

Baines, M. E. & Evans, P. G. H. 2012 *Atlas of the Marine Mammals of Wales.* 2nd Edition. CCW Marine Monitoring Report **No. 68.** 143pp.

Barlow, J. 1988. Harbor porpoise (*Phocoena phocoena*) Abundance Estimation in California, Oregon and Washington: I. Ship Surveys. *Fishery Bulletin* **86,** 417 – 432.

Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L., Borchers, D. L. & Thomas, L. 2001. *Introduction to Distance Sampling: Estimating Abundance of Biological Populations*. Oxford University Press, Oxford. 432 pp.

Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L., Borchers, D. L. & Thomas L. (editors) 2004. *Advanced Distance Sampling.* Oxford University Press, Oxford. 434 pp.

Buckland, S. T. & Turnock, B. J. 1992. A Robust Line Transect Method. *Biometrics* **54,** 1221 – 1237.

Burnham, K. P. & Anderson, D. R. 2002. *Model Selection and Multi-Model Inference: A Practical Information-Theoretic Approach.* Springer, London. 488pp.

Burt, M. L., Borchers D. L., Jenkins, K. & Rexstad, E. A. (in prep.) MRDS: Using Mark-Recapture Distance Sampling in Line Transect Surveys.

Camphuysen, K. C. J. 2004. The return of the harbour porpoise (*Phocoena phocoena*) in Dutch coastal waters. *Lutra* **47**, 113–122.

Cañadas, A., Desportes, G. & Borchers, D. 2004. The estimation of the detection function andg(0) for short beaked common dolphins (*Delphinus delphis*), using double platform data collected during the NASS-95 Faroese survey. *Journal of Cetacean Research Management* **6(2),** 191 – 98.

Canning, S.J., Santos, M. B., Reid, R. J., Evans, P. G. H., Sabin, R. C., Bailey, N. & Pierce, G. J. 2008. Seasonal Distribution of White-Beaked Dolphins (*Lagenorhynchus Albirostris*) in UK Waters with New Information on Diet and Habitat Use. *Journal of the Marine Biological Association of the United Kingdom*, **88(6),** 1159 – 1166.

Council of the EEC 1992. Council Directive 92/43/EEC of 21 May 1992 on the Conservation of Natural Habitats and of Wild Fauna and Flora.

Darcy, M. 2013. Methods For Multivariate Longitudinal Count And Duration Models With Applications In Economics. PhD thesis: https://dspace.library.cornell.edu/handle/1813/31012.

Davison, A. C. & Hinckley, D. V. 2007. *Bootstrap Methods and Their Application*. Cambridge University Press, Cambridge. 582 pp.

Díaz López, B., Bernal Shirai, J. A., Prieto, A.B. & Fernández, P. M. 2008. Diving Activity of a Solitary Wild Free ranging Bottlenose Dolphin (*Tursiops truncatus*). *Journal of the Marine Biological Association of the United Kingdom*, **88,** 1153 – 1157.

Díaz López, B. 2009. The Bottlenose Dolphin (*Tursiops truncatus*) Foraging Around a Fish Farm: Effects of Prey Abundance on Dolphins' Behavior. *Current Zoology* **55**, 243 – 248.

Embling, C.R ., Gillibrand, P. R., Gordon, J., Shrimpton, J., Stevick, P. T. & Hammond, P.S. 2010. Using Habitat Models to Identify Suitable Sites for Marine Protected Areas for Harbour Porpoises (*Phocoena phocoena*). *Biological Conservation* **143,** 267 – 279.

ESRI, 2002. ESRI Data and Maps Media Kit. ESRI Redlands, CA.

ESRI, 2004. ESRI MapObjects 2.3. ESRI Redlands, CA.

ESRI, 2010. ArcMap 10.0. ESRI Redlands, CA.

European Commission 2009. *Report from the Commission to the Council and the European Parliament Composite - Report on the conservation Status of Habitat Types and Species as required under Article 17 of the Habitats Directive*. European Commission, Brussels. 18 pp.

Evans, P.G.H. (1990) European Cetaceans and Seabirds in an Oceanographic Context. *Lutra* **33,** 95 – 125.

Evans, P.G.H. & Wang, J. (2003) *Re-Examination of Distribution Data for the Harbour Porpoise around Wales and the UK with a View to Site Selection for this Species.* Report for the Countryside Council for Wales. Countryside Council for Wales Contract Science Report **No: 634**, 1 – 116.

Evans, W. E. 1971. Orientation Behaviour of Delphinids: Radiotelemetric Studies. *Annals of the New York Academy of Sciences.* **188,** 142 – 160.

Faraway, J. J. 2006. *Extending the Linear Model with R.* Chapman & Hall, London. 301 pp.

Haelters, J., Kerckhof, F., Thierry G., Jacques, T.G. & Degraer, S. 2011. The harbour porpoise *Phocoena phocoena* in the Belgian part of the North Sea: trends in abundance and distribution. *Belgian Journal of Zoology*, **141,** 75 – 84.

Hammond, P. S. 2006. *Small Cetaceans in the European Atlantic and North Sea (SCANS-II).* LIFE04NAT/GB/000245. Final Report. St Andrews, UK. 52 pp.

Hammond, P. S., Berggren, P., Benke, H., Borchers, D. L., Collet, A., Heide-Jørgensen, M. P., Heimlich, S., Hiby, A. R., Leopold, M. F. & Øien, N. 2002. Abundance of Harbour Porpoise and Other Cetaceans in the North Sea and Adjacent Waters*. Journal of Applied Ecology* **39,** 361 – 376.

Hammond, P. S., Macleod, K., Gillespie, D., Swift, R., Winship, A., Burt, M. L., Cañadas, A., Vázquez, J. A., Ridoux. V., Certain, G., Van Canneyt, O., Lens, L., Santos, B., Uriarte, A., Hernandez, C. & Castro. R. 2009. *Cetacean Offshore Distribution and Abundance European Atlantic (CODA).* 43 pp.

Hammond, P.S., Macleod, K., Berggren, P., Borchers, D.L., Burt, M.L., Cañadas, A., Desportes, G., Donovan, G.P., Gilles, A., Gillespie, D., Gordon, J., Hiby, L., Kuklik, I., Leaper, R., Lehnert, K., Leopold, M., Lovell, P., Øien, N., Paxton, C.G.M., Ridoux, V., Rogan, E., Samarra, F., Scheidat, M., Sequeira, M., Siebert, U., Skov, H., Swift, R., Tasker, M.L., Teilmann, J., Van Canneyt, O. & Vázquez, J.A.. (in press). Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. *Biological Conservation.*

Hanley, J. A., Negassa, A., Edwardes, M. D. D. & Forrester, J. E. E. 2003. Statistical Analysis of Correlated Data using Generalized Estimating Equations: An Orientation. *American Journal of Epidemiology* **157,** 364 – 375.

Hardin, J. W. & Hilbe, J. M. 2002. *Generalized Estimating Equations.* Chapman & Hall. London. 240 pp.

Hedley, S. L. 2000. *Modelling Heterogeneity in Cetacean Surveys.* Ph.D Thesis, University of St Andrews.

Hedley, S. L. & S. T Buckland. 2004. Spatial Models for Line Transect Sampling. *Journal of Agricultural, Biological and Environmental Statistics* **9,** 181 – 199.

Hedley, S. L., Buckland, S.T. & Borchers. D. L. 2004. Spatial Distance Sampling Models. In Buckland, S.T., D.R. Anderson, D.R. Burnham, K.P. Laake, D.L. Borchers & L. Thomas (editors). *Advanced Distance Sampling*. Oxford University Press. Pp. 48 – 70.

Heagerty, P. J. & Kurland, B. F. 2012. Misspecified maximum likelihood estimates and generalised linear mixed models. *Biometrika* **88(4),** 973 – 985.

Hiby, L. & Lovell, P. 1998. Using Aircraft in Tandem Formation to Estimate the Abundance of Harbour Porpoise. *Biometrics* **54,** 1280 – 1289.

Houghton, J. D. R., Doyle T. K., Davenport, J. & Hays, G. C. 2006. Developing a Simple, Rapid Method for Identifying and Monitoring Jellyfish Aggregations from the Air. *Marine Ecology Progress Series* **314,** 159 – 170.

Horvitz, D.G. & Thompson, D.J. 1952. A Generalization of Sampling Without Replacement From a Finite Universe. *Journal of the American Statistical Association* **47,** 663 – 685.

IAMMWG. 2015. Management Units for cetaceans in UK waters (January 2015). JNCC Report No. 547, JNCC Peterborough.

Johnson, M. E., Moore, L. M. & Ylvisaker, D. 1990. Minimax and maximin distance designs. *Journal of Statistical Planning and* Inference **26,** 131 – 148.

Joyce, G.G., Øien, N., Calambokides, J. & Cubbage, J. C. 1989. Surfacing Rates of Minke Whales in Norwegian waters. *Report of the International Whaling Commission* **39,** 431 – 434.

Kinze, C. C., Addink, M., Smeenk C., Hartmann, M. G., Richards, H. W., Sonntag R. P. & Benke, H. 1997. The White-Beaked Dolphin (*Lagenorhynchus albirostris*) and the White-Sided dolphin (*Lagenorhynchus acutus*) in the North and Baltic Seas: Review of Available Information. *Reports of the International Whaling Commission* **47,** 675 – 682.

Kwang, M.J., 2012. Effects on Regression Estimates under Misspecified Generalized Linear Mixed Models for Counts Data. *Korean Journal of Applied Statistics,* **25(6),** 1037 – 1047.

Laake, J.L. & D. L. Borchers. 2004. Methods for Incomplete Detection at Distance Zero. In Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L., Borchers, D. L. B. & Thomas, L. (editors). *Advanced Distance Sampling.* Oxford University Press. 108 – 189.

Laake, J., Borchers, D., Thomas, L., Miller, D. & Bishop J. 2011. *Mark-Recapture Distance Sampling* (mrds)v. 2.0.1. (available from https://github.com/jlaake/mrds)

Laake, J. L., Calambokidis, J., Osmek, S. D. & Rugh, D. J. 1997. Probability of Detecting Harbour Porpoise from Aerial surveys: Estimating *g*(0). *Journal of Wildlife Management* **61,** 63 – 75.

Litiere, S., Aonso, A. & Molengerghs, G. 2007. Type I and Type II error under random effects misspecification in generalized linear mixed models, *Biometrics* **63,** 1038 – 1044.

Litiere, S., Aonso, A. & Molengerghs, G. 2008. The impact of a misspecified random effects distribution on the estimation and the performance of inferential procedures in generalized linear mixed models, *Statistics in Medicine* **27,** 3125 – 3144.

Lockyer, C. & Morris, R. 1987. Observation on Diving Behaviour and Swimming Speeds in a Wild Juvenile *Tursiops truncatus. Aquatic Mammals* **13,** 31 – 35.

Macleod, K. 2004 The abundance of white-sided dolphins (*Lagenorhynchus acutus*) during summer off northwest Scotland. *Journal of Cetacean Research and Management*, **6,** 33 – 40.

Macleod, K., Fairbairns, R., Gill, A., Fairbairns, B., Gordon, J., Blair-Myers, C. & Parsons, E. C. M. 2004. Seasonal Distribution of Minke Whales *Balaenoptera acutorostrata* in Relation to Physiography and Prey off the Isle of Mull, Scotland. *Marine Ecology Progress Series* **277,** 263 – 274.

Marques, F. & Buckland, S. T. 2004. Covariate Models for the Detection Function. In *Advanced Distance Sampling.* In Buckland, S.T., D.R. Anderson, D.R. Burnham, K.P. Laake, D.L. Borchers & L. Thomas (editors). Oxford University Press. Oxford. 31 – 47..

Marques, T. A., Thomas, L., Fancy, S. G. & Buckland, S.T. 2007. Improving estimates of bird density using multiple covariate distance sampling. *The Auk* **127**, 1229 – 1243.

Mendenhall, W. (1982) *Statistics for Management and Economics.* Boston.

Ministerial Meeting of the Oslo and Paris Commissions 1992. *The Convention for the Protection of the Marine Environment of the North-East Atlantic.* 33 pp.

Northridge, S. P., Tasker, M. L., Webb, A. & Williams J. M. 1995a. Distribution and Relative Abundance of Harbour Porpoises (*Phocoena phocoena* L.), White-Beaked Dolphins (*Lagenorhynchus albirostris* Gray), and Minke Whales (*Balaenoptera acutorostrata* Lacepède) around the British Isles. *ICES Journal of Marine Science* **52,** 55 – 66.

Northridge, S. P., Tasker, M. L., Webb, A. & Williams J. M. 1995b. Erratum. *ICES Journal of Marine Science* **52,** 1005 – 1012.

Otani, S., Naito, Y., Kawamura, A., Kawasaki, M., Nishiwaki, S. & Kato, A. 1998. Diving Behavior and Performance of Harbor Porpoises, *Phocoena phocoena*, in Funka bay, Hokkaido, Japan. *Marine Mammal Science,* **14,** 209 –220.

Palka, D.L. & Hammond, P.S. (2001). Accounting for responsive movement in line transect estimates of abundance. *Canadian Journal of Fisheries and Aquatic Sciences* **58,** 777 – 787.

Pan, W. 2000, Akaike's Information Criterion in Generalized Estimating Equations. *Biometrics* **57,** 120 – 125.

Paxton, C. G. M. & Thomas, L. 2010. *Phase I Analysis of the Joint Cetacean Protocol Data*. Report to the JNCC January 2010. 70 pp.

Paxton, C. G. M., Mackenzie, M., Burt, M. L., Rexstad E. & Thomas, L. 2011. *Phase II Data Analysis of Joint Cetacean Protocol Data Resource.* Report to the JNCC 2012. 126 pp.

R Development Core Team 2011. *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.

Reid, J. B., Evans, P. J. H. & Northridge, S. P. 2003. *Atlas of Cetacean Distribution in North-West European Waters.* Joint Nature Conservation Committee, Peterborough. 76 pp.

Reynolds, R. W., Rayner, N. A., Smith, T. M., Stokes, D. C. & Wang, W. 2002. An Improved In Situ and Satellite SST Analysis for Climate. *Journal of Climate* **15,** 1609 – 1625.

Reynolds, R. W. & Smith, T. M. 1995. A High resolution Global Sea Surface Temperature Climatology. *Journal of Climate* **8,** 1571 – 1583.

Scott-Hayward, L. A. S., Mackenzie, M. L., Donovan, C., Walker, C. G. & Ashe, E. in press. Complex Region Spatial Smoother (CReSS). *Journal of Computational and Graphical Statistics.*

Skov, H. & Thomsen, F. 2008. Resolving Fine-Scale Spatio-temporal Dynamics in the Harbour Porpoise *Phocoena phocoena.. Marine Ecology Progress Series* **373,** 173 – 186.

Simons, D. J. & Chabris, C. F. 1999. Gorillas in Our Midst: Sustained Inattentional Blindness for Dynamic Events. *Perception* **28,** 1059 – 1074.

Steidl, R. J. & Thomas, L. 2001. Power Analysis and Experimental Design. In S.M. Scheiner, & Gurevitch, J. (eds). *Design and Analysis of Ecological Experiments 2nd edition.* Oxford University Press, New York. 14 – 36.

Stockin, K. A., Weir, C. R. & Pierce, G. J. 2006. Examining the importance of Aberdeenshire (UK) coastal waters for North Sea bottlenose dolphins (*Tursiops truncatus*) *Journal of the Marine Biological Association* **86,** 201 – 207.

Stone, C. J., Webb, A., Barton, C., Ratcliffe, N., Reed, T. C., Tasker, M. L., Camphuysen, C. J. & Pienkowski, M. W., 1995. *An Atlas of Seabird Distribution in North-West European Waters.* Joint Nature Conservation Committee, Peterborough. 326 pp.

Thomas, L., Burnham, K.P. & Buckland, S.T. 2004. Temporal Inferences from Distance Sampling Surveys. In Buckland, S. T., D. R. Anderson, K. P. Burnham, J. Laake, D. L. Borchers & L. Thomas (editors). Oxford University Press Oxford. 71 – 107.

Thomas L. 2009. *Potential Use of Joint Cetacean Protocol Data for Determining Changes in Species' Range and Abundance: Exploratory Analysis of Southern Irish Sea Data.* Report to the JNCC January 2009. 35 pp.

Thomsen, F., Laczeny, M. & Piper, W. 2006. A Recovery of Harbour Porpoises (*Phocoena phocoena*) in the Southern North Sea? A Case Study off Eastern Frisia, Germany. *Helgoland Marine Research* **60,** 189 – 195.

van der Meij, S. E. T. & Camphuysen C. J. 2006. Distribution and diversity of whales and dolphins (Cetacea) in the Southern North Sea: 1970-2005. *Lutra* **49,** 3 – 28.

Walker, C. G., Mackenzie, M. L., Donovan, C. R. & Sullivan, M. J. 2011. SALSA – A Spatially Adaptive Local Smoothing Algorithm. *Journal of Statistical Computation and Simulation*, **81,** 179 – 191.

Weir, C. R., Pollock, C., Cronin, C. & Taylor, S. (2001) Cetaceans of the Atlantic Frontier, North and West of Scotland. *Continental Shelf Research* **21,** 1047 – 1071.

Weir, C. R., Macleod, C. D. & Calderan, S. V. 2009. Fine-scale Habitat Selection by White-beaked and Short-Beaked Common Dolphins in the Minch (Scotland, UK): Evidence for Interspecific Competition or Coexistence? *Journal of the Marine Biological Association of the United Kingdom* **89,** 951 – 960.

Wells, R.S., Manire, C.A., Byrd, L., Smith, D.R., Gannon, J.G., Fauquier, D. & Mullin, K.D. 2009. Movements and Dive Patterns of a Rehabilitated Risso's dolphin, *Grampus griseus*, in the Gulf of Mexico and Atlantic Ocean *Marine Mammal Science* **25(2),** 420 – 429

Winship, A. J. 2008. Analysis of Harbour Porpoise Sighting Rates from the European Seabirds at Sea database. Appendix D2.4 of *Small Cetaceans in the European Atlantic and North Sea* (SCANS-II) Final Report. EU Life project LIFE04NAT/GB/000245.

Wood, S. N. 2006. *Generalized Additive Models: An Introduction with R.* Chapman and Hall. London. 391 pp.

Wood, S. N. 2011. *mgcv 1.7-6* .Available from http://cran.r-project.org.

Würsig, B, 2009. Bow riding. In *Encyclopedia of Marine Mammals.* (Perrin, W. F., Würsig, B., Thewissen, J. G. M. eds). Academic Press. London. 131 – 133.

WWT Consulting 2009. *Distributions of cetaceans, seals, turtles, sharks and ocean sunfish recorded from Aerial Surveys 2001-2008.* Report to Department of Energy and Climate Change.

Appendix 1. A Users Guide To Submitting Data to the Joint Cetacean Protocol.

Summary of Data Requirements

Data must be organised into two separate tables: one containing effort and the other sightings information with associated dates, times and positions relating them together, or a common code (see accompanying Excel spreadsheets at http://www.ruwpa.st-and.ac.uk/dpwebi/jcp/ and field descriptions below).

Effort data should be supplied with geographic information showing start and end positions and times of each transect or survey segment. Ideally effort should be offered in two formats: as spatial and temporal waypoints and as segmented data with the start and end positions of each segment.

In addition to sightings and effort tables, type of survey (e.g. line transect, double platform etc.) and boundaries of the targeted survey region should also be given.
Any abbreviations or codes should be clearly explained.

Data Tables

Please use the following guidelines and associated for information on how data must be submitted. Attributes highlighted by <u>underlining</u> must be provided.

Waypoint Effort

<u>Date; Time; TimeZone; Lat; Lon</u>; Observers; NoObs

Include the date, time and position (in decimal degrees and waypoints regularly throughout transect whenever possible) that surveyors started and finished observing as well any observer change. Date should be recorded as day, month and year (dd/mm/yyyy), and time in 24 hour format with time zone detailed in a separate field. This allows estimation of speed and enables anomalous positions/timings to be identified; Observer names and the number of observers should also be included.

<u>VesselName</u>; EyeHeight; VesselType; ObsvPlatform
Vessel name; eye height (in meters above water); vessel type (e.g. ship); and observation platform type (e.g. crow’s nest).

<u>Effort; Method; NoPlatforms</u>
Type of effort (e.g. start, mid-point or end of transect/segment); survey method (e.g. line transect with distance to sightings); number of platforms (single/double) and any change in status (e.g. method) must be provided. Effort can be split into segments with identical environmental/watch conditions (see below).

TransectID; *Strata; Course;* *PlatSpeedKm*
A unique code or transect ID, which identifies each effort record/segment, and links this to any sightings data, should be provided whenever possible. This could be facilitated by each survey being assigned a unique ID prefix for effort segments.
Survey strata (supplemented with explanatory text when data are submitted); platform course and speed should all be included when available.

Seastate; *Vis; Cloud; Swell; WindSpe; WindDir; PrecipTyp; PrecipInt; Depth; SST*
Beaufort sea state must be included and, when available, other environmental variables such as visibility, cloud cover, wind speed, precipitation type, precipitation intensity, general sightability, depth, sea surface temperature etc.

Segment Effort

Transects when broken down into discrete sections or segments, when analysed with platform speed, will allow identification of errors in the data. It is, therefore, useful that data are provided as such whenever possible. Please see the segmented data spreadsheet at http://www.ruwpa.st-and.ac.uk/dpwebi/jcp/ for a full list of attributes and use the descriptions detailed above and below.

Single Platform Sightings

Date; Time; TimeZone; Lat; Lon

These should be recorded for each sighting of a cetacean/group and link sightings data to effort data (also see above). Ideally this should be by date and time, rather than just a code, as the data may have to be re-segmented.

TransectID
Links sightings to effort.

RadialDist; PerpenDist; Inclination; Bearing; etc.
Additional fields for identifying position of cetacean in relation to platform. These distance data should be provided whenever possible and be clearly identified as radial or perpendicular distance or by other survey relevant details (inclination, bearing, aeroplane height etc.).

Species; SpeCertainty; TotalGroup; Adult; Juv; Calf
Please provide details of species and group size; ideally broken down by class and confidence of accurate identification to species level.

VesselName
Please provide a vessel name even if only one vessel was used. Data from a given vessel should be collected together in CHRONOLOGICAL order.

SegmentID
Links sightings to segmented effort table if provided.

UniqueSightingID
Combination of data contributors prefix code and TransectID which provide a unique reference for each sighting.

Additional Data

Please provide any additional data that you feel will be beneficial e.g. behaviours, cues etc.

Double Platform Sightings

If double platform data have been collected, an obvious way to record this is with two lines for each platform with a unique sighting identifier, a platform identifier (primary or secondary etc.) and another field indicating whether the particular animal was seen or not. Thus every encounter should have two lines associated with it. Please see the accompanying double platform sightings data spreadsheet for a full list of attributes and use the descriptions detailed above.

Metadata

Please complete as many of the fields as possible and send additional files, if deemed appropriate, e.g. shapefile of survey area, survey report, etc. If not all species were recorded or identified upon observation please make a note of which species would be recorded or identified. Also please explicitly state if all relevant sightings have been forwarded to the analysts. Omission of sightings data is fine if and only if the analysts know it has been omitted! Details should be supplied of the observers' search protocol (i.e. how far forward are they searching and the pattern of searching about the bow).

Please Note

Casual, unsystematic, observations not associated with formal commencement of search effort must NOT be included

Serious consideration should be given to assigning a unique identifier to each sighting in the combined database, prior to hand-over for analysis. This could be facilitated by each survey being assigned a unique ID prefix for sightings. This would go a long way towards preventing the occurrence of overlapping data by different data providers.

> If not contained in the above tables (the preferred option), there should be a statement briefly describing the conditions of the survey, number of observers, search pattern, platform heights, mode of survey (casual watches etc.), type of survey (single or double platform etc.), vessel types etc.

Data Checking

The JCP has established an online checking tool at http://www.ruwpa.st-and.ac.uk/dpwebi/jcp/ This allows the data to be checked for whether all the columns are present and whether there are any problems with the data.

The specific tests are:

1. Are all essential column names present?
2. Are all records in chronological order within *VesselName* (this is important for subsequent resegmentation etc.)?
3. Does all effort have explicit start and end positions identified?
4. Are all the start and end codes in the correct order?
5. Are the speeds excessive (incorrect speeds reflect typically errors in time or position)?
6. Is each sighting record associated with bona fide effort?

Very important

Data **must** be in csv format with separate effort and sightings files (named “effort.csv” and “sightings.csv” respectively).

There **must** be no commas, semicolons and quotes in the data (this makes the data checker breakdown). Also there must be no “#VALUE” or “#DIV0” (caused by functions not working in Excel) terms in **any** of the cells in the files.
The maximum possible survey speed should be given in knots.

Simply upload the relevant files by pressing the buttons on the web page.

Any queries about running the above software should be addressed to Charles Paxton: cgp2@st-andrews.ac.uk .

Appendix 2. Description of the Datasets and Inclusion Criteria

This appendix describes the data sets in detail and should be read in conjunction with section 2.1. There were 38 usable distinct datasets in Phase III comprising several hundred distinct surveys in space and time. This section describes each dataset. Some datasets overlapped with each other. Duplicate data were identified and deleted.

Table A2.1 summarizes the total annual search effort for each of the datasets. Only search effort collected within depths of less than 300m and in Beaufort sea states less than 4 was included (with the exception of some aerial surveys where sea state was stated only to be less than a figure greater than 4). Table A2.2 summarizes search effort by vessel type. Data was collected from both sides of the vessel or plane unless otherwise stated.

APEM/Scottish Power Renewables surveys
These aerial photo surveys were carried out in 2010 of the coast of Norfolk (figure A2.a). Because they are photo surveys, there was assumed to be no perception bias. This data was only used for the harbour porpoise analysis as animals that were not harbour porpoise were not identified to species.

Cardigan Bay Marine Wildlife Centre surveys (CBMWC)
These data were collected using the *littleboats Sulaire* and *Orca* from 2005 to 2009 (figure A2.b). The sightings data typically contained exact perpendicular distances.

Centrica surveys
These dedicated boat surveys were undertaken in 2010 in the region of the *Irish Sea* developer region (i.e. between Anglesey and the Isle of Man, figure A2.c). Perpendicular distances were available for this survey.

Cetacean Offshore Distribution and Abundance in the Offshore Atlantic (CODA).
This dedicated large scale boat survey took place in 2007, on the European continental slope and abyssal waters, so only a little effort was included in the analysis (figure A2.d). Details can be found in Hammond *et al* (2009). Perpendicular distances were collected under a double platform protocol allowing estimation of $g(0)$.

Cornwall Wildlife Trust surveys
These data were collected from two *bigboats* during 2009 and 2010 on surveys to and from the Isles of Scilly (figure A2.e). Exact perpendicular distances were available for each sighting.

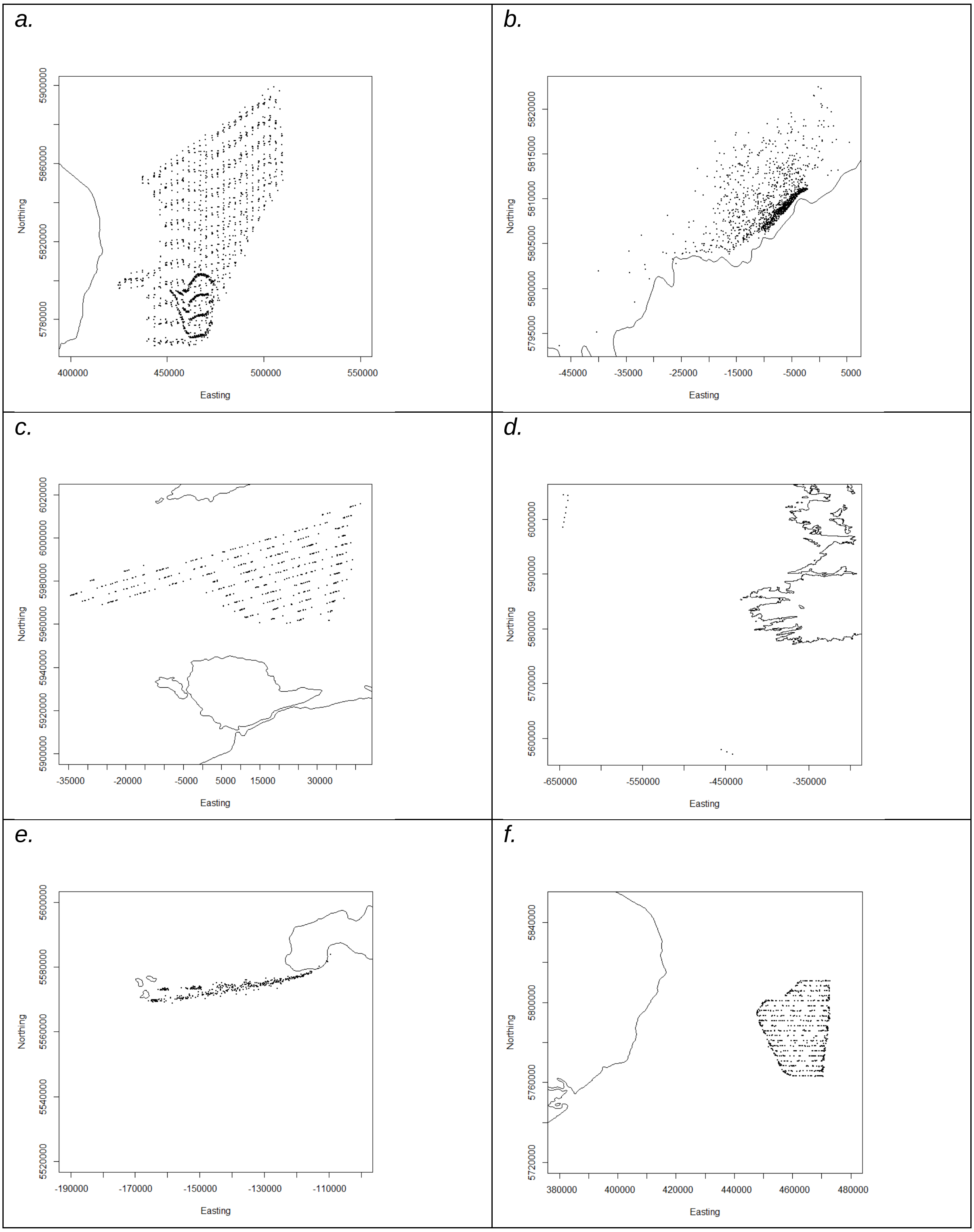

Figure A2. Realized (i.e. as used in models) survey effort for each data source. a. APEM. b. CBMWC, c. Centrica surveys, d. CODA, e. Cornish Wildlife Trust. f. EAOW. Each point represents the centre of a segment of effort.

East Anglia Offshore Wind Ltd. (EAOW) and Scottish Power Renewables Survey
This *bigboat* survey was undertaken in 2010 off the coast of Norfolk (figure A2.f). No distances were available.

RWE NPower plc Atlantic Array Offshore Wind Farm Project Surveys
Two *bigboats* were used for these dedicated marine mammal surveys conducted in 2010 and 2011 by Chris Pierpoint at the Atlantic Array in the Bristol Channel (figure A2g). Exact perpendicular distances were available for each sighting.

European Seabirds at Sea (ESAS) shipboard surveys
These surveys date back to 1979 but only data from 1994 onwards have been utilised here. Numerous *littleboats, bigboats* and *ferries* were used on these surveys from 1994 to 2010 (figure A2.h, black). Details of the early surveys can be found in Stone *et al* (1995). Perpendicular distances were generally collected in distance intervals although sometimes sightings were just known to be within a 300 m horizontal distance with the exact distance not known. ESAS North Sea harbour porpoise sightings were previously analysed by Winship (2008); however, data analysed here were not treated as per that analysis in a number of respects. Winship's (2008) altering speed criteria (< 50 km/h) for identifying periods of non-effort was not implemented and vessel type was included in the analysis (see below). However, like Winship (2008), effort associated with surface speeds in excess of, that possible for boat based surveys, was excluded. Banded distances for these surveys were assumed to be initially observed perpendicular distances although for ESAS, this is not necessarily the case (Northridge *et al* 1995).

European Seabirds at Sea (ESAS) aeroplane survey
This survey was conducted in 1994 (figure A2.h, red). Perpendicular distances were usually collected in distance intervals, although some data were recorded solely as within 300 m. No animals were actually seen on this survey in 1994.

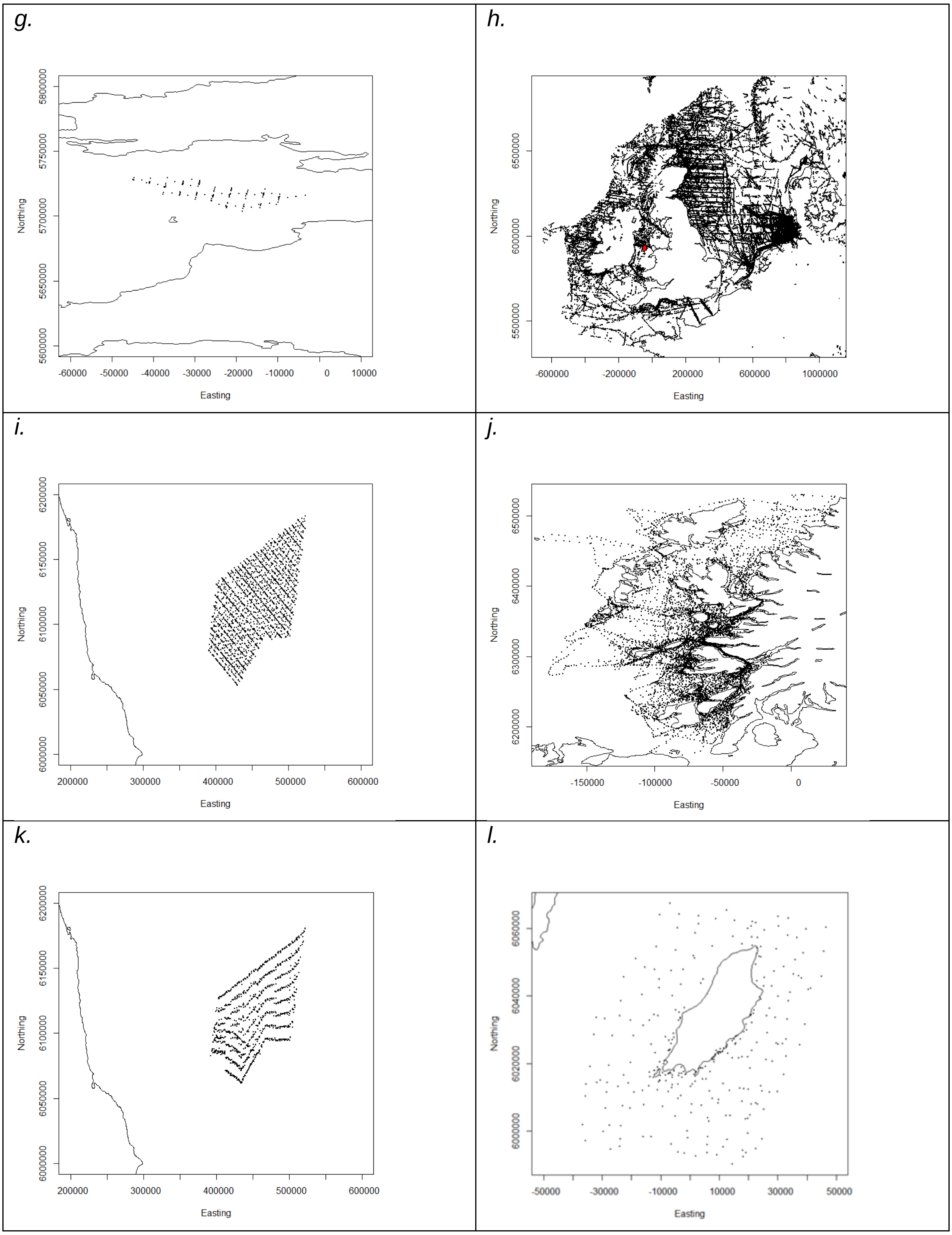

Figure A2.1. Continued. Realized (i.e. as used in the spatial models) survey effort for each data source. g. RWEnPower survey, h. ESAS surveys, black: ship, red: aerial effort, i. Forewind. j. HWDT, k. HIDEF, l, Manx Whale and Dolphin Trust . Each point represents the centre of a segment of effort.

Forewind Ltd Survey

This boat survey was undertaken in 2010, in the region of the Dogger Bank (figure A2.i). No distances were available.

Hebridean Whale and Dolphin Trust (HWDT) surveys

These eight surveys took place between 2003 and 2010 using a *bigboat* around the Hebrides and north-east coast of Scotland (figure A2.j). Exact perpendicular distances were available for each sighting.

HiDef

Aerial photo-surveys undertaken over the Dogger Bank in 2010 (figure A2.k). All animals at the surface were assumed to be seen.

Manx Whale and Dolphin Trust survey

This survey was conducted around the Isle of Man in 2008 (figure A2.l) by the Manx Whale and Dolphin Trust. A *littleboat* was used and perpendicular distances were available for all sightings.

Irish Whale & Dolphin Group (IWDG) surveys

A variety of *bigboat* size survey vessels have been used to collect these data for the years 2003 to 2009 (figure A2.m). Additionally six ferries have been used to collect data for the years 2001 to 2010. Exact perpendicular distances were available for each sighting.

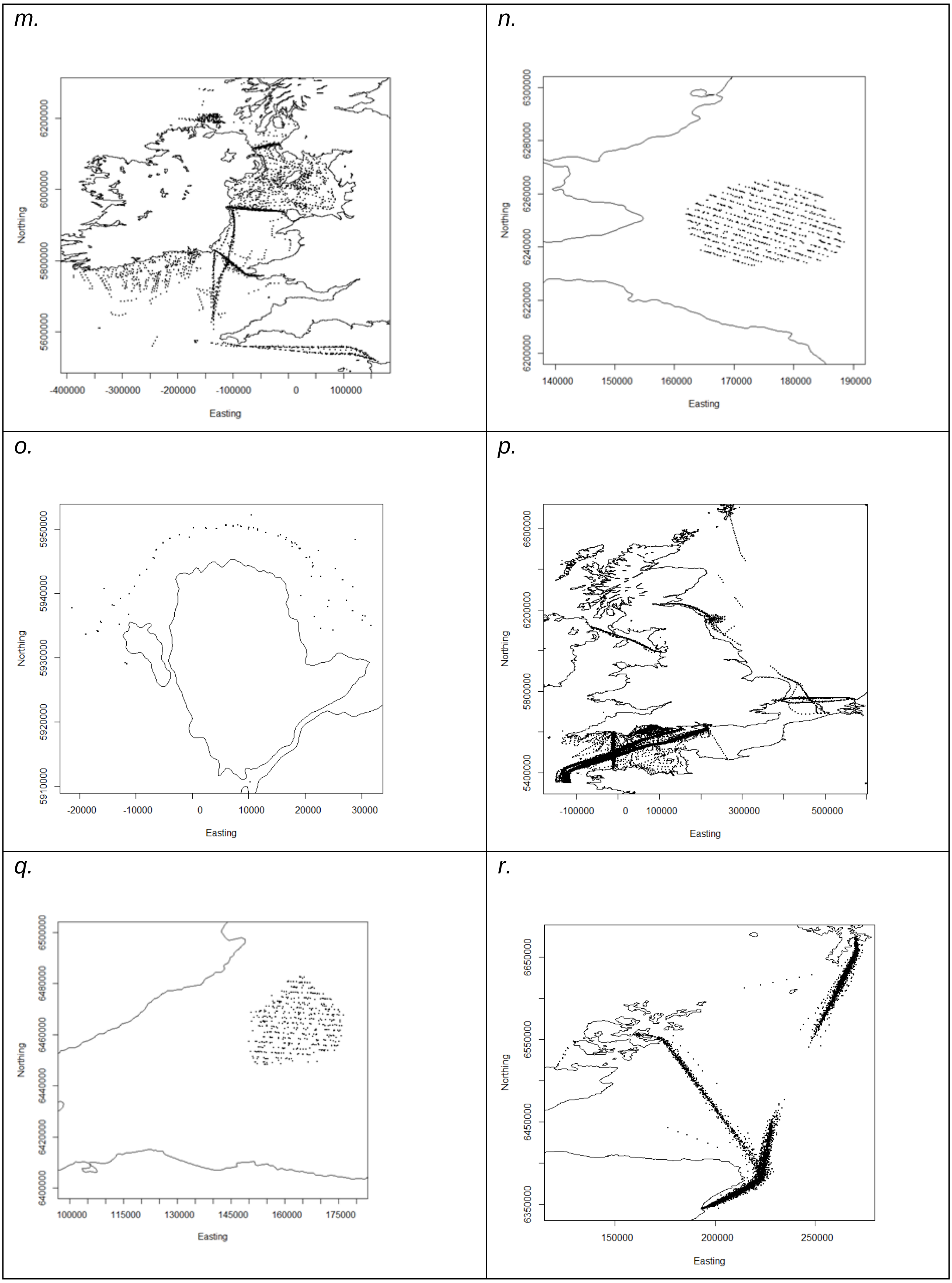

Figure A2.1. Continued. Realized (i.e. as used in the spatial models) survey effort for each data source, m. IWDG, n. MainstreamRP, o. MANW, p. MARINELIFE, q. MORL boat, r. NORCET. Each point represents the centre of a segment of effort.

Mainstream Renewable Power (Neart na Gaoithe)
Boat surveys undertaken in 2009 and 2010 off the coast of Fife (figure A2n). Distances were mostly available. This one-sided survey was treated as two-sided here.

Marine Awareness North Wales (MANW) Wildlife Trust surveys
Six surveys were conducted using *littleboats* from 2002 to 2004 and 2006 to 2008 around Anglesey (figure A2o). Exact perpendicular distances were available for each sighting.

Marine Conservation Research Ltd/International Fund for Animal Welfare (IFAW)
A boat survey undertaken over the continental slope of the western approaches in 2010, perpendicular distances were available. Only a very small amount of effort was available on the continental shelf (not shown).

MARINElife/BDRP surveys
A mix of *littleboats*, *bigboats* and *ferries* were used to collect data from 1995 to 2010 (figure A2p). Exact perpendicular distances were available for all sightings.

Moray Offshore Renewables Ltd (MORL) boat surveys
Surveys undertaken in the outer Moray Firth in 2010 (figure A2q). Distances were available.

Northern North Sea Cetacean ferry (NORCET) surveys
Platform of opportunity surveys undertaken from ferries going to and from Aberdeen (figure A2r). Data was available from 2004 to 2009. These surveys had an asymmetrical survey method with observations 90° to the left and 45° to the right of the bow.

North Wales Windfarm Environmental Assessment surveys
These two surveys were conducted in *bigboats* (10 m observation platform, Goold pers com) off the coast of North Wales in 2003 and 2004 (figure A2s). Exact perpendicular distances were available for all sightings.

ORCA and Company of Whales (CoW) surveys
Surveys were conducted in 2009 and 2010 from various ferry routes (figure A2t). Exact perpendicular distances were available for all sightings.

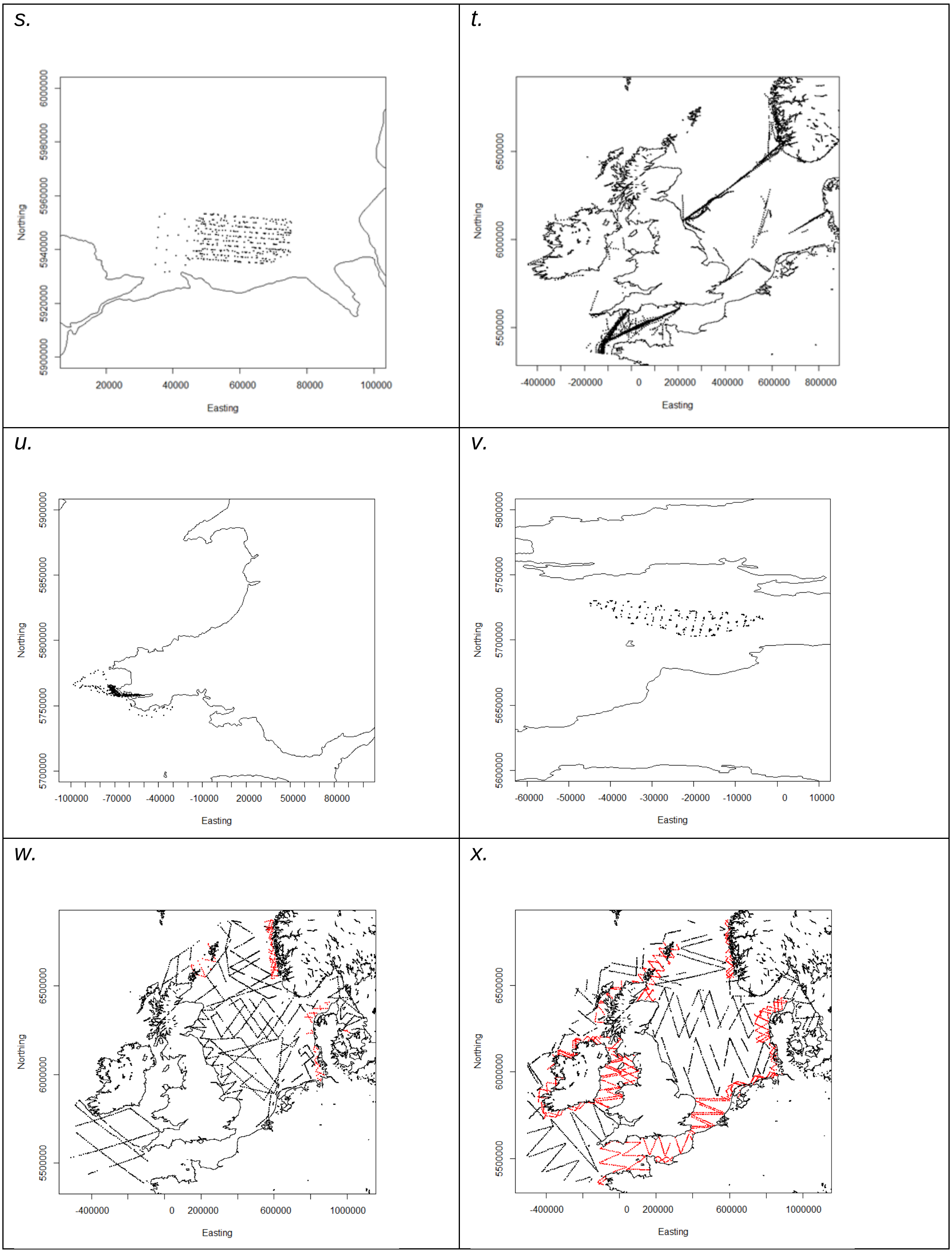

Figure A2.1. Continued. Realized (i.e. as used in the spatial models) survey effort for each data source s. NWWEA, t. ORCA/CoW, u. Pembrokeshire Porpoise (SWF), v. RWE nPower Renewables, w. SCANS, v. SCANS-II. For the latter two surveys aerial effort is shown in red. Boat effort in black. Each point represents the centre of a segment of effort.

Pembrokeshire porpoise surveys (Sea Watch Foundation)

The main species of interest for these two surveys of south Wales (figure A2u) were harbour porpoise although other cetaceans were identified. Exact perpendicular distances were available for some sightings.

RWE NPower plc Atlantic Array Offshore Wind Farm Project Surveys

Nineteen seabird surveys (2009 – 2011) were conducted from *bigboats* by ECON (figure A2v). Marine mammals were also recorded. Perpendicular distances were recorded in distance intervals for each sighting in the style of ESAS.

SCANS

This survey was conducted in 1994 (Hammond *et al* 2002) using *bigboats* (figure A2w, black) with additional aerial coastal surveys (figure A2w, red). The majority of search effort was conducted in double platform mode and exact perpendicular distances were available for each sighting.

SCANS-II

This survey was conducted in 2005 using *bigboats* (figure A2x, black) with additional aerial coast surveys (figure A2x, red). The majority of search effort was conducted in double platform mode and exact perpendicular distances were available for each sighting. The data are described in detail in SCANS-II (2008).

Scottish Power Argyll Array Survey

A boat survey undertaken off Islay in 2009 and 2010 (figure A2y). Perpendicular distances were not available.

Sea Energy Renewables (now EDP Renewables and Repsol Nuevas Energias UK) Inch Cape survey

A boat survey undertaken off Angus in 2010 (figure A2z). Perpendicular distances were not available.

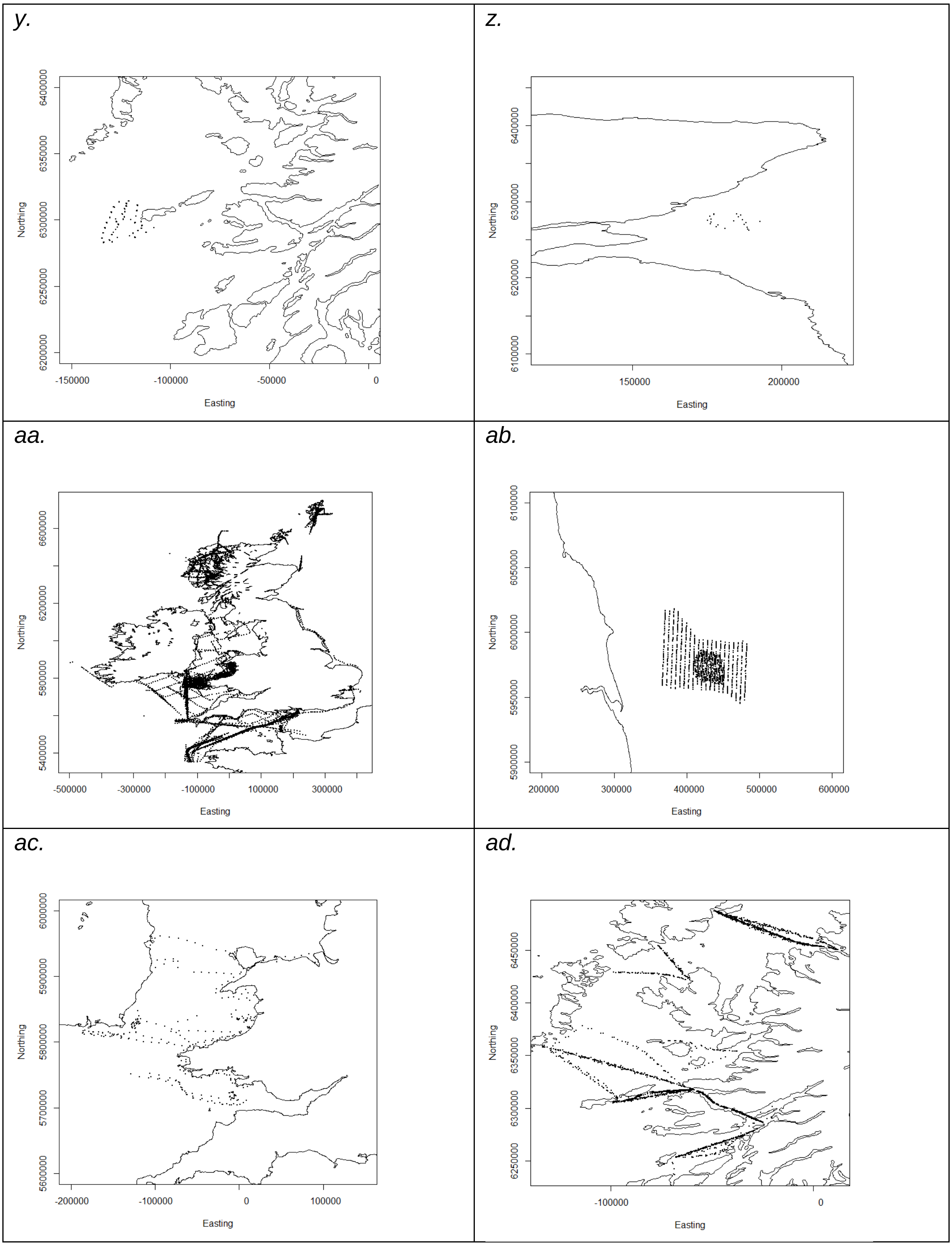

Figure A2.1. Continued. Realized (i.e. as used in the spatial models) survey effort for each data source y. Scottish Power Argyll array survey, z. Sea energy renewables Inch Cape survey, aa. Sea Watch Foundation surveys, ab. SMart Wind surveys, ac. Swansea University surveys, ad. University of Aberdeen platform of opportunity surveys, Each point represents the centre of a segment of effort.

Sea Watch Foundation (SWF) surveys

Surveys conducted from 1994 to 2008 around the British Isles (figure A2aa). Earlier surveys have been excluded from this analysis. Both *big* and *littleboats* were used. The Irish Sea data are described in detail in Baines & Evans (2009). A *littleboat* was used in this survey conducted in 2008 around Cardigan Bay and the North Wales coast. Exact perpendicular distances were available for most sighting and some data was collected in double platform mode.

SMart Wind

Boat surveys off the Yorkshire coast in 2010 (figure A2.ab). Distances were available. However we have just recently discovered this survey was conducted on one side of the vessel only. It was treated as two-sided in the analysis.

Swansea University surveys

Swansea University conducted aerial surveys in 2004 and 2005 (Houghton *et al* 2006) across the Irish Sea (figure A2.ac). Exact perpendicular distance were available for each sighting. This survey was mistakenly referred to as the “CCW survey” in Paxton & Thomas (2010).

University of Aberdeen platform of opportunity surveys

These data were collected from 2001 to 2006 on 36 ferry routes to, from and around the Hebrides (figure A2.ad). Exact perpendicular distances (as opposed to distances in interval intervals) were available for each sighting.

University of Aberdeen aerial surveys

These were aerial visual surveys conducted in 2010 in the Moray Firth (figure A2.ae). Distances were available. There were also aerial photo surveys in the same area (see below).

University of Aberdeen boat survey

A *bigboat* survey undertaken in the Moray Firth in 2009 (figure A2.af). Distances were available.

University of Aberdeen SAC Survey

A *littleboat* survey undertaken in the inner Moray Firth in 2004 and 2005 (figure A2.ag). Distances were available.

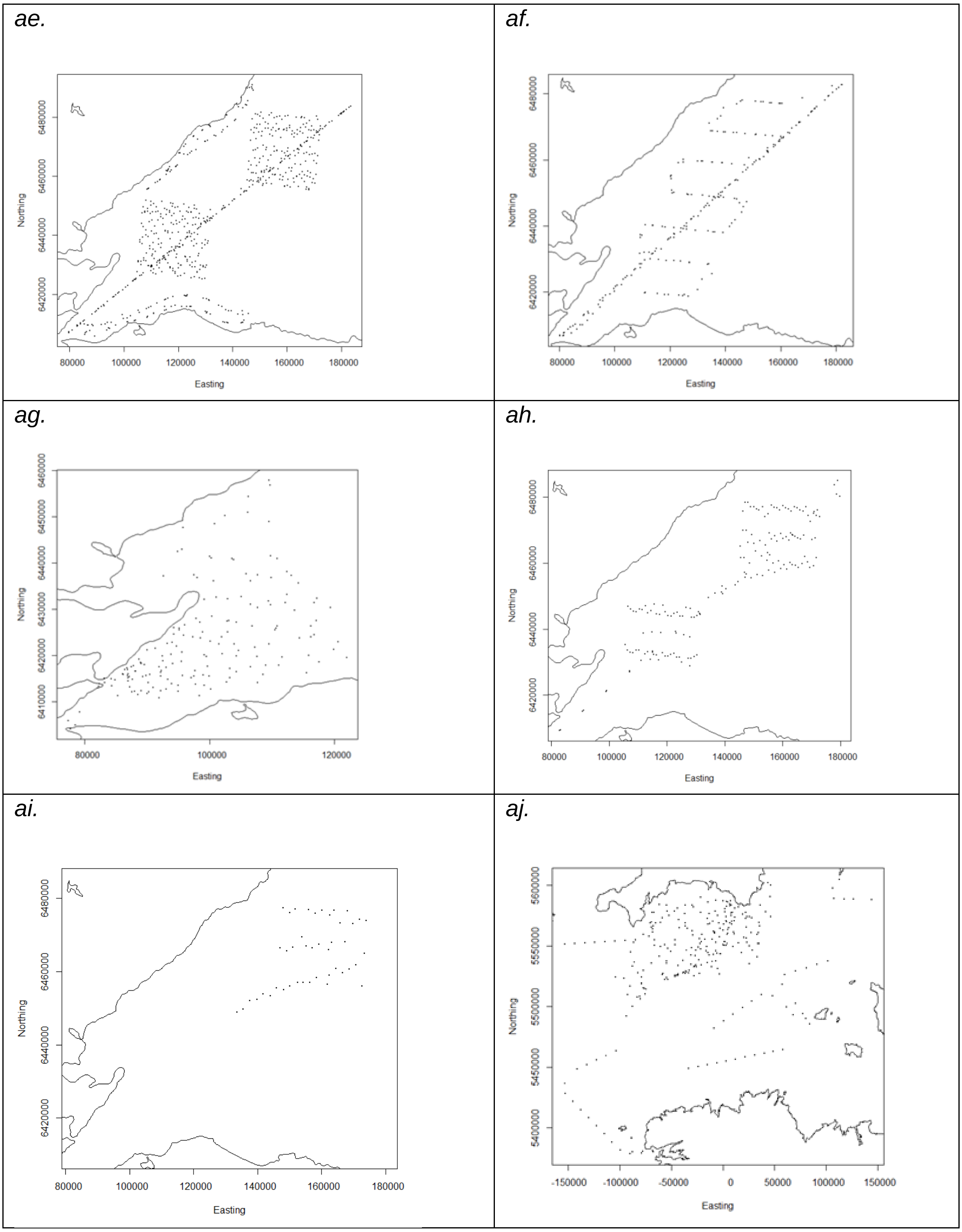

Figure A2.1. Continued. Realized (i.e. as used in the spatial models) survey effort for each data source ae. University of Aberdeen aerial surveys, af. University of Aberdeen boat surveys, ag. University of Aberdeen/SAC surveys, ah. University of Aberdeen/MORL/HiDef/The Crown Estates, ai. University of Aberdeen/MORL/HiDef, ai. Universityof Aberdeen/MORL/HiDef/theCrown Estates, aj WDCS survyes. Each point represents the centre of a segment of effort.

ak.

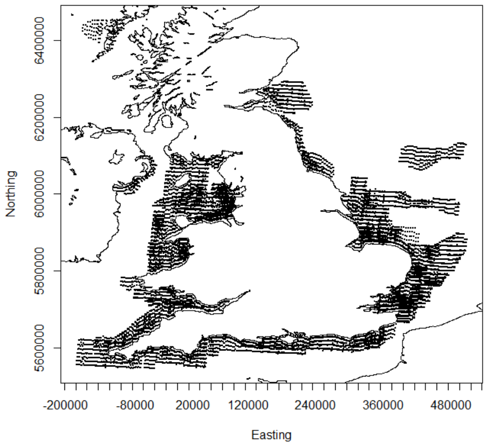

Figure A2.1. Continued. Realized (i.e. as used in the spatial models) survey effort for each data source ak. WWT surveys. Each point represents the centre of a segment of effort.

University of Aberdeen/MORL/HiDef/The Crown Estates
An aerial survey undertaken in the Moray Firth in 2010 (figure A2.ah). Distances were available.

University of Aberdeen/MORL/HiDef
An aerial survey undertaken in the Moray Firth in 2009 (figure A2.ai). Distances were available.

Whale and Dolphin Conservation Society
Surveys undertaken in the western English Channel from platforms of opportunity in 2004 and 2005 (figure A2.aj). Distances were available. In revision it was discovered that not all species sightings details hand been supplied (de Boer pers comm.) and in the case of Risso’s dolphins, bottlenose dolphins and white-sided dolphins, we assumed they were not seen rather than we did not have the information.

Wildfowl and Wetlands Trust (WWT) surveys
WWT conducted aerial surveys from 2001 to 2009 (figure A2.1.ak). Perpendicular distances were recorded in three or four distance intervals. Some of these data are described in WWT Consulting (2009).

Table A2.1. Realized annual search effort (km) for each dataset

a) 1994 - 2000

Dataset	***VesselType***	***1994***	***1995***	***1996***	***1997***	***1998***	***1999***	***2000***	***Total***
ESAS	Littleboats, bigboats & Ferries	37690	23836	15083	29150	20110	6029	6973	**138870**
ESAS	Aeroplane	185	0	0	0	0	0	0	**185**
MARINElife	Bigboat	0	1426	4071	4836	4014	3894	3267	**21508**
Orca	Ferry	0	0	0	0	516	116	467	**1099**
SCANS	Bigboat	16265	0	0	0	0	0	0	**16265**
SCANS	Aeroplane	2374	0	0	0	0	0	0	**2374**
Sea Watch Foundation	Littleboat, bigboats and ferries	9750	8304	9670	9447	8457	2848	2811	**51385**
Total		**66264**	**33566**	**28824**	**43433**	**33097**	**12887**	**13518**	**231686**

b) 2001 - 2006

Dataset	***VesselType***	***2001***	***2002***	***2003***	***2004***	***2005***	***2006***	***Total***
CBMWC	Littleboat	0	0	0	0	1460	1984	**3444**
ESAS	Littleboats, bigboats & Ferries	12539	11797	11012	5849	13211	7427	**61835**
HWDT	Littleboat	0	0	1390	1011	1487	3379	**7267**
IWDG	Ferries & Bigboats	706	3567	6119	5335	3000	3152	**21880**
MANW	Littleboat		375	199	281	0	87	**942**
MARINElife	Bigboat	4262	3461	4306	3562	5010	5508	**26109**
NORCET	Ferry	0	0	0	900	876	7659	**9435**
Orca	Ferry	1378	1943	1671	492	603	5397	**11483**
North Wales Windfarm	Littleboat	0	0	269	2382	0	0	**2651**
SCANS-II	Bigboats	0	0	0	0	13059	0	**13059**
SCANS-II	Aeroplane	0	0	0	0	14647	0	**14647**
Sea watch Foundation	Littleboat, bigboats and ferries	4081	5236	2808	2222	4764	5246	**24357**
University of Aberdeen ferry surveys	Ferry	136	2535	6017	5287	3884	3710	**21567**
University of Aberdeen SAC	Littleboat	0	0	0	224	961	0	**1185**
University of Swansea	Aeroplane	0	0	0	2840	242	0	**3081**
WDCS	Bigboat	0	0	0	1475	1090	0	**2565**
WWT	Aeroplane	3294	12378	16380	22849	52061	32120	**139083**
Total		**26396**	**41292**	**50172**	**54709**	**116355**	**75669**	**364593**

c) Years 2007 - 2010

Dataset	***VesselType***	***2007***	***2008***	***2009***	***2010***	***Total***
APEM	Aeroplane	0	0	0	20906	**20906**
CBMWC	Littleboat	1153	1369	1684	0	**4206**
Centrica	Bigboat	0	0	0	5728	**5728**
Cornish	Bigboat	0	0	1730	1915	**3645**
CODA	Bigboats	91	0	0	0	**91**
ESAS	Littleboats, bigboats & Ferries	5695	4209	5549	11606	**27059**
Forewind	Bigboat	0	0	0	13772	**13772**
HIDEF	Aeroplane	0	0		17993	**17993**
HWDT	Littleboat	3369	6214	7573	3953	**21109**
IFAW	Bigboat	0	0	0	27	**27**
IWDG	Bigboats and ferries	2592	6731	8062	3443	**20828**
MainstreamRP	Bigboat	0	0	632	3527	**4159**
MANW	Littleboat	37	49	0	0	**86**
MANX	Littleboat	2001	0	0	0	**2001**
Marinelife	Bigboat	6963	15030	20341	23003	**65337**
MORL boat	Bigboat	0	0	0	3112	**3112**
NORCET	Ferries	4433	12130	8988	0	**25551**
Orca	Ferry	7053	6604	7411	6285	**27353**
Pembrokeshire Porpoise Surveys	Littleboat	2193	1631	0	0	**3824**
RWEnPower	Bigboat	0	0	0	1466	**1466**
RWEnPower (Seabird Survey)	Bigboat	0	0	1711	2193	**3904**
Scottish Power and EAOW	Bigboat	0	0	0	2739	**2739**
Scottish Power Argyll Array	Bigboat	0	0	518	1823	**2341**
Sea Energy Renewables (Inchcape boat)	Bigboat	0	0	0	203	**203**
Sea Watch Foundation	Littleboat, bigboats and ferries	3084	3944	0	0	**7028**
Smartwind Hornsea	Bigboat	0	0	0	13674	**13674**
University of Aberdeen	Aeroplane	0	0	0	4629	**4629**
University of Aberdeen/MORL/HiDef	Aeroplane	0	0	437	0	**437**
University of Aberdeen/MORL/HiDef/Crown Estate	Aeroplane	0	0	0	1708	**1708**
University of Aberdeen	Bigboat	0	0	1450	0	**1450**
WWT	Aeroplane	58300	66325	46738	0	**171363**
Total (not including APEM)		**96964**	**124236**	**112824**	**122802**	**456826**

Table A2.2. Effort (km) by vessel type by year (for full data set i.e. not the abbreviated harbour porpoise data set).

Year	**Littleboat**	**Bigboat**	**Ferry**	**Plane**
1994	11605	38959	13141	2559
1995	5413	15612	12540	0
1996	3730	8687	16407	0
1997	3693	21158	18582	0
1998	4113	12369	16615	0
1999	1486	6953	4449	0
2000	1822	7226	4571	0
2001	3275	9893	9934	3294
2002	4122	12361	12431	12378
2003	3171	12327	18295	16380
2004	4909	8451	15661	25689
2005	6174	28443	14788	66950
2006	6771	10943	25835	32120
2007	9054	9156	20455	58300
2008	8062	10931	38918	66325
2009	8748	22809	34092	47176
2010	3468	50984	44019	24330
Total	**89614**	**287261**	**320731**	**355501**

Appendix 3. Parameters of Detection Functions

Table A3.1. Parameters of the fitted detection functions.

Species	***Data type***	***Model Type***	***Parameters***	
			Source	***Estimate (SE)***
Harbour porpoise	Non-ESAS boats	HR	Scale	
			Intercept	5.852 (0.043)
			VesselType = Ferry	-0.074 (0.047)
			VesselType = Littleboat	-0.717 (0.047)
			SeaState	-0.207 (0.026)
			Shape	
			Intercept	0.734 (0.035)
	ESAS boats	HN	Scale	
			Intercept	3.958 (0.17)
			SeaState	0.696 (0.99)
			VesselType = Ferry	-0.391 (0.82)
	WWT (old binning) aerial	HN	Scale	
			Intercept	5.249(0.156)
	WWT (new binning) aerial	HN	Scale	
			Intercept	4.728 (0.009)
	University aerial	HN	Scale	
			Intercept	4.940 (0.090)
Minke whale	Non-ESAS boats	HN	Scale	
			Intercept	5.555(0.153)
			SeaState	-0.129 (0.067)
			Shape	
			Intercept	0.422 (0.095)
	ESAS	HN	Scale	
			Intercept	3.957 (0.122)
	WWT (new binning) aerial	HN	Scale	
			Intercept	5.772 (0.434)
Dolphin	Non-ESAS boats	HR	Scale	
			Intercept	5.481 (0.108)
			VesselType = Ferry	0.090 (0.086)
			VesselType = Littleboat	-0.361 (0.087)
			Species = CD	0.047(0.071)
			Species = RD	0.711 (0.134)
			Species = WBD	0.290 (0.113)
			Species = WSD	0.081 (0.219)
			SeaState	-0.188 (0.029)
			size	
			Shape	
			Intercept	0.690(0.050)
	ESAS boats	HN	Scale	
			Intercept	4.939 (1.597)
			Species = CD	-1.423 (1.597)
			Species = RD	-0.644 (1.799)
			Species = WBD	-1.172 (1.596)
			Species5WSD	-1.330 (1.604)
			SeaState	-0.005 (0.039)
			VesselType=Ferry	-0.199 (0.119)
	University aerial	HN	Scale	
			Intercept	4.887 (0.169)
	SCANS aerial	HN	Scale	
			Intercept	4.273 (0.113)
	WWT (new binning) aerial	HN	Scale	
			Intercept	4.827 (0.053)

Appendix 4. Example Densities

In each of the following figures there are six graphs illustrating for a given time point the input densities $\widehat{D}_c$, the point estimate prediction of density and the lower (2.5%) and upper (97.5%) cell confidence limits. The first three graphs show the figures for day 227 for each reporting period. The last three graphs show the winter (day 45), spring (day 135) and autumn (day 315) 2010 figures. In the former case figures are all means over the estimates for day 227 across all years of the reporting period.

The graph of $\widehat{D}_c$ is a gridded representation of the mean estimated density per segment at a 900 km^2 resolution (the grid is 53 by 60 covering a north-south distance of 1560 km and an east-west distance of 1770 km). Empty cells indicate no effort in the cell in question. Colours represent the mean of $\widehat{\widehat{D}_c}$ for that cell i.e. a mean based on the $\widehat{D}_c$ of all segments with centres within that cell.

Thus a comparison of the corrected observed densities with the point estimate prediction surface represents a visual “goodness of fit” test for the models. Note that the corrected density are averaged over the entire reporting period whereas the prediction surfaces are for one *Dayofyear* of one *Year* hence there may be discrepancies if, for example a high density region was only observed in a particular *Dayofyear/Year* combination not being predicted over/shown. Scales are the same for a particular species. Because the corrected observed densities are plotted on a 53 by 60 grid, the extent of spatial coverage of the prediction area is exaggerated. A more precise summary of the spatial coverage of effort can be obtained from Figures 8 - 12. N.B. The effort for harbour porpoise was less than for other species because of the sea state constraint.

JNCC has been supplied with gridded predictions for the entire JCP area allowing predictions of density for day 227 (in years 1994 – 2010) and days 45, 135, 227 and 315 for 2010. From these estimates abundances for any region (with associated confidence intervals) can be calculated.

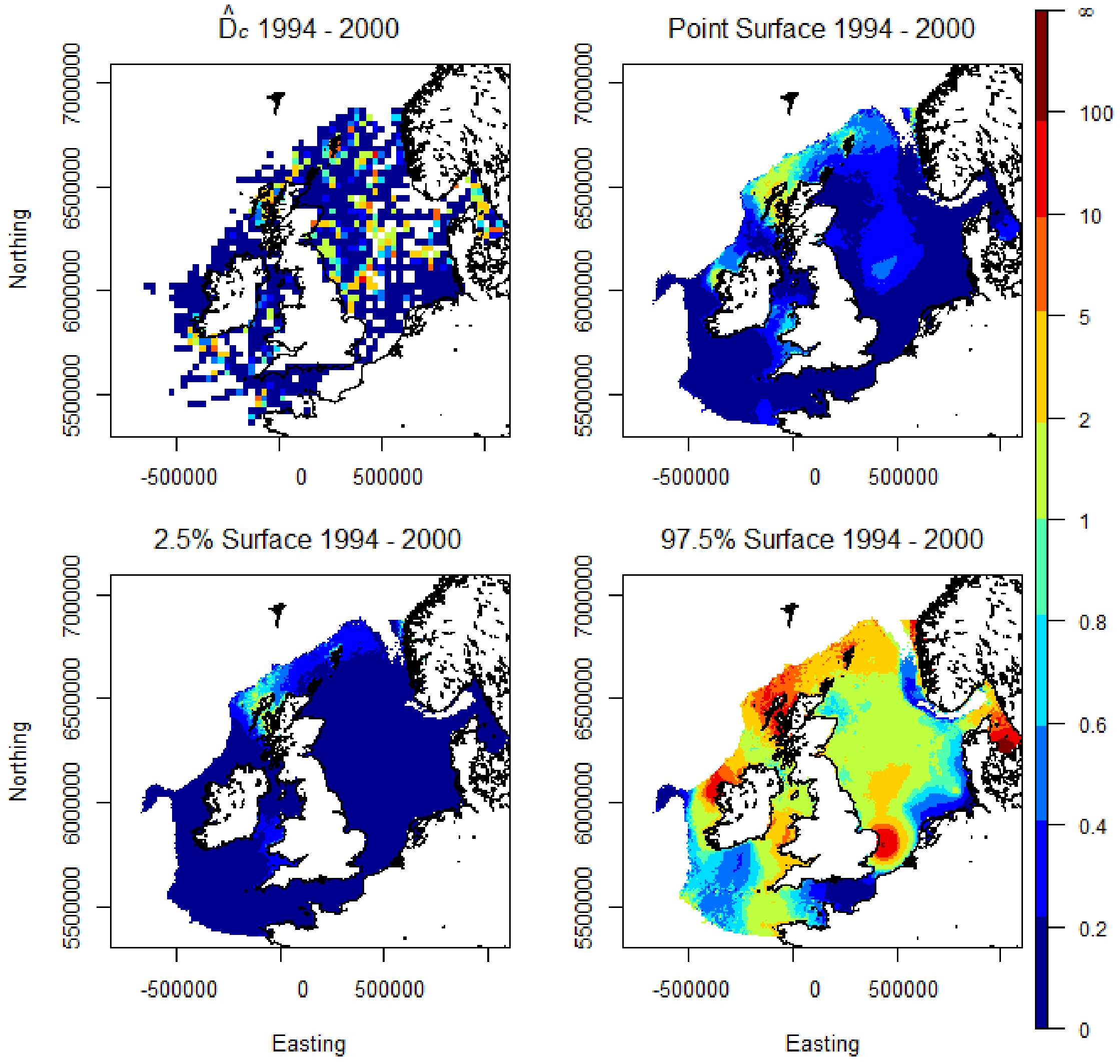

Figure A4.1. Harbour porpoise densities (animals/km^2) 1994 – 2000. Top left: mean $\widehat{D}_c$ summers 1994 – 2000 combined. Top right: predicted mean summer densities 1994 – 2000. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

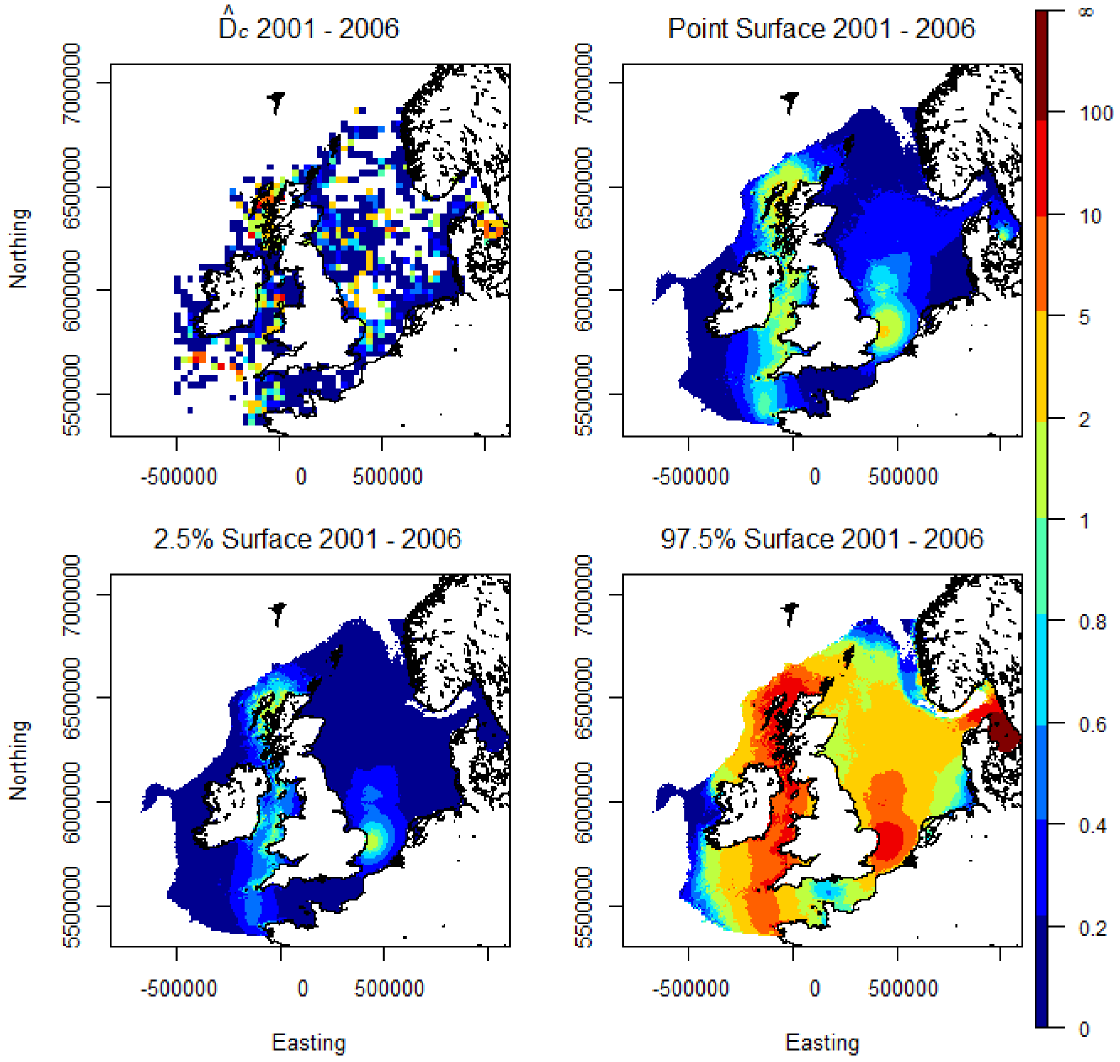

Figure A4.2. Harbour porpoise densities (animals/km^2) 2001 – 2006. Top left: mean $\hat{D}_c$ summers 2001 – 2006 combined. Top right: predicted mean summer densities 2001 – 2006. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

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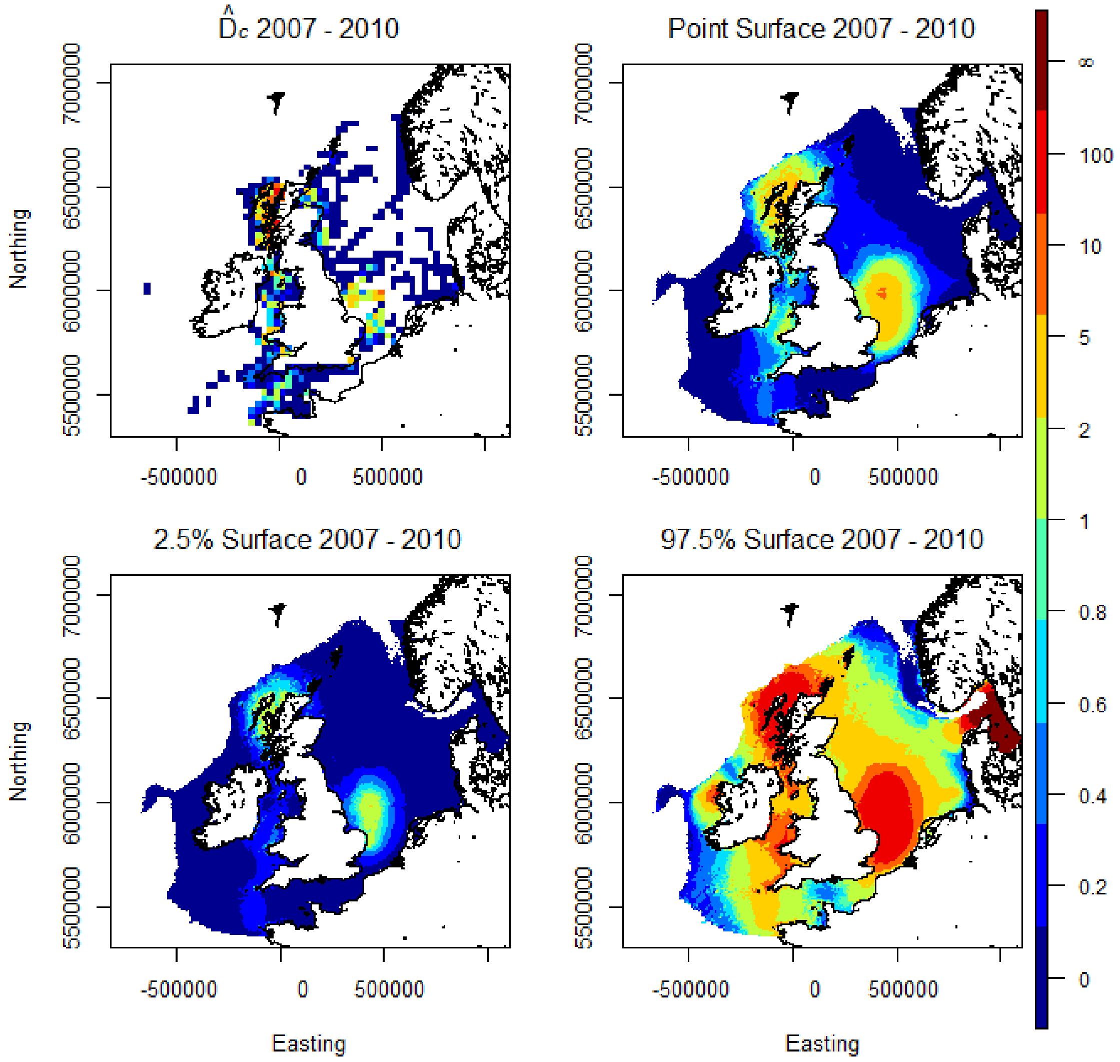

Figure A4.3. Harbour porpoise densities (animals/km^2) 2007 – 2010. Top left: mean summers 2007 – 2010 combined. Top right: predicted mean summer densities 2007 – 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

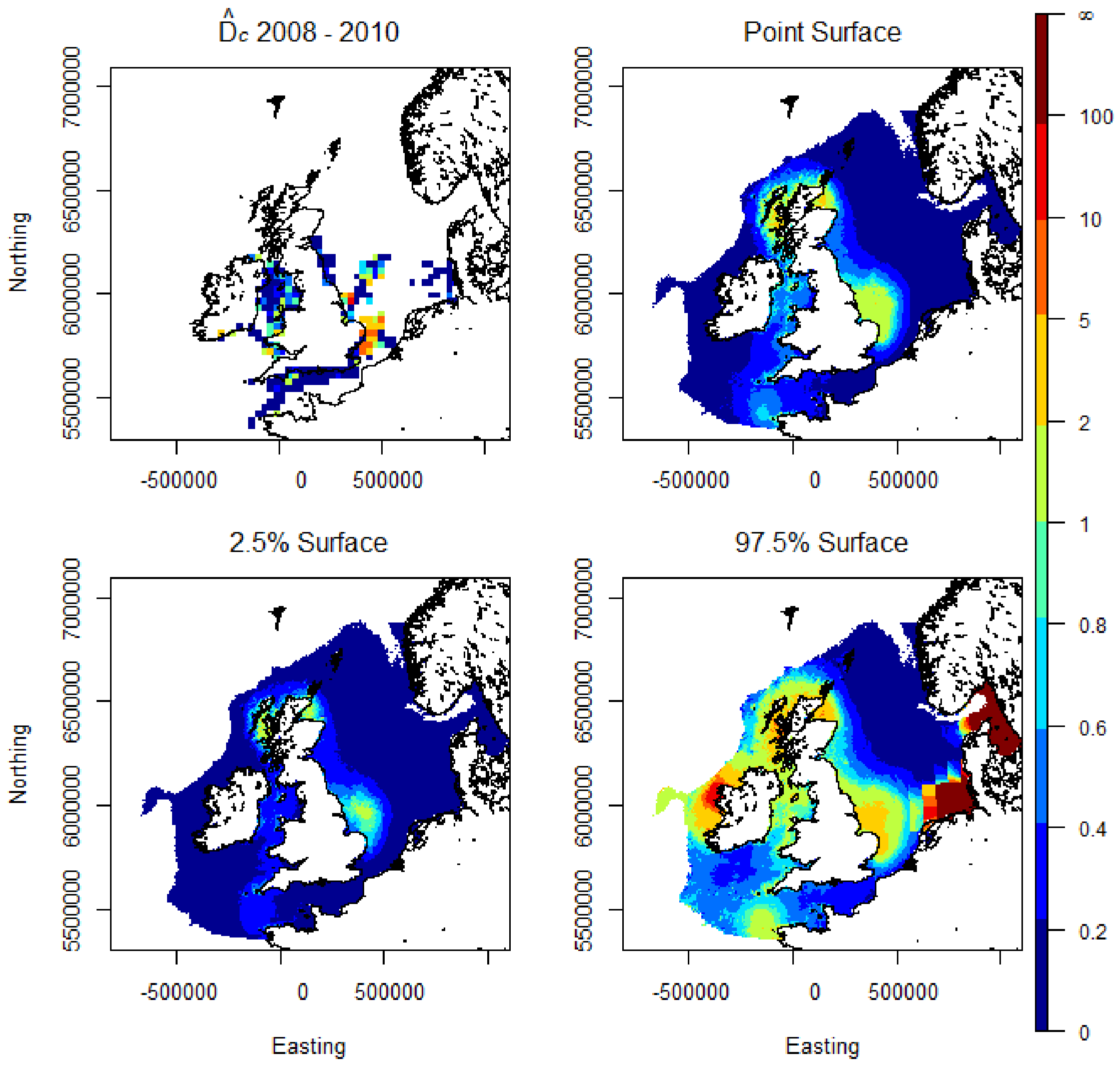

Figure A4.4. Harbour porpoise densities (animals/km^2) winter 2010. Top left: mean $\widehat{D}_c$ all winters 2008 – 2010 to show data being drawn upon for this prediction with a temporal interaction. Top right: predicted densities winter 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

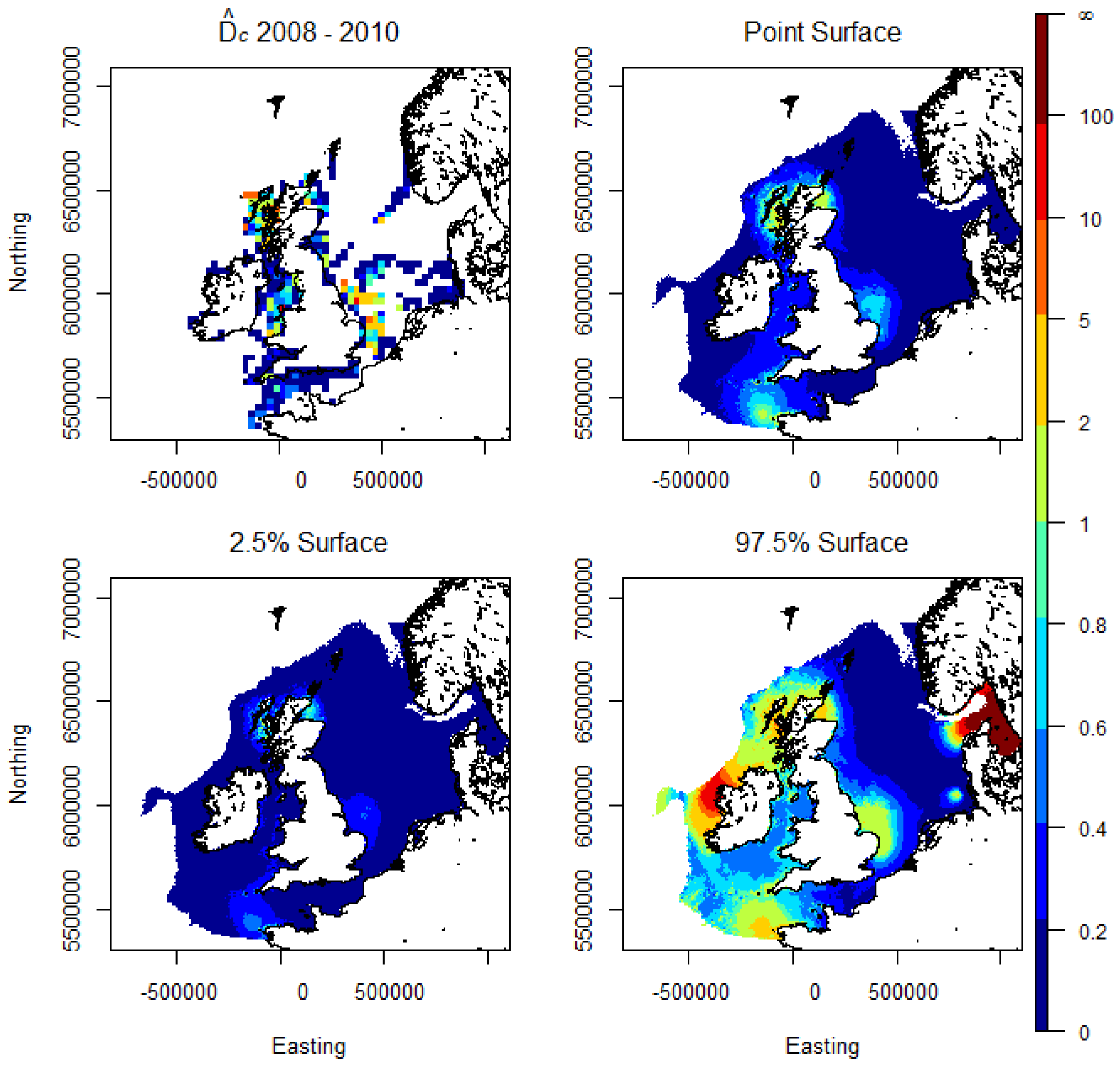

Figure A4.5. Harbour porpoise densities (animals/km^2) spring 2010. Top left: mean $\widehat{D}_c$ all springs 2008 – 2010 to show data being drawn upon for this prediction with a temporal interaction. Top right: predicted densities spring 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

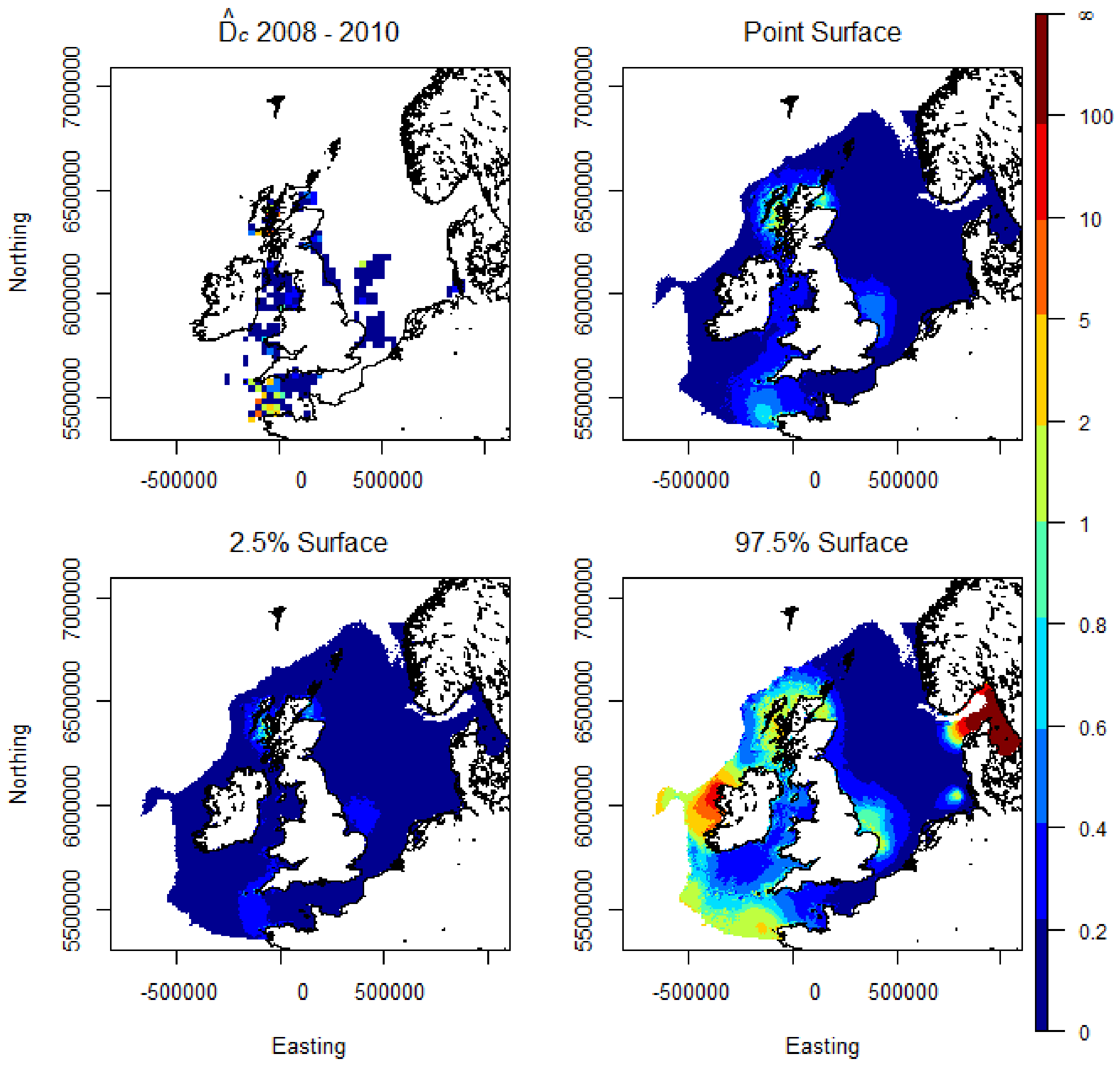

Figure A4.6. Harbour porpoise densities (animals/km^2) autumn 2010. Top left: mean $\hat{D}_c$ all autumns 2008 – 2010 to show data being drawn upon for this prediction with a temporal interaction.. Top right: predicted densities autumn 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

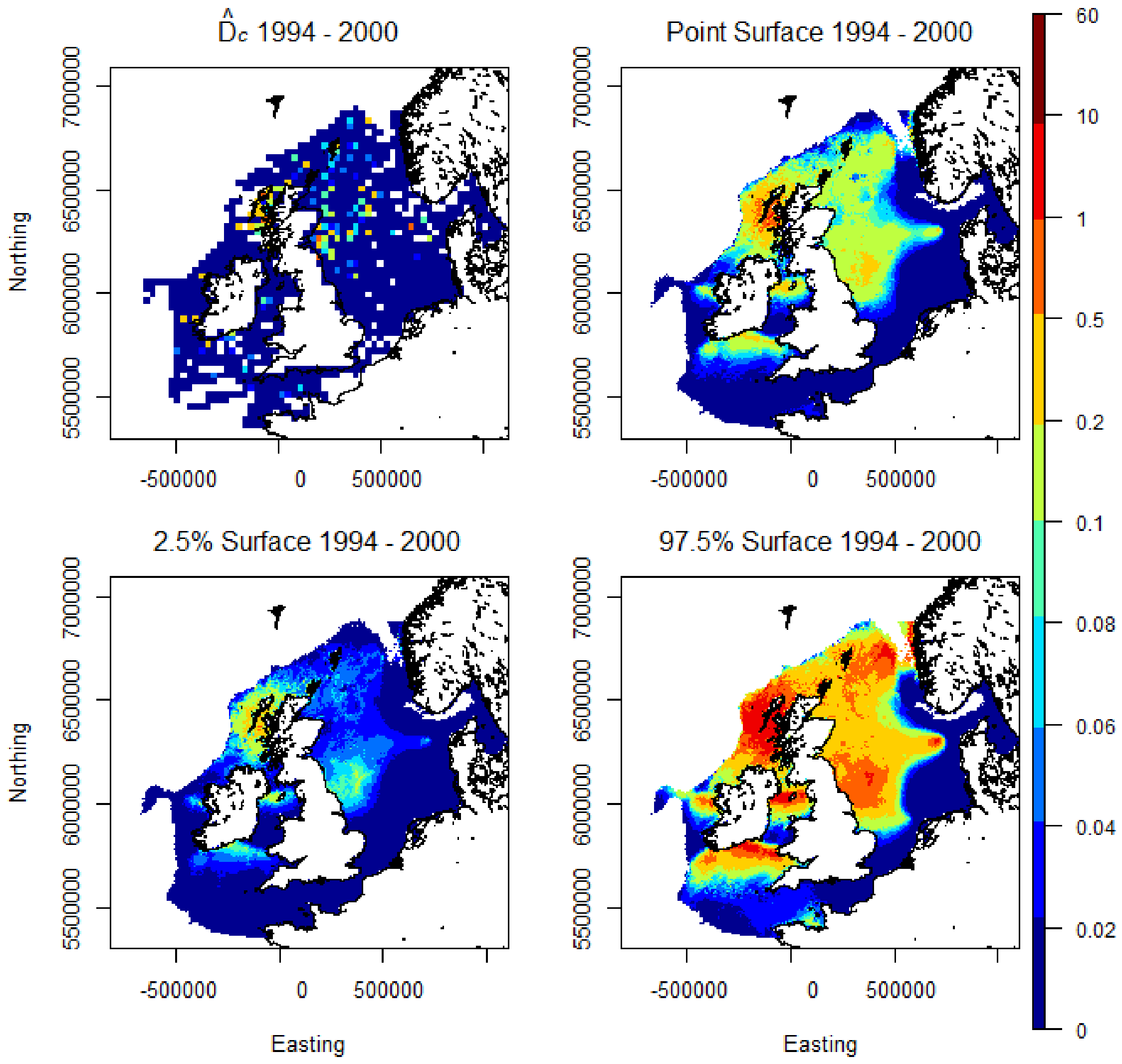

Figure A4.7. Minke whales densities (animals/km^2) 1994 – 2000. Top left: mean $\widehat{D}_c$ summers 1994 – 2000 combined. Top right: predicted mean summer densities 1994 – 2000. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

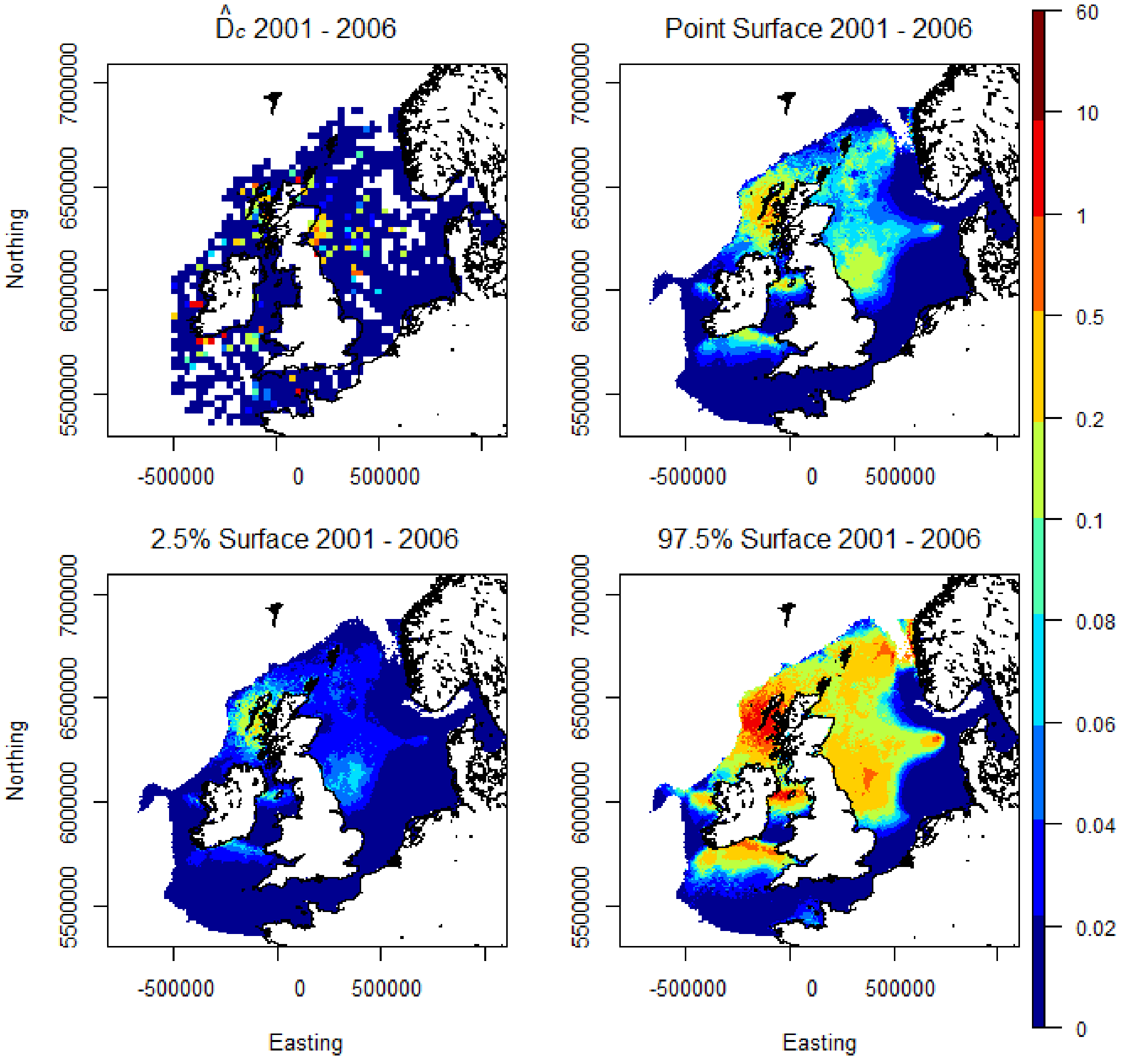

Figure A4.8. Minke whales densities (animals/km^2) 2001 – 2006. Top left: mean $\widehat{D}_c$ summers 2001 – 2006 combined. Top right: predicted mean summer densities 2001 – 2006. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

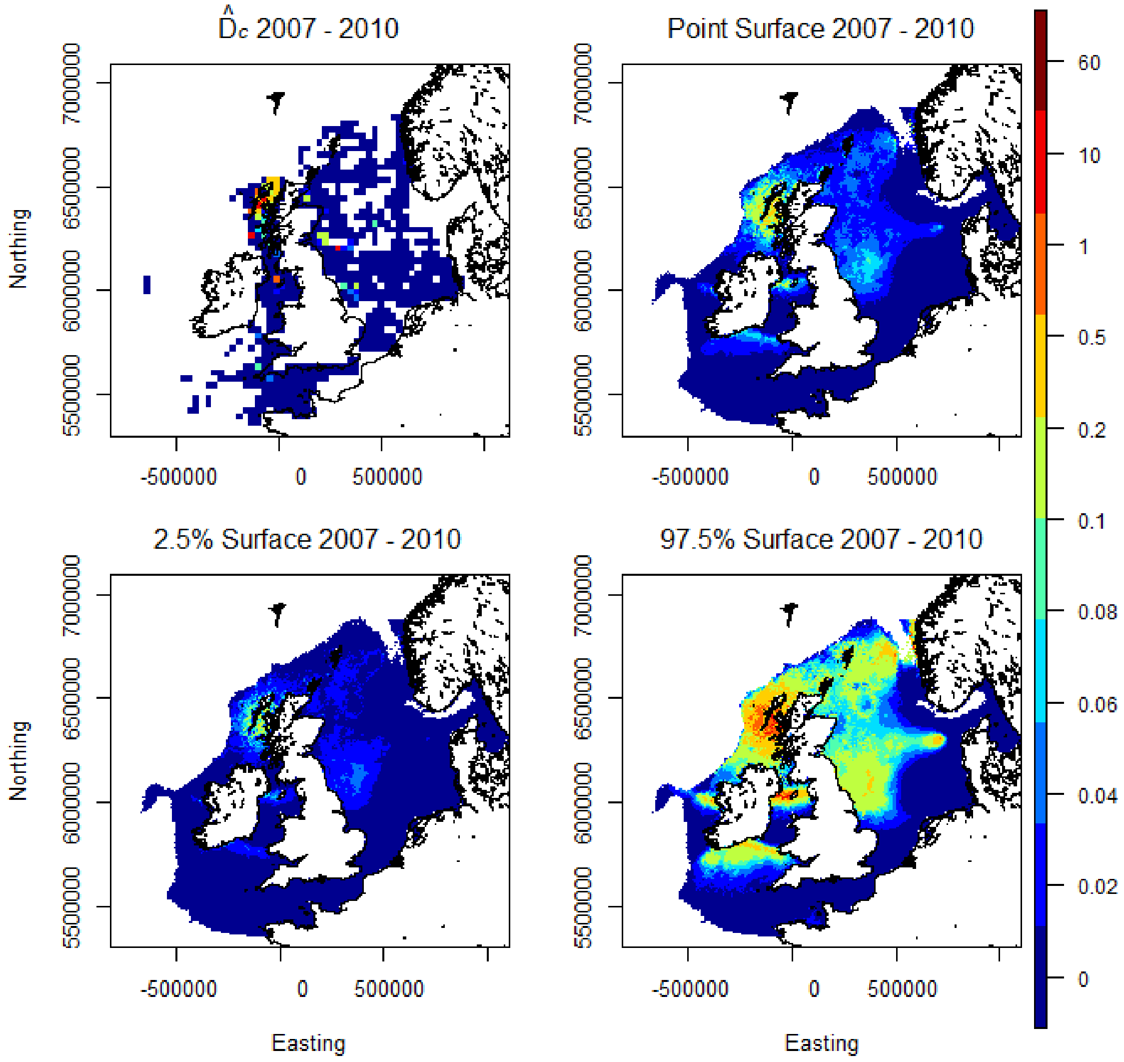

Figure A4.9. Minke whales densities (animals/km^2) 2007 – 2010. Top left: mean $\widehat{D}_c$ summers 2007 – 2010 combined. Top right: predicted mean summer densities 2007 – 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

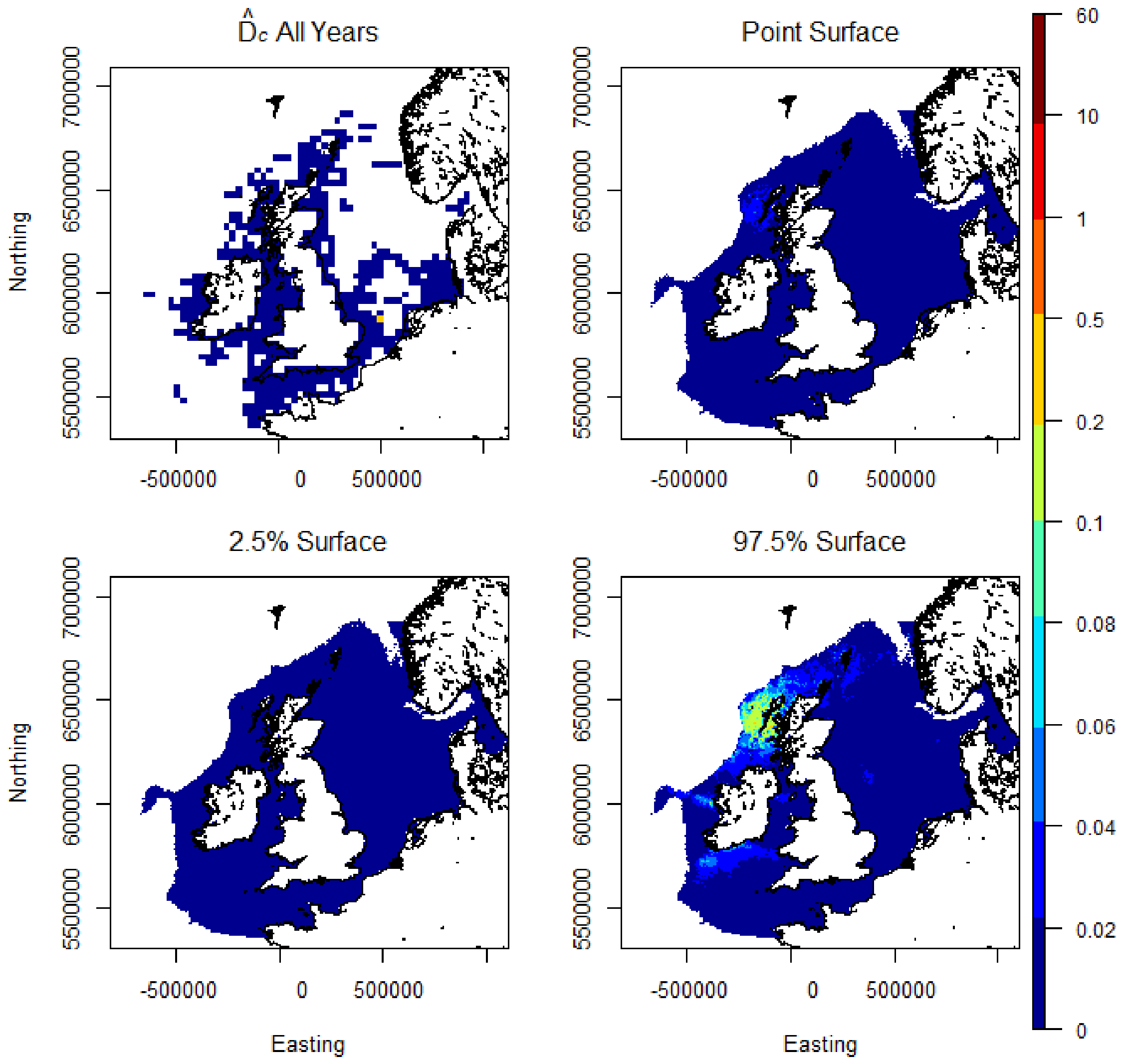

Figure A4.10. Minke whales densities (animals/km^2) winter 2010. Top left: mean $\widehat{D}_c$ winters all years. Top right: predicted densities winter 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

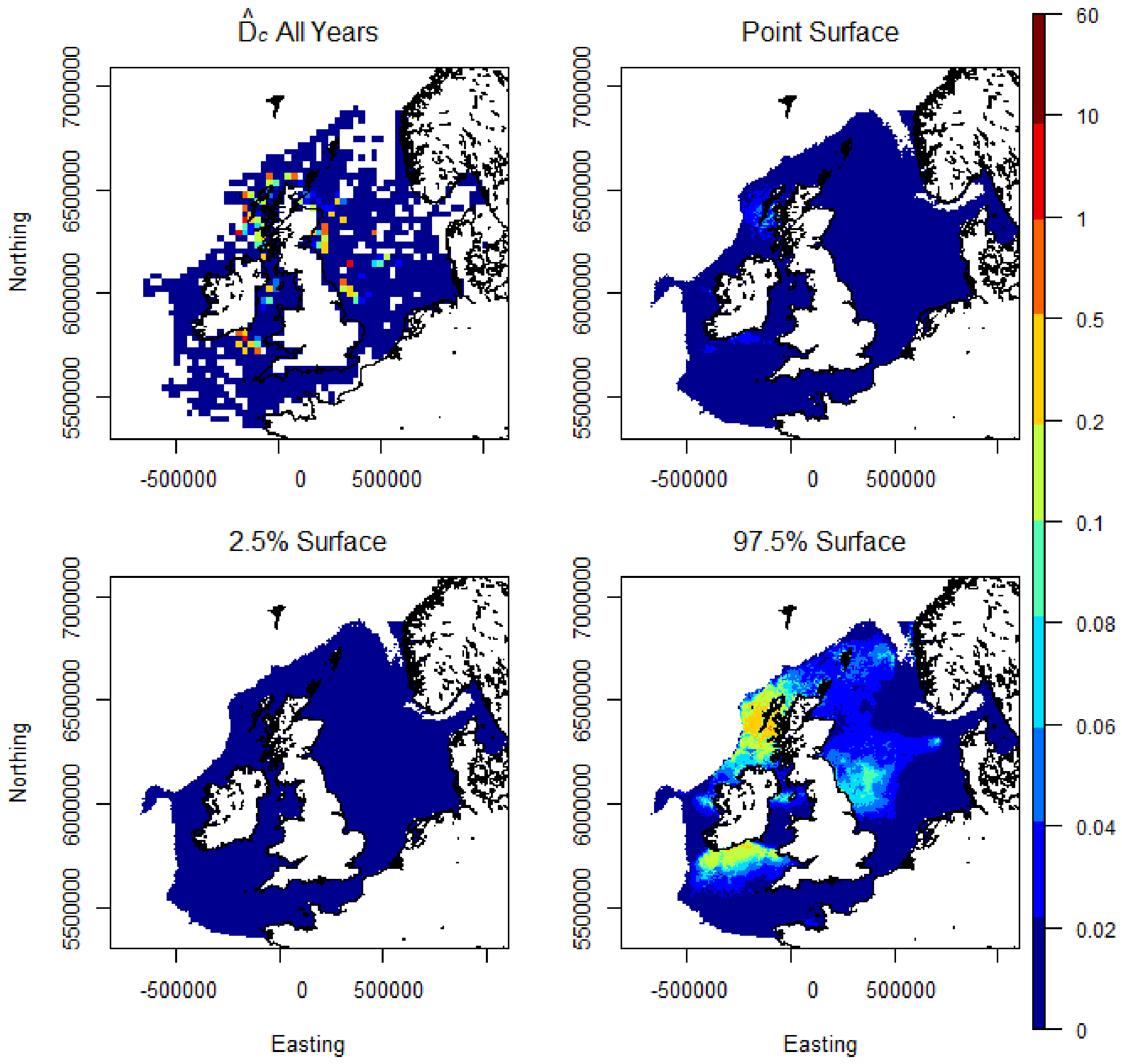

Figure A4.11. Minke whales densities (animals/km^2) spring 2010. Top left: mean $\widehat{D}_c$ spring all years. Top right: predicted densities summer 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

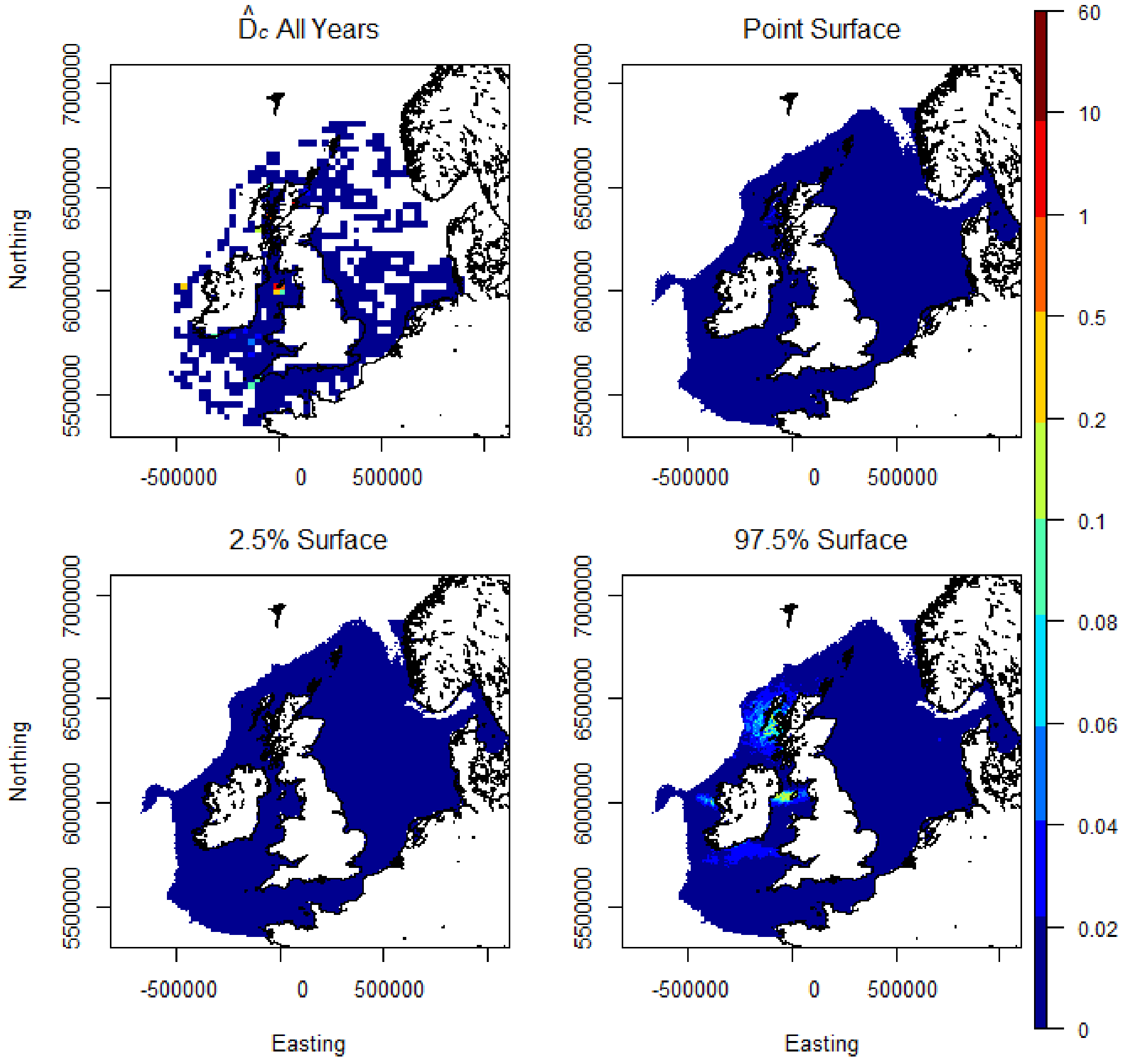

Figure A4.12. Minke whales densities (animals/km^2) autumn 2010. Top left: mean $\widehat{D}_c$ autumn all years. Top right: predicted densities autumn 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

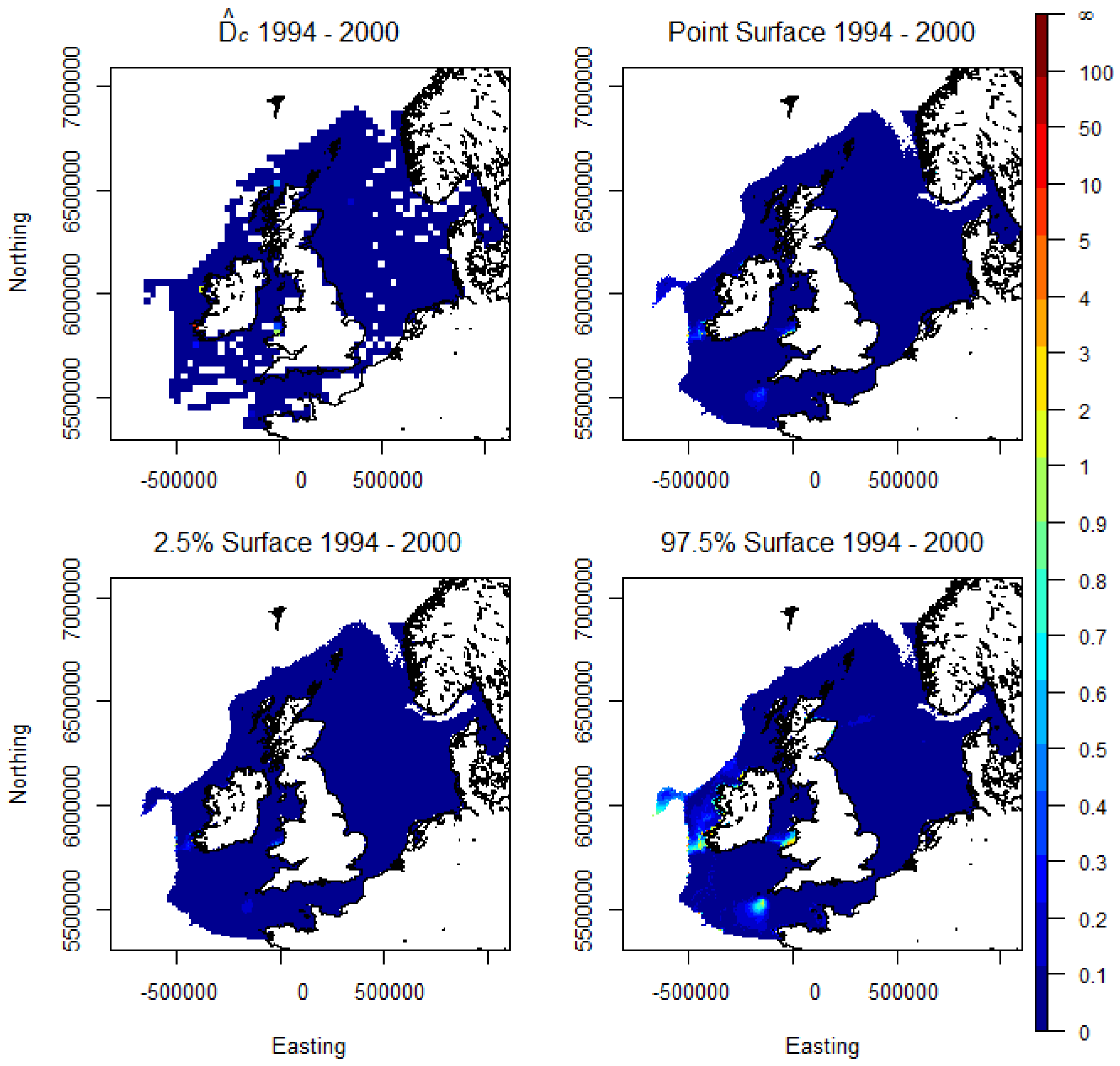

Figure A4.13. Bottlenose dolphins densities (animals/km^2) 1994 – 2000. Top left: mean $\widehat{D}_c$ summers 1994 – 2000 combined. Top right: predicted mean summer densities 1994 – 2000. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

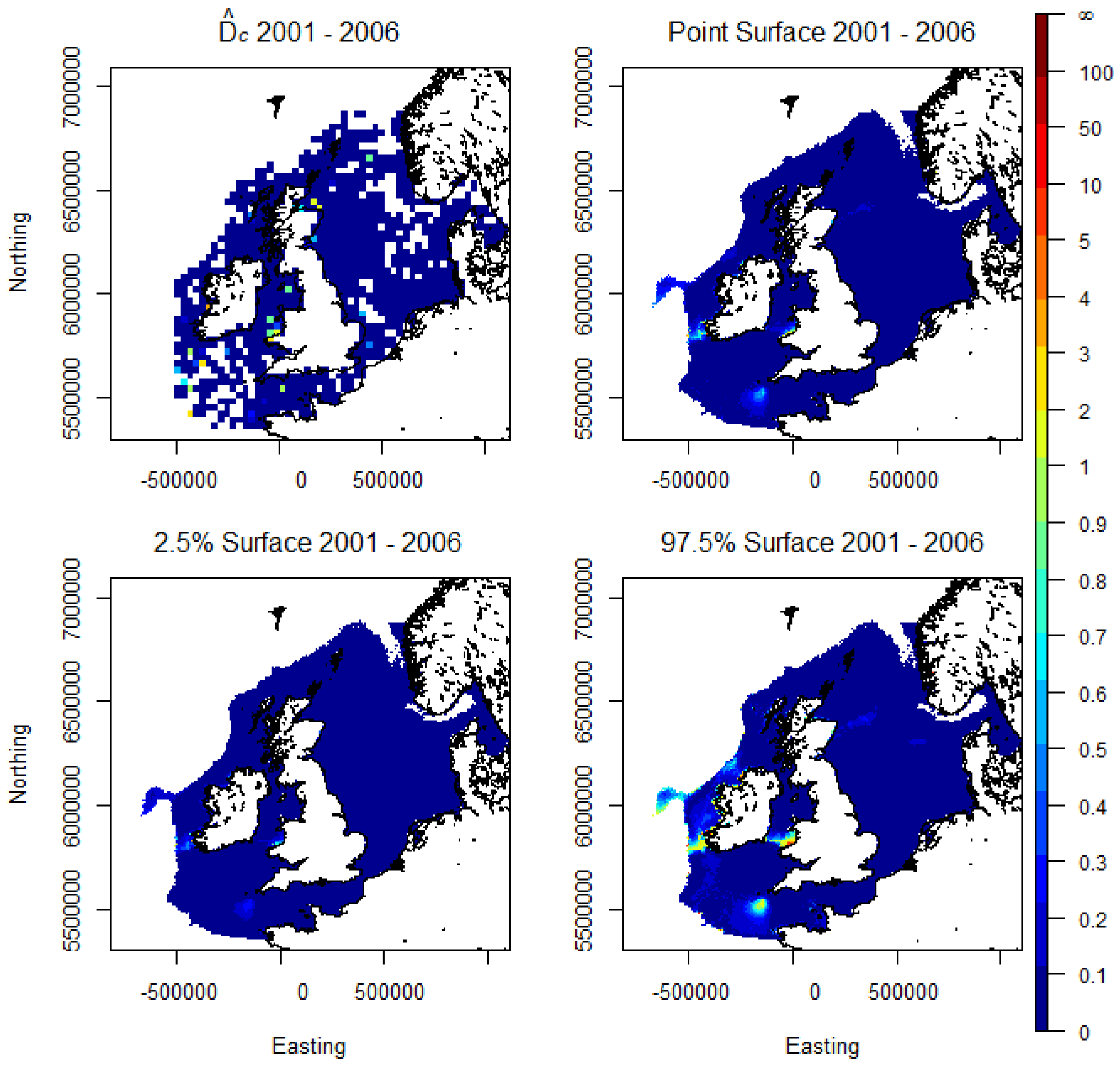

Figure A4.14. Bottlenose dolphin densities (animals/km^2) 2001 – 2006. Top left: mean $\widehat{D}_c$ summers 2001 – 2006 combined. Top right: predicted mean summer densities 2001 – 2006. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities, bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

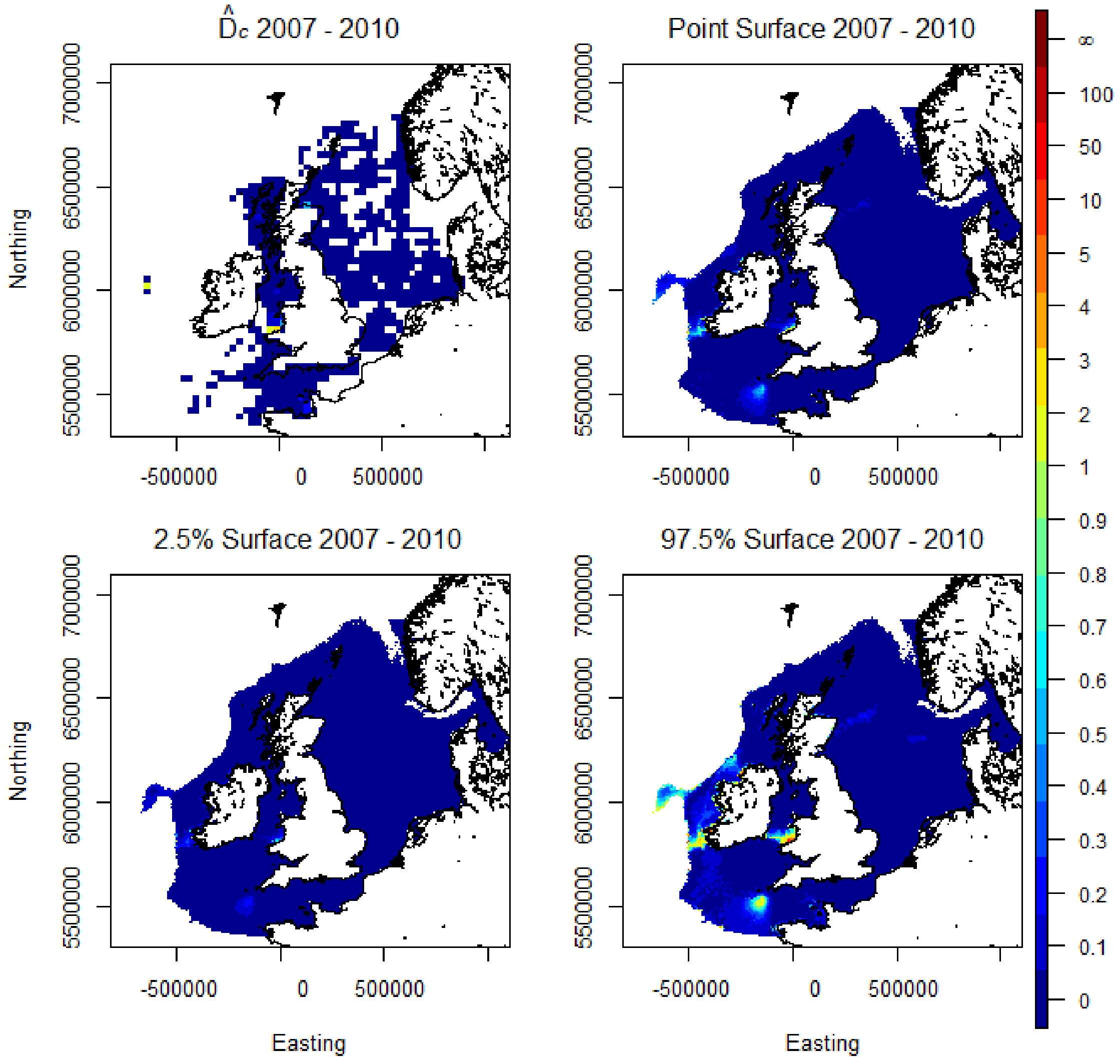

Figure A4.15. Bottlenose dolphins densities (animals/km^2) 2007 – 2010. Top left: mean $\widehat{D}_c$ summers 2007 – 2010 combined. Top right: predicted mean summer densities 2007 – 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

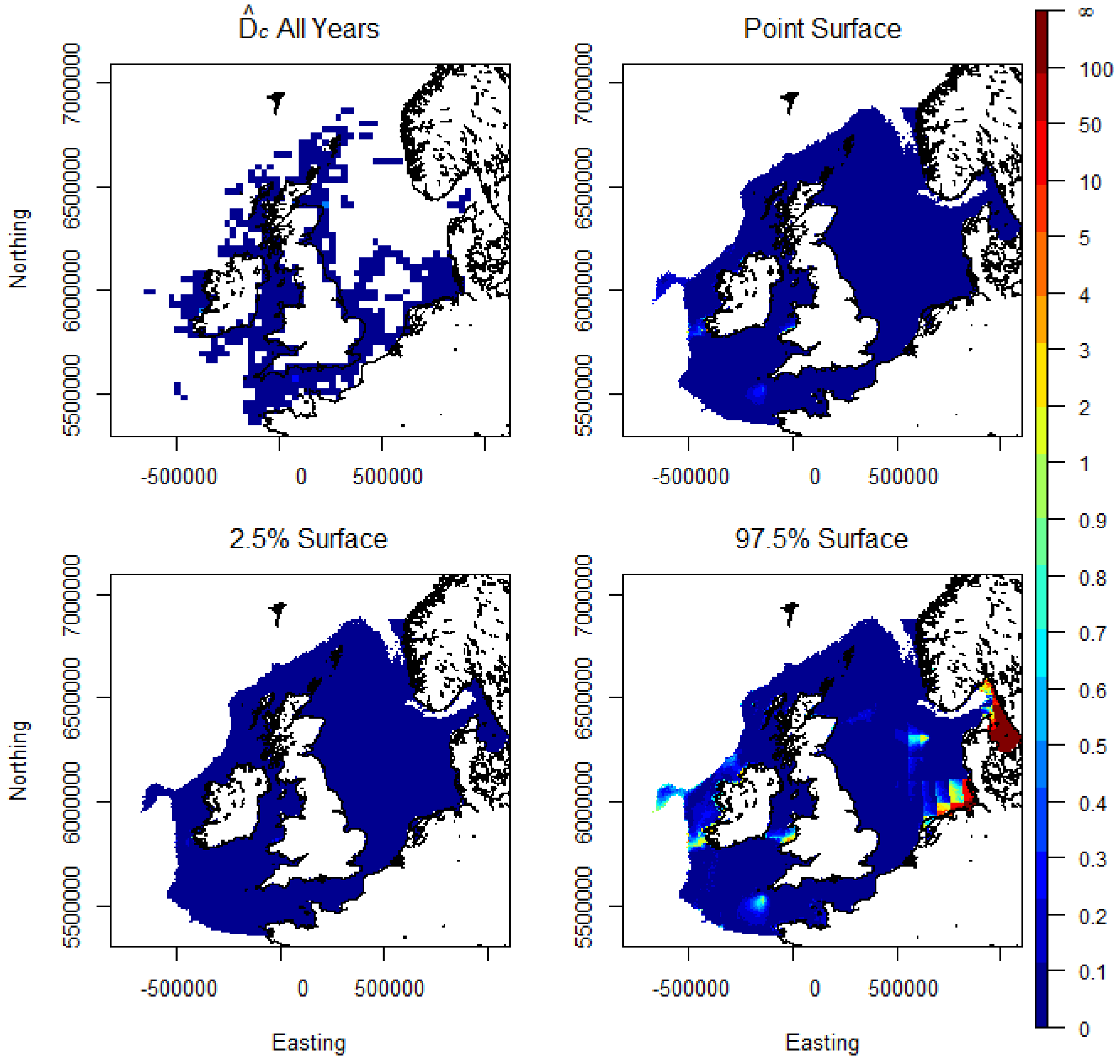

Figure A4.16. Bottlenose dolphins winter densities (animals/km^2) 2010. Top left: mean $\widehat{D}_c$ winter all years . Top right: predicted densities winter 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

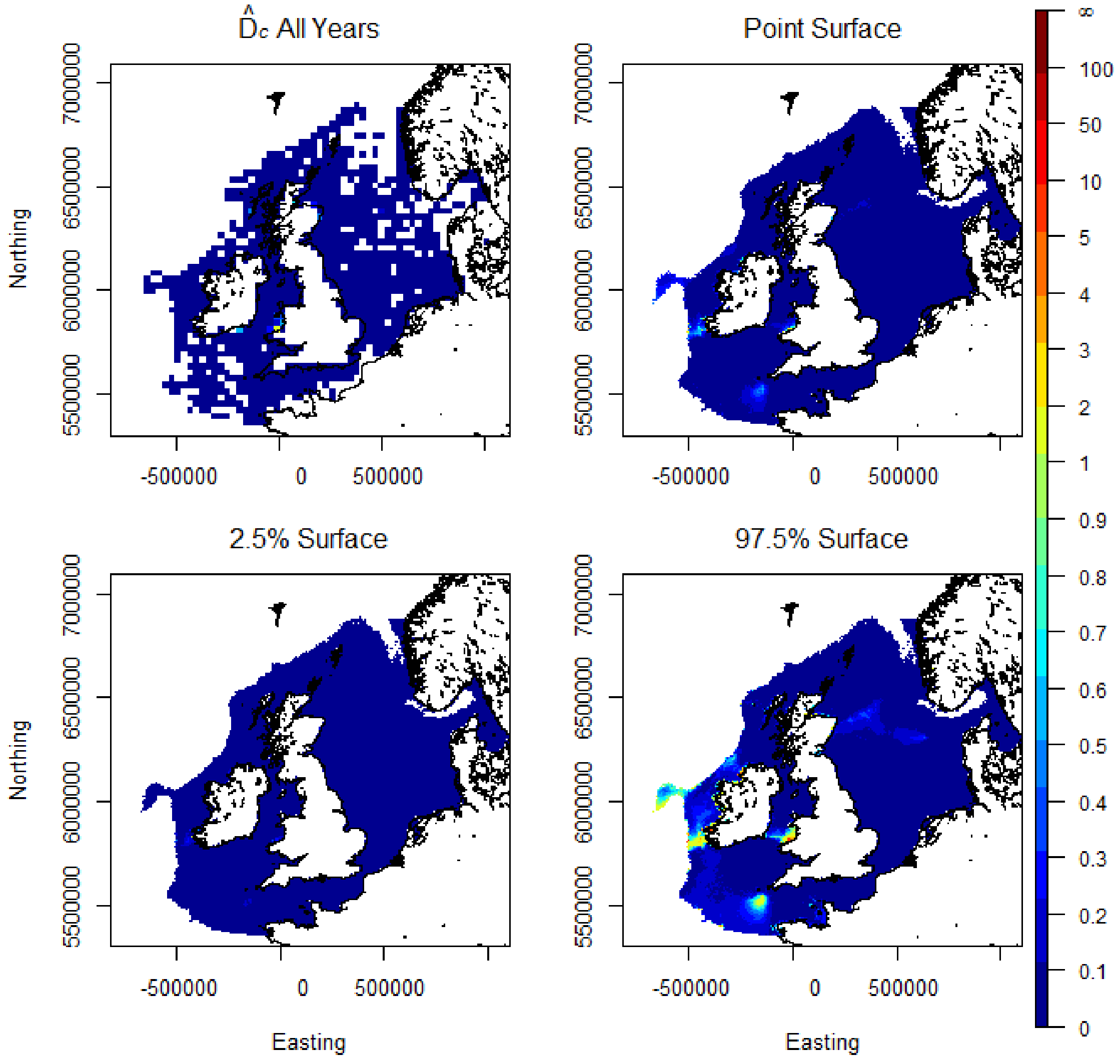

Figure A4.17. Bottlenose dolphins densities (animals/km^2) Spring 2010. Top left: mean $\widehat{D}_c$ spring all years. Top right: predicted densities spring 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

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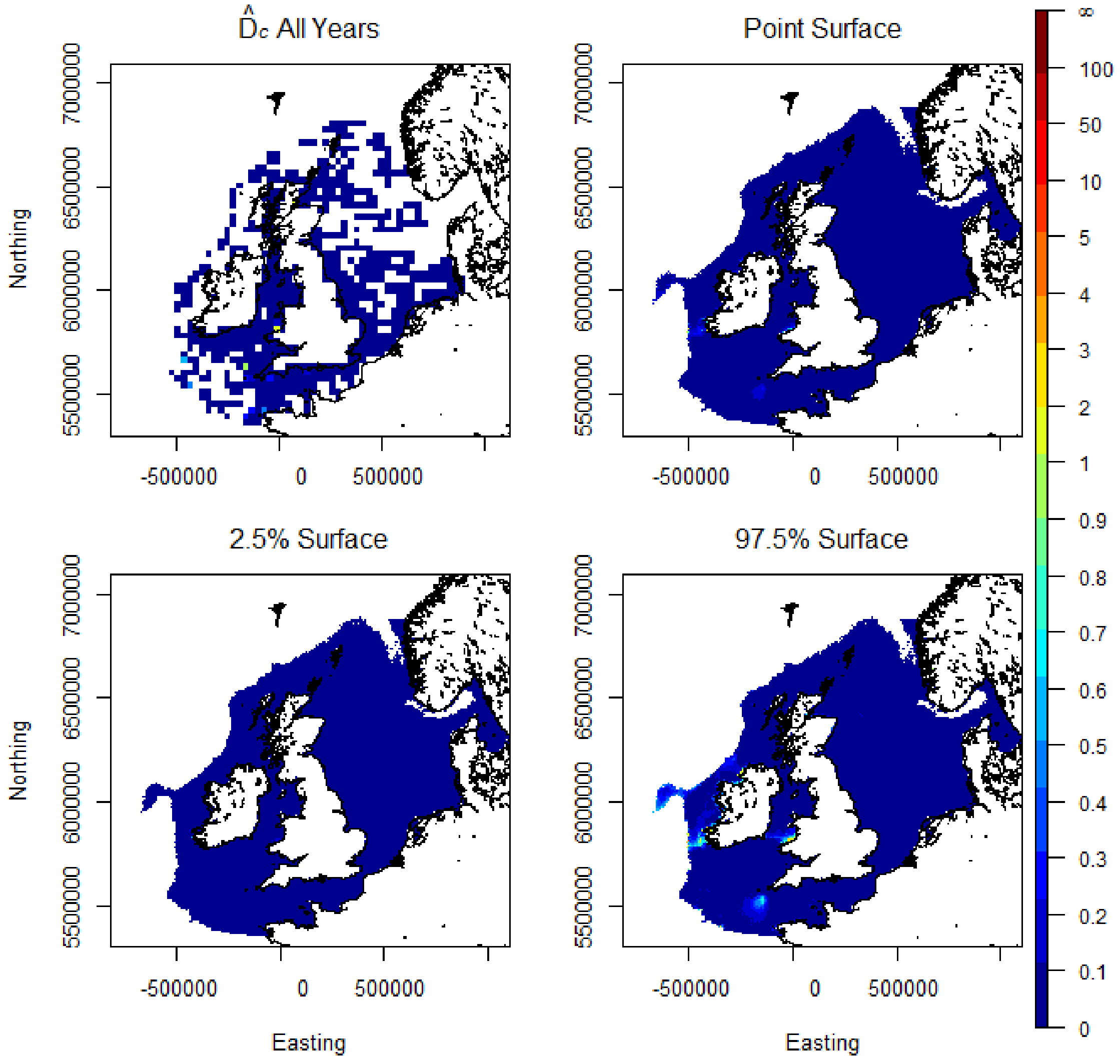

Figure A4.18. Bottlenose dolphins densities (animals/km^2) autumn 2010. Top left: mean $\widehat{D}_c$ autumn all years. Top right: predicted densities autumn 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

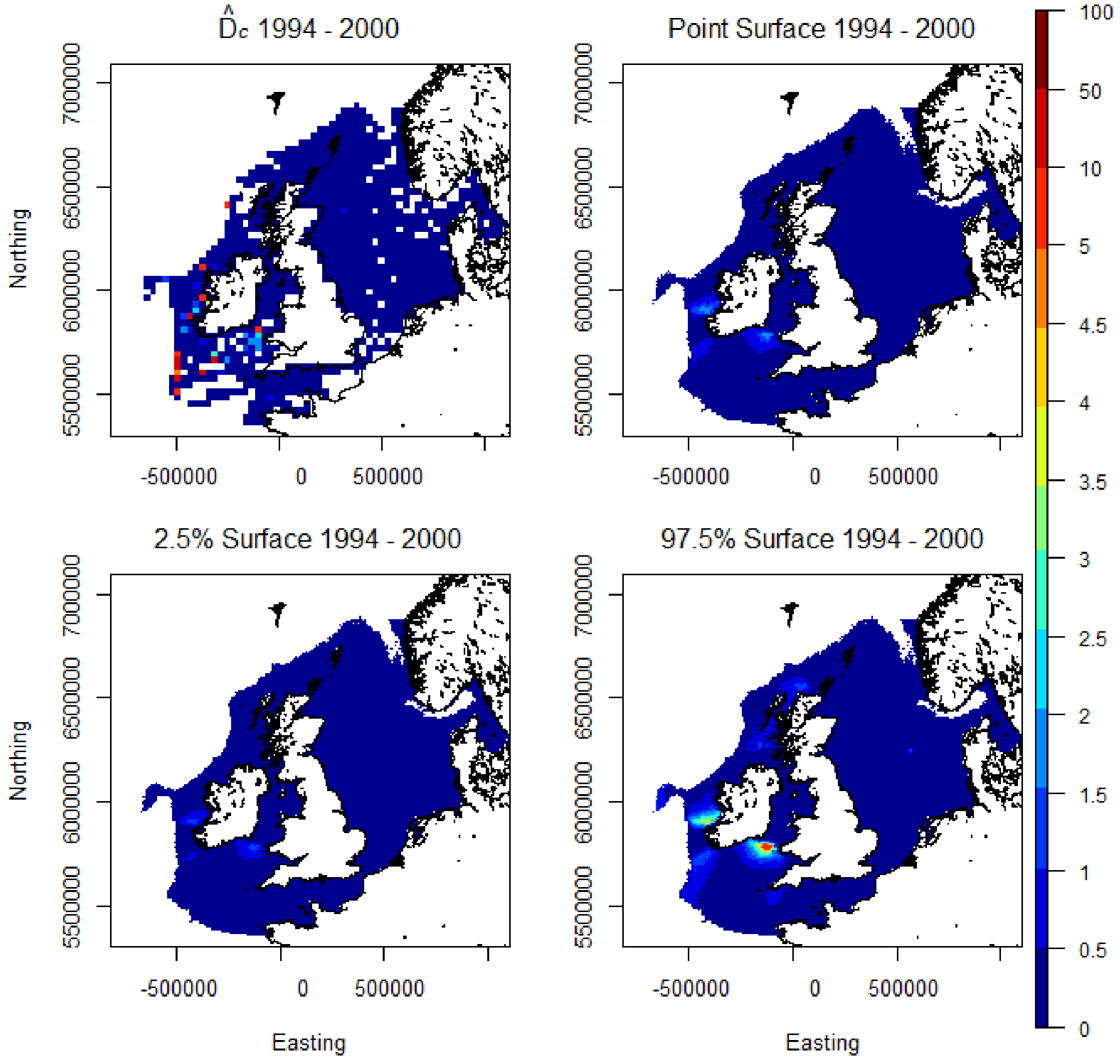

Figure A4.19. Short-beaked common dolphins densities (animals/km^2) 1994 – 2000. Top left: mean $\hat{D}_c$ summers 1994 – 2000 combined. Top right: predicted mean summer densities 1994 – 2000. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

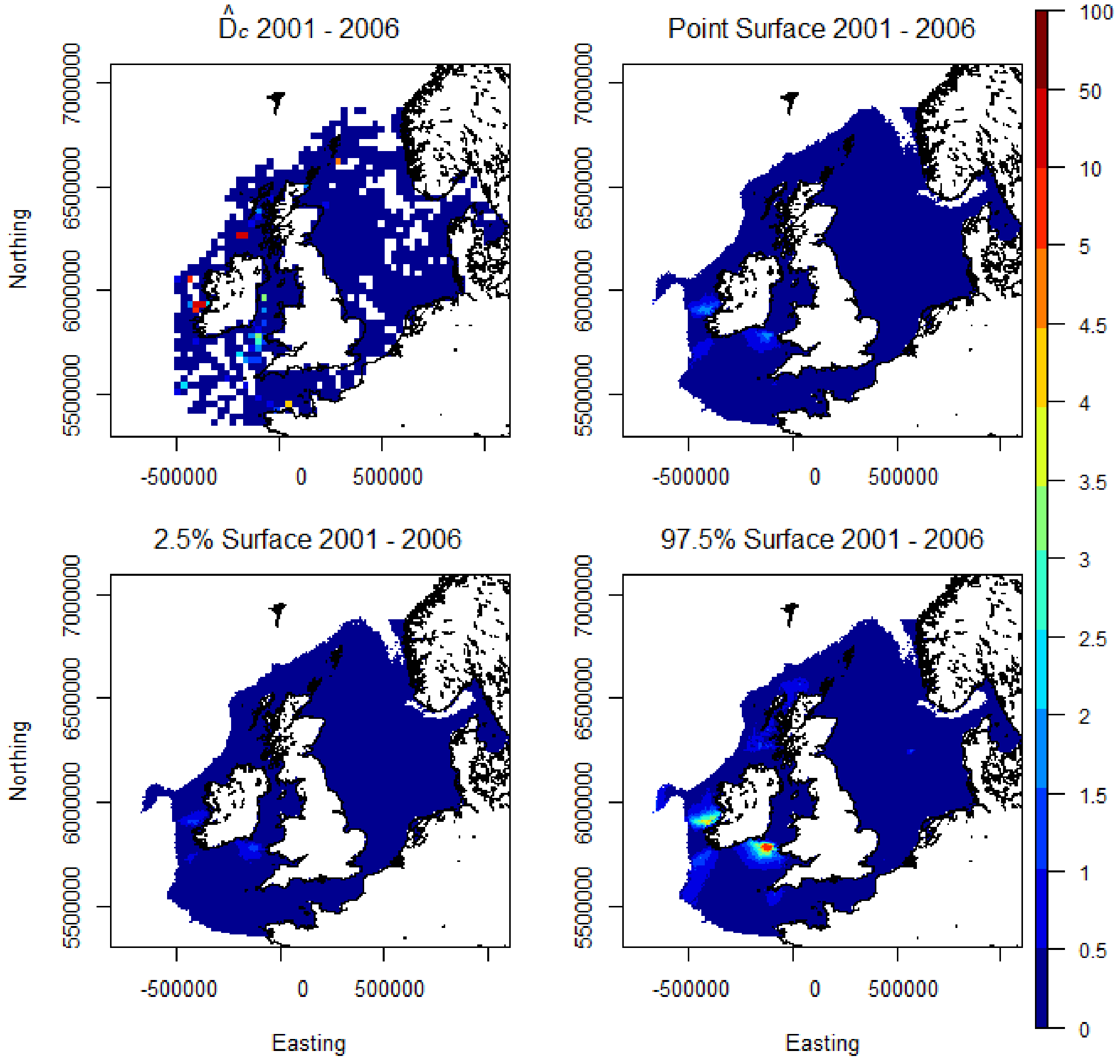

Figure A4.20. Short-beaked common dolphins densities (animals/km^2) 2001 – 2006. Top left: mean $\widehat{D}_c$ summers 2001 – 2006 combined. Top right: predicted mean summer densities 2001 – 2006. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

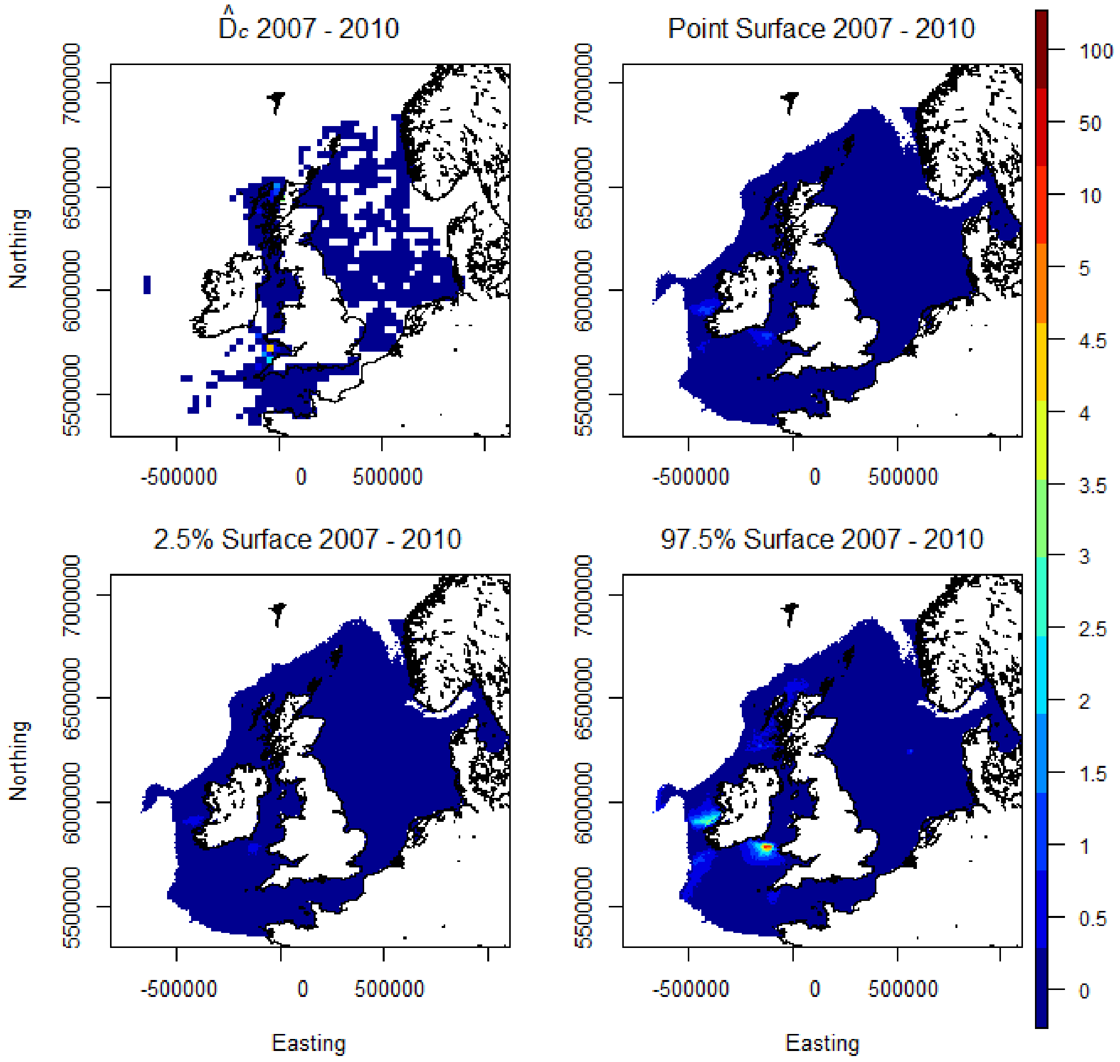

Figure A4.21. Short-beaked common dolphins densities (animals/km^2) 2007 – 2010. Top left: mean $\widehat{D}_c$ summers 2007 – 2010 combined. Top right: predicted mean summer densities 2007 – 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

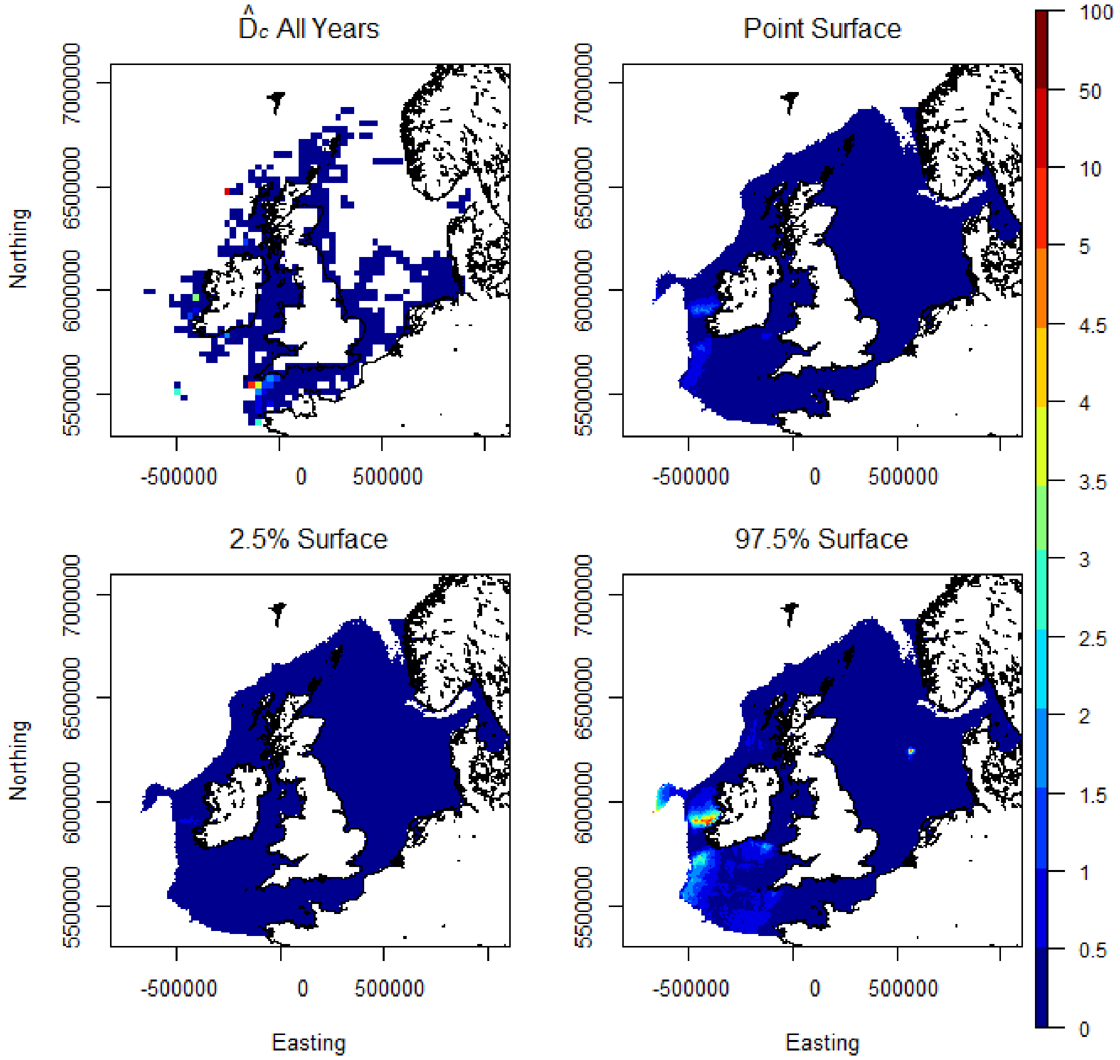

Figure A4.22. Short-beaked common dolphins densities (animals/km^2) winter 2010. Top left: mean $\widehat{D}_c$ winter all years. Top right: predicted densities winter 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

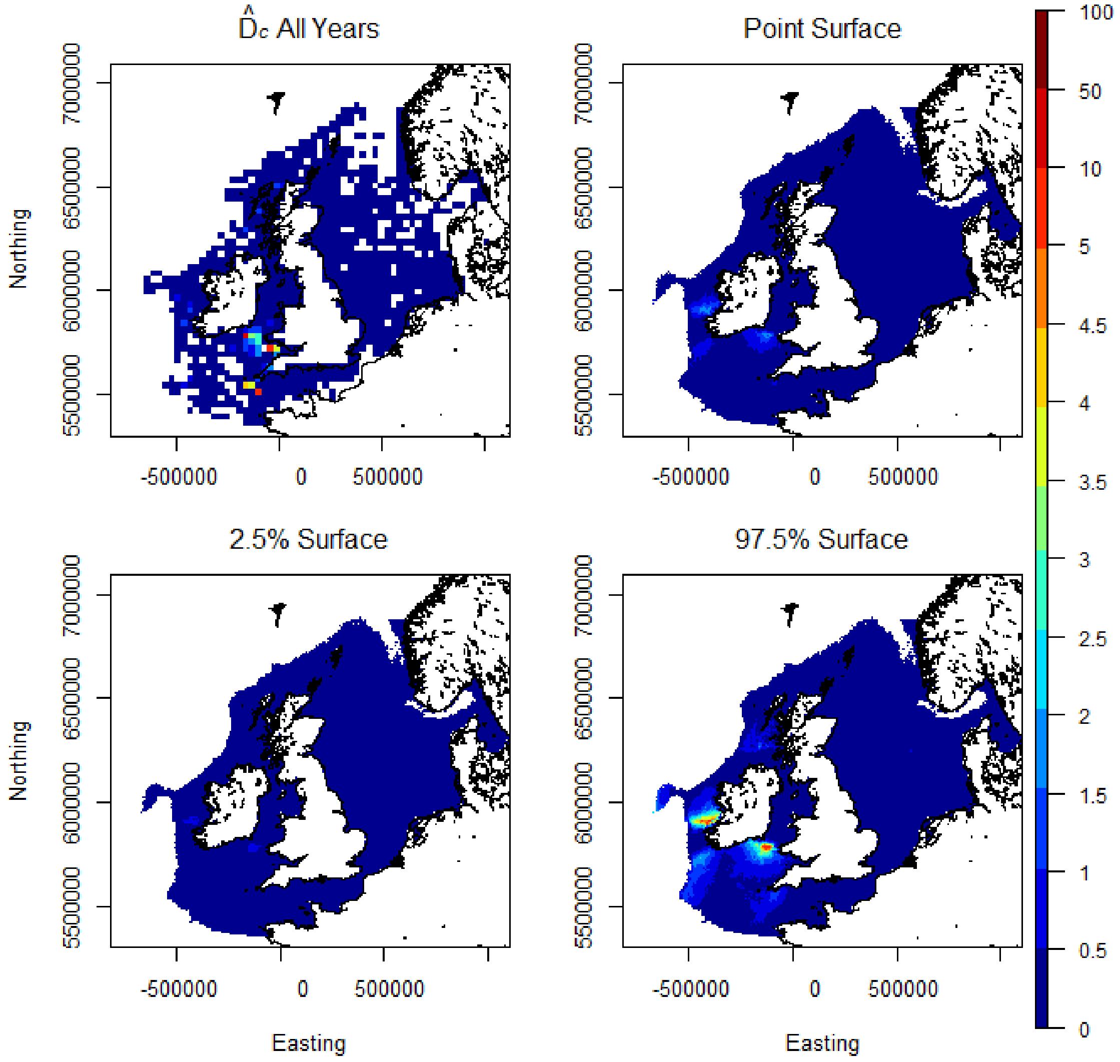

Figure A4.23. Short-beaked common dolphins densities (animals/km^2) spring 2010. Top left: mean $\widehat{D}_c$ spring all years. Top right: predicted densities spring 2010, bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

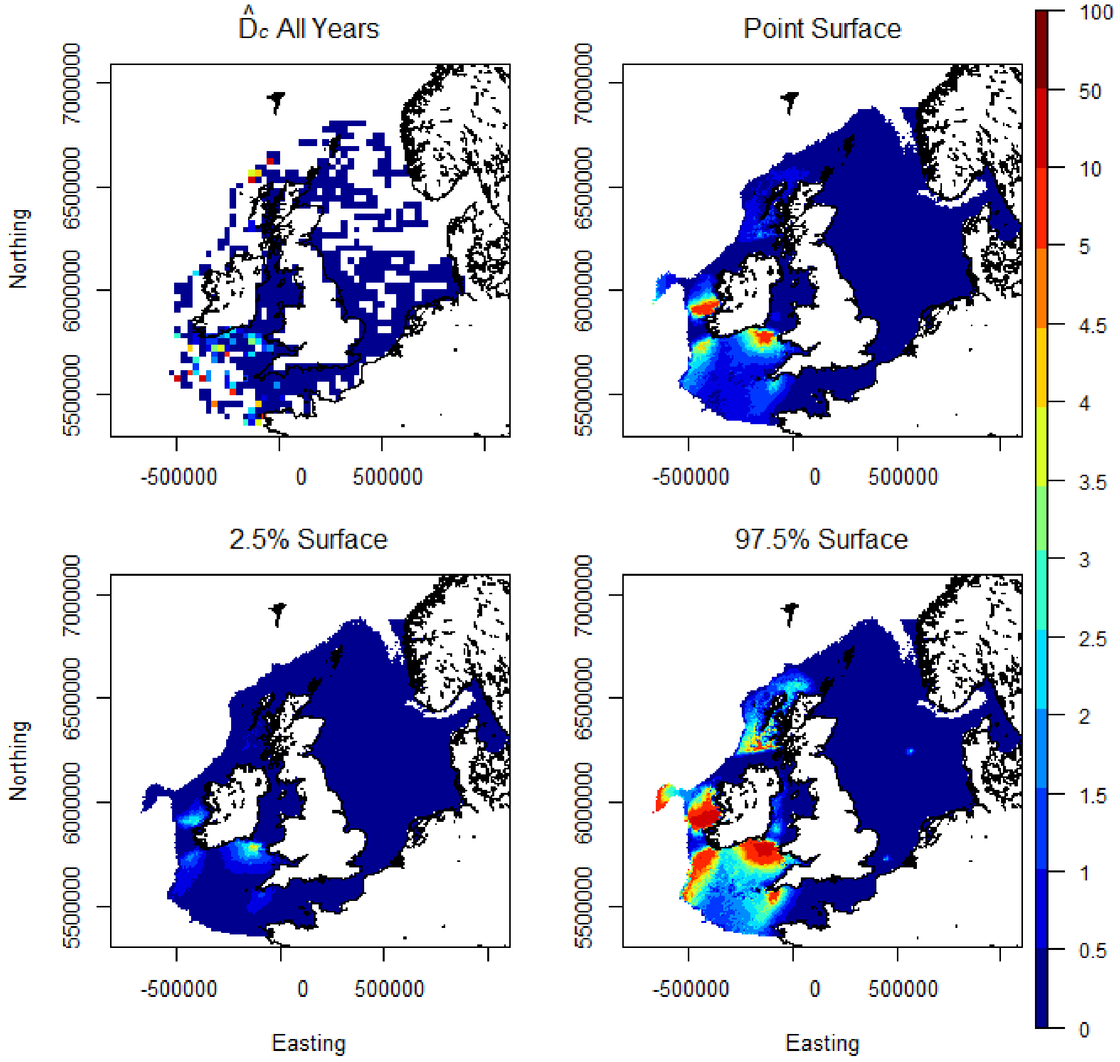

Figure A4.24. Short-beaked common dolphins densities (animals/km^2) autumn 2010. Top left: mean $\widehat{D}_c$ autumn all years. Top right: predicted densities autumn 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

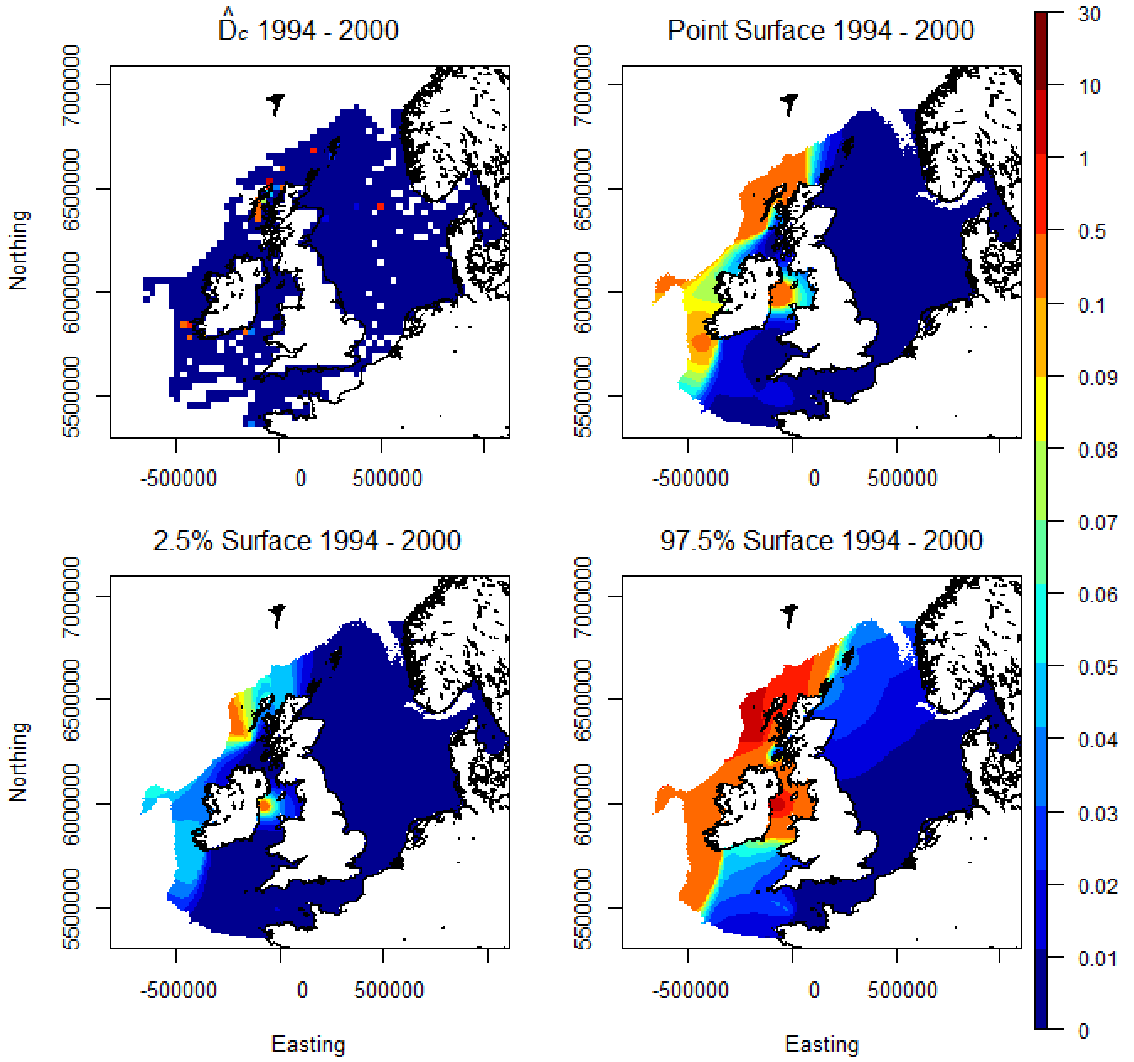

Figure A4.25. Risso's dolphins densities (animals/km^2) 1994 – 2000. Top left: mean $\widehat{D}_c$ summers 1994 – 2000 combined. Top right: predicted mean summer densities 1994 – 2000. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

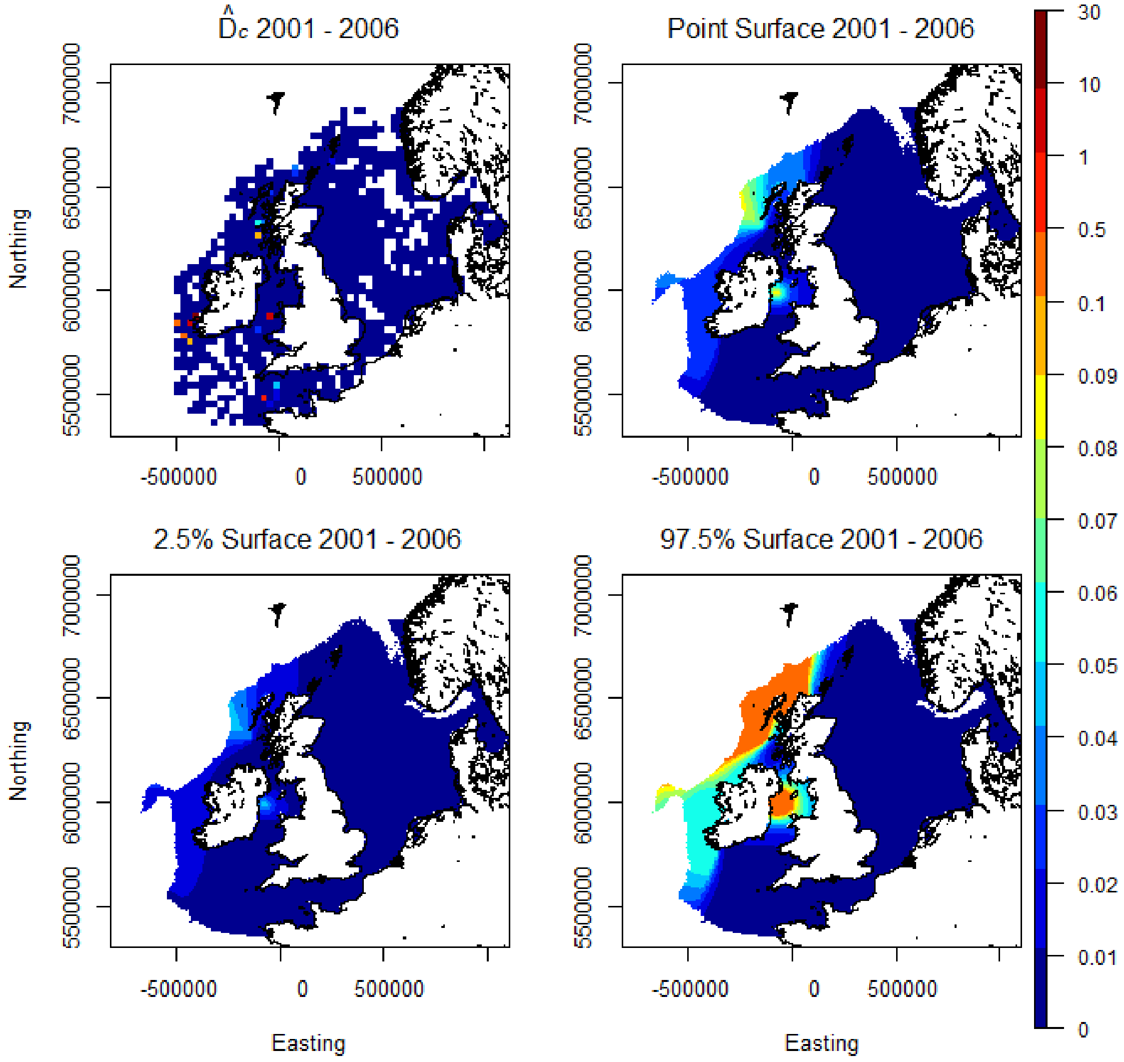

Figure A4.26. Risso's dolphins densities (animals/km^2) 2001 – 2006. Top left: mean $\widehat{D}_c$ summers 2001 – 2006 combined. Top right: predicted mean summer densities 2001 – 2006. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

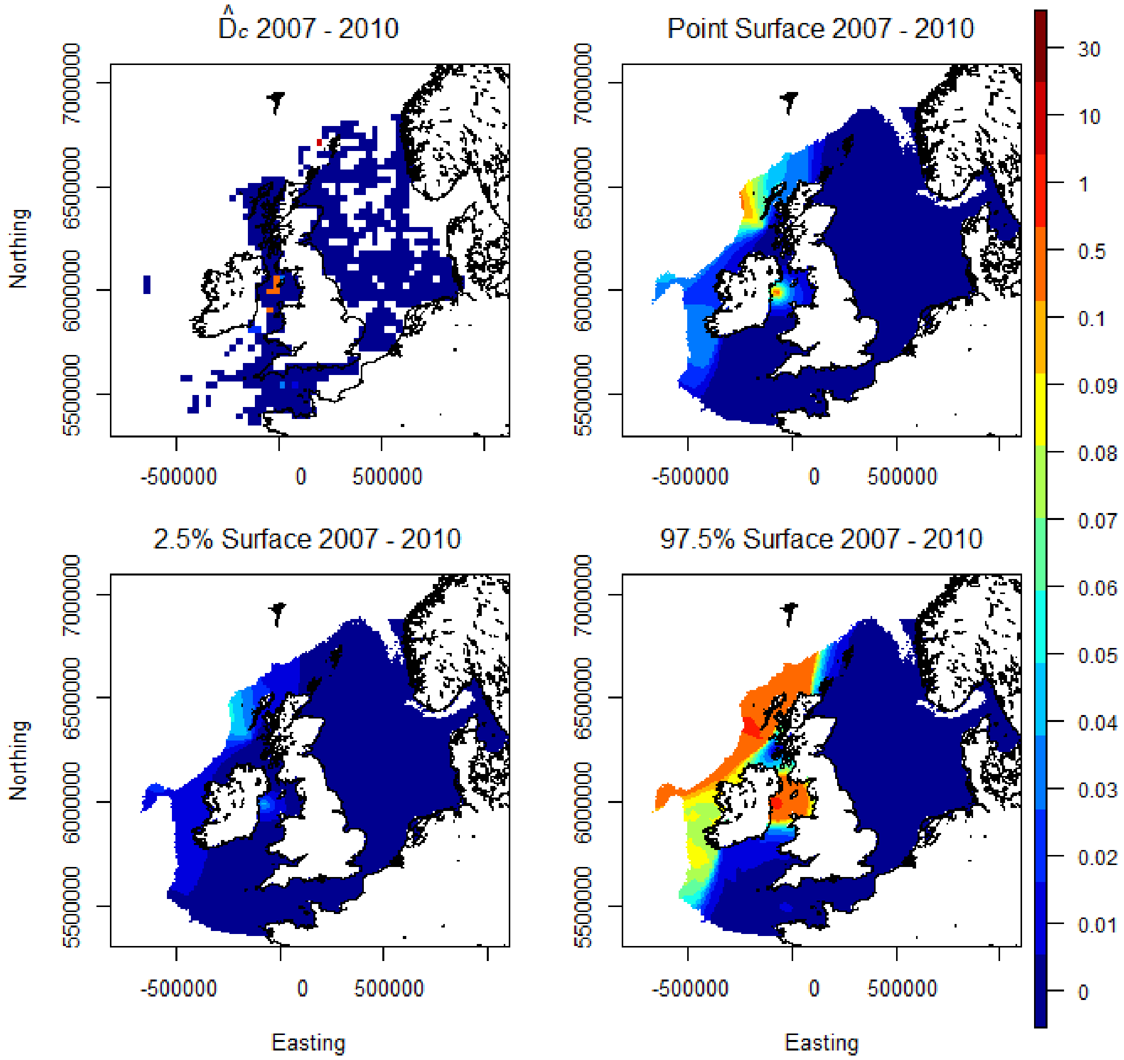

Figure A4.27. Risso's dolphins densities (animals/km^2) 2007 – 2010. Top left: mean $\widehat{D}_c$ summers 2007 – 2010 combined. Top right: predicted mean summer densities 2007 – 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

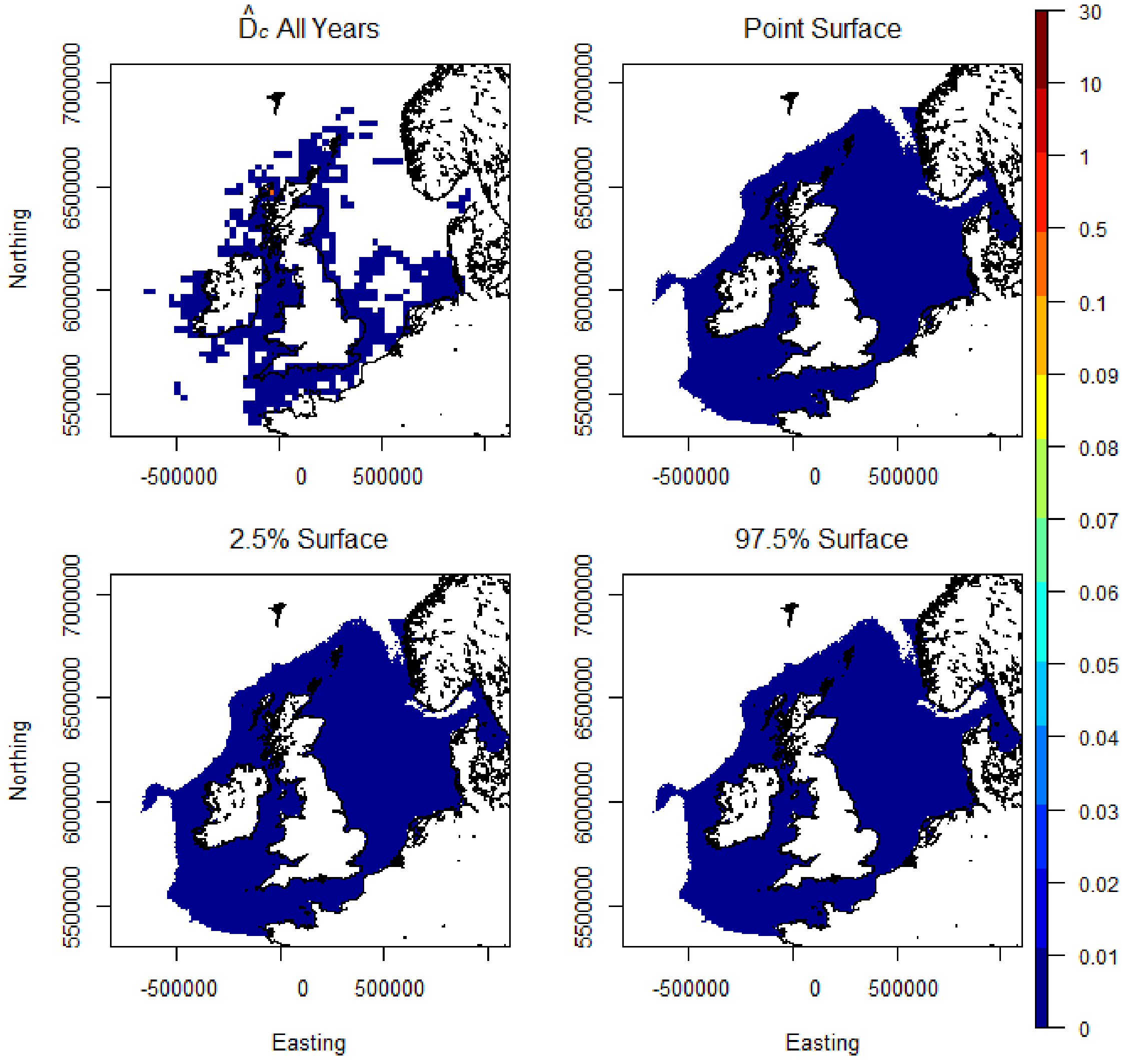

Figure A4.28. Risso's dolphins densities (animals/km^2) winter 2010. Top left: mean $\widehat{D}_c$ all winters combined. Top right: predicted densities winter 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

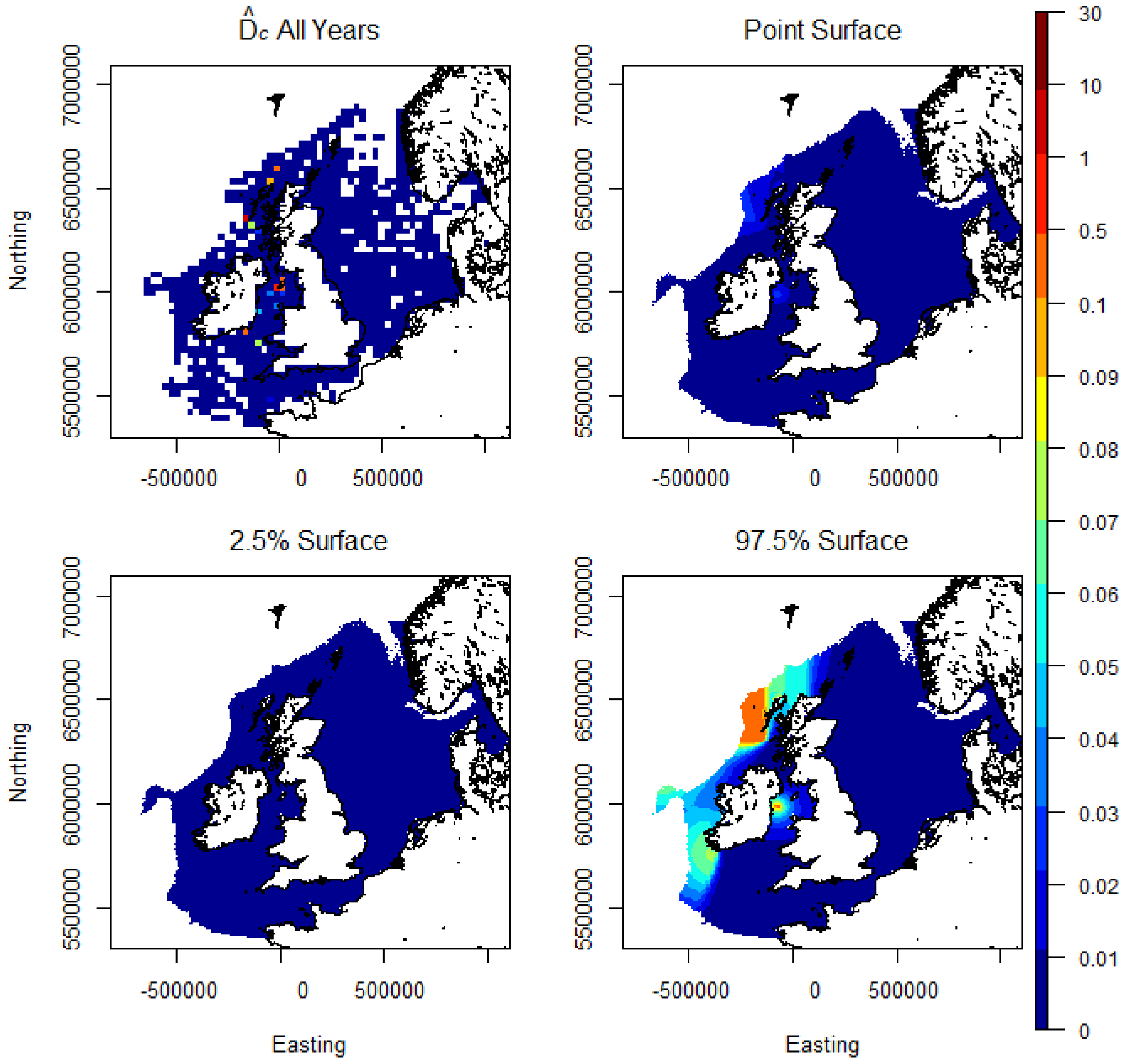

Figure A4.29. Risso's dolphins densities (animals/km^2) spring 2010. Top left: mean $\widehat{D}_c$ all springs combined. Top right: predicted densities spring 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort

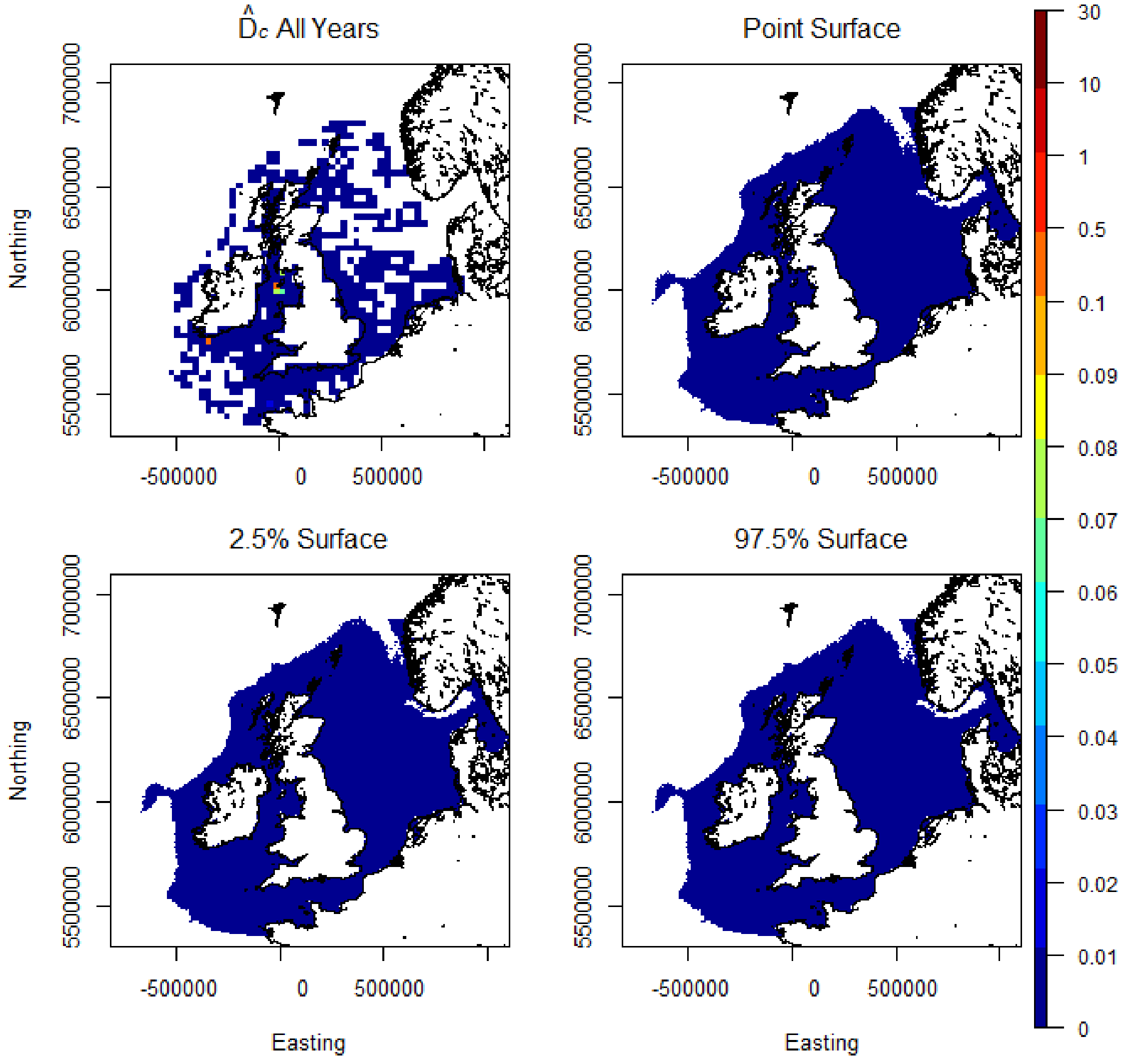

Figure A4.30. Risso's dolphins densities (animals/km^2) autumn 2010. Top left: mean $\hat{D}_c$ all autumns combined. Top right: predicted densities autumn 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

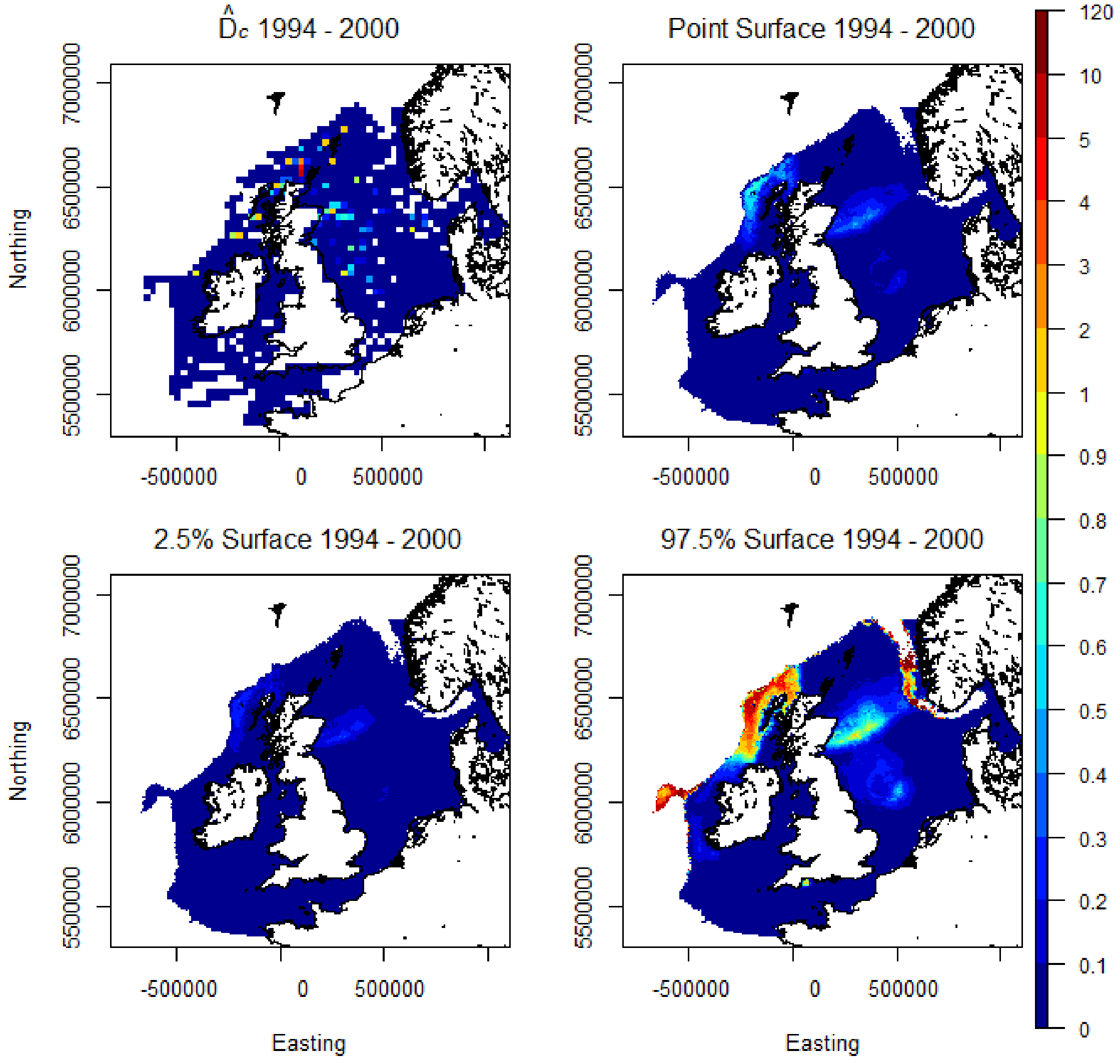

Figure A4.31. White-beaked dolphins densities (animals/km^2) 1994 – 2000. Top left: mean $\widehat{D}_c$ summers 1994 – 2000 combined. Top right: predicted mean summer densities 1994 – 2000. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

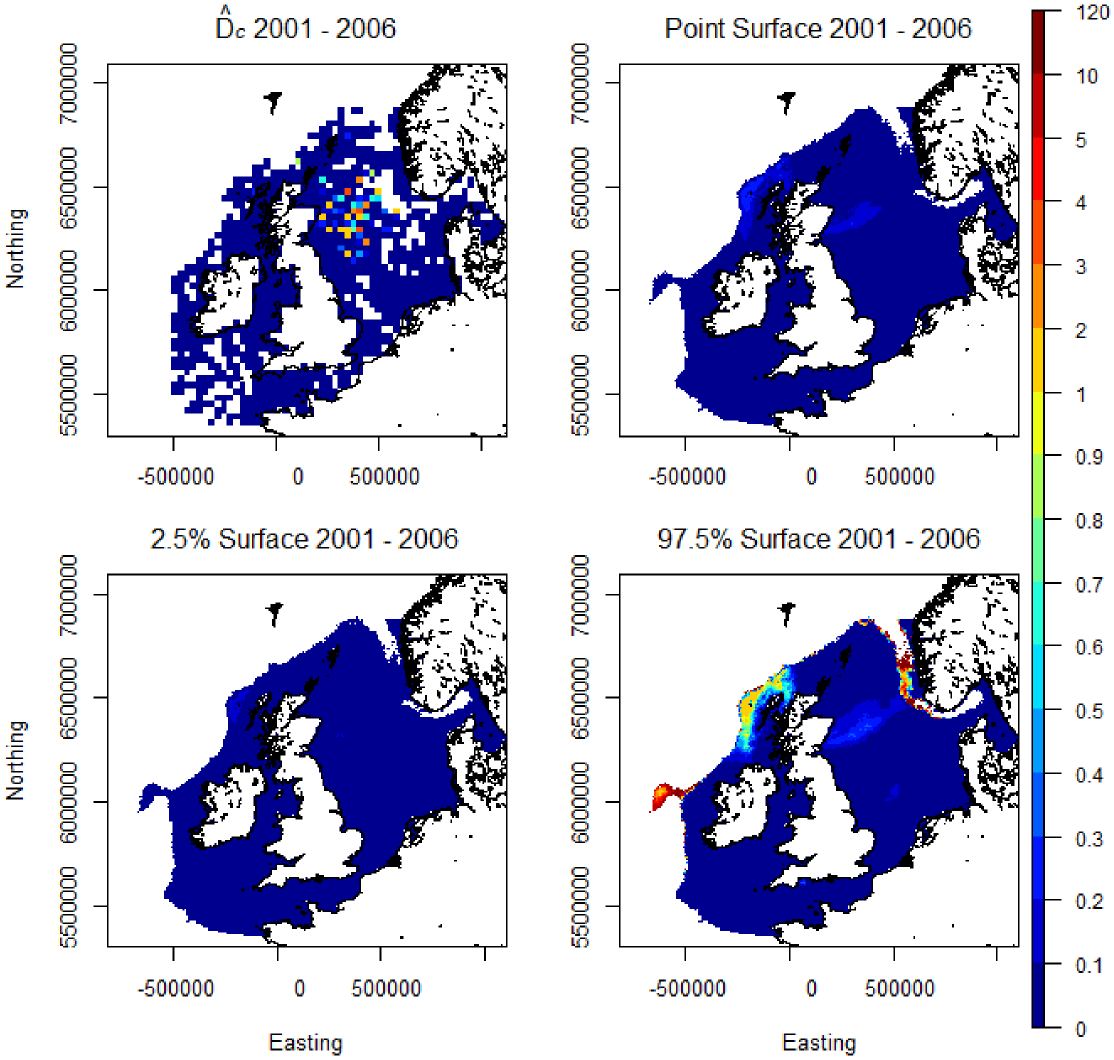

Figure A4.32. White-beaked dolphins densities (animals km^2) 2001 – 2006. Top left: mean $\hat{D}_c$ summers 2001 – 2006 combined. Top right: predicted mean summer densities 2001 – 2006. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

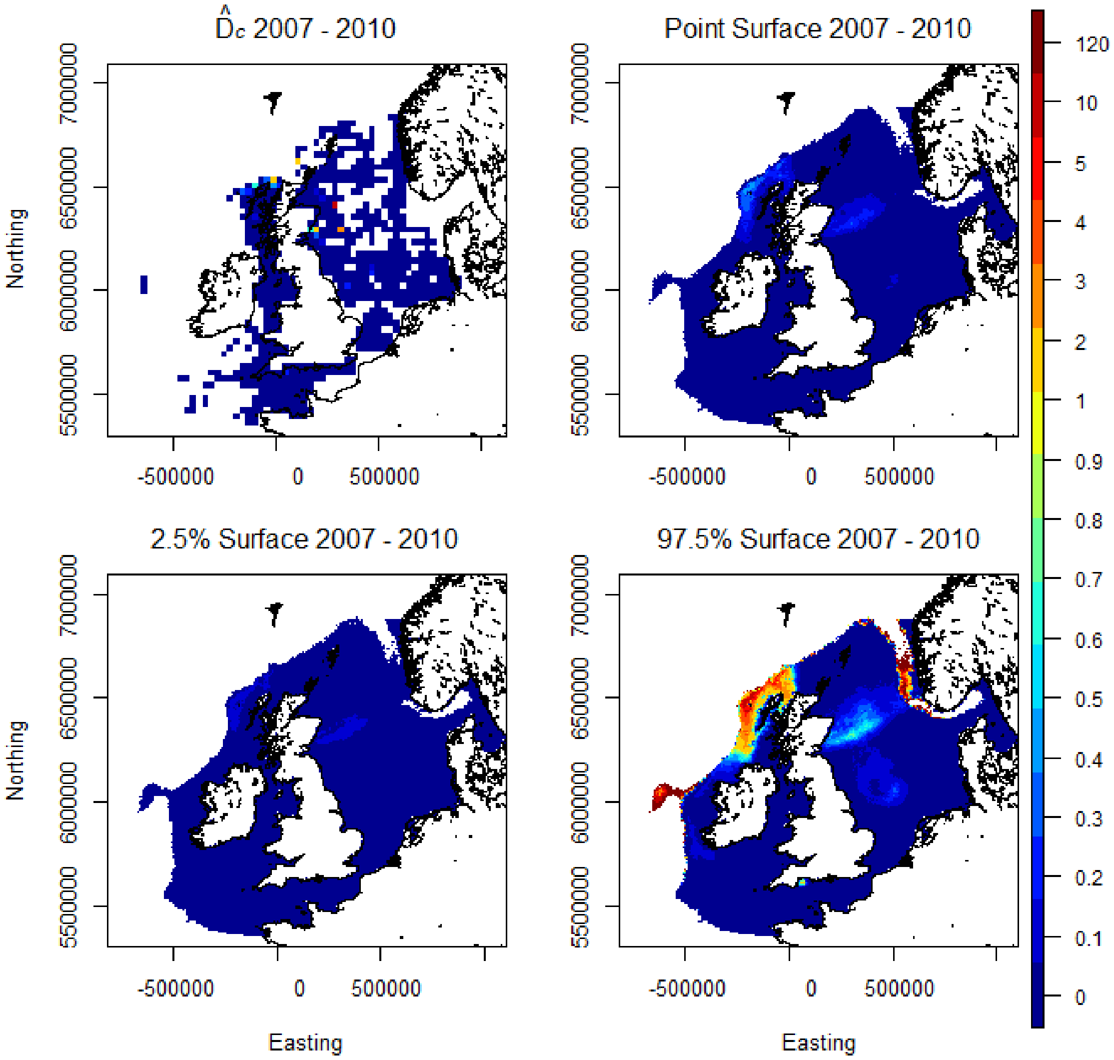

Figure A4.33. White-beaked dolphins densities (animals/km^2) 2007 – 2010. Top left: mean $\widehat{D}_c$ summers 2007 – 2010 combined. Top right: predicted mean summer densities 2007 – 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

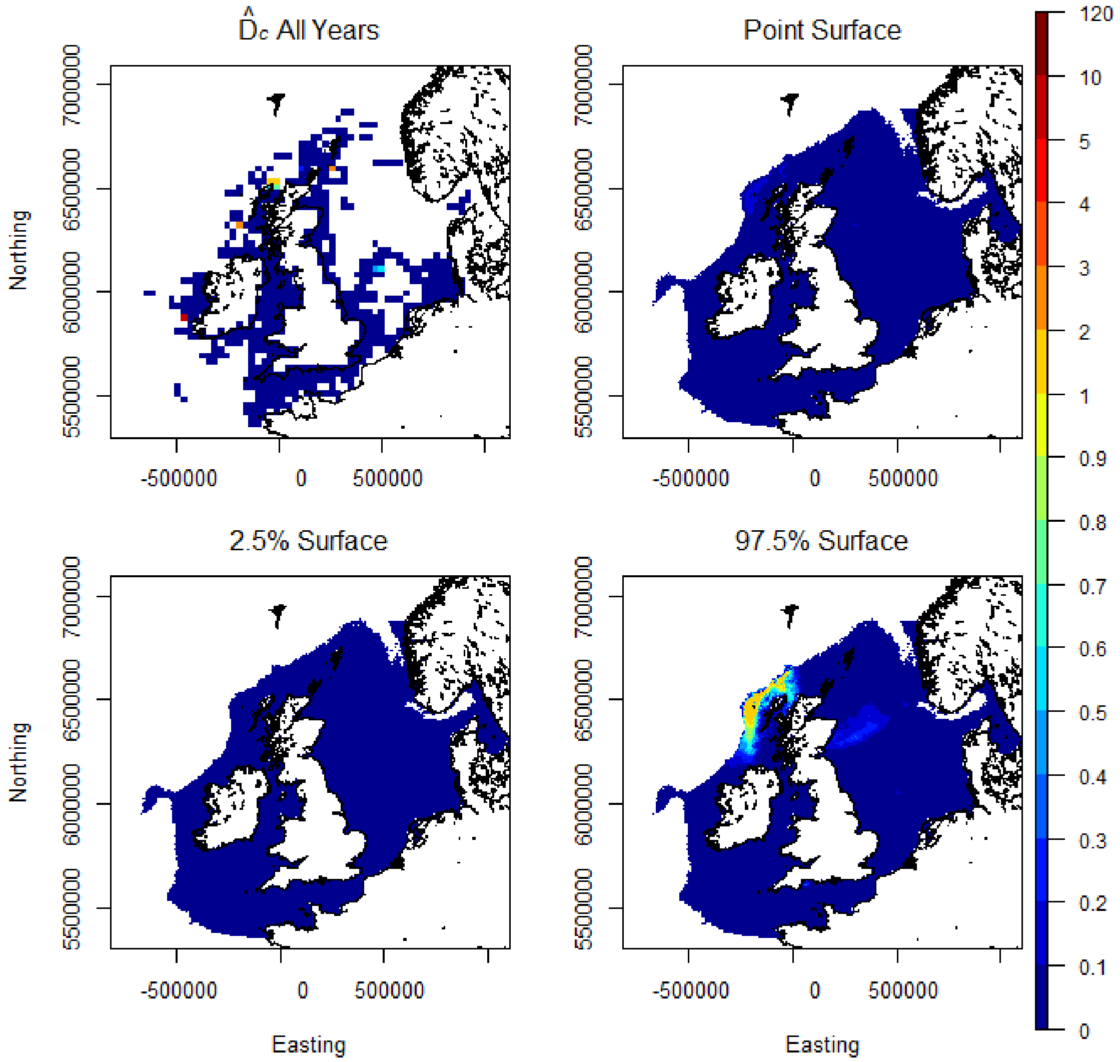

Figure A4.34. White-beaked dolphins densities (animals/km^2) winter 2010. Top left: mean $\hat{D}_c$ 2010 combined. Top right: predicted densities winter 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

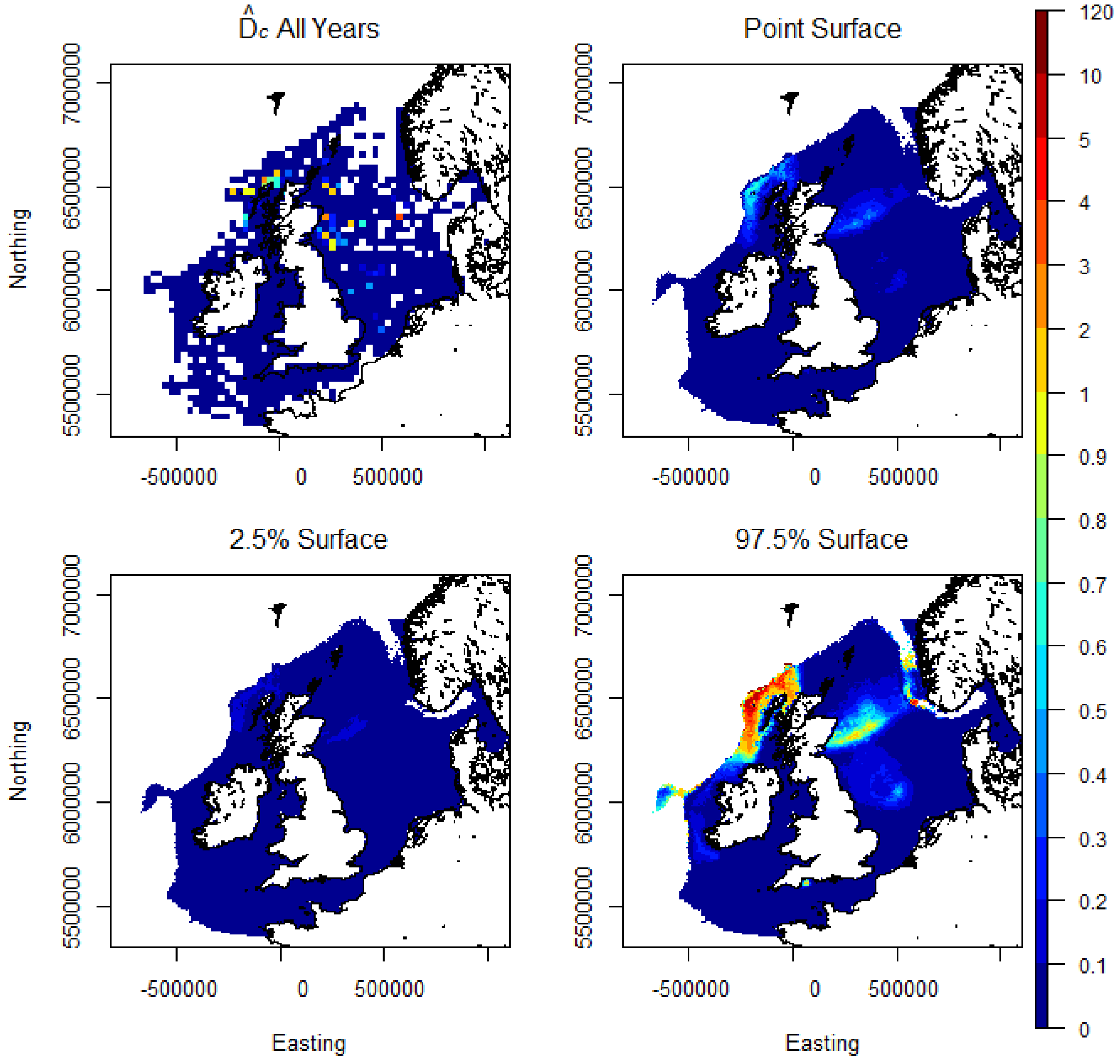

Figure A4.35. White-beaked dolphins densities (animals/km^2) spring 2010. Top left: mean $\widehat{D}_c$ 2010 combined. Top right: predicted densities spring 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities, bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

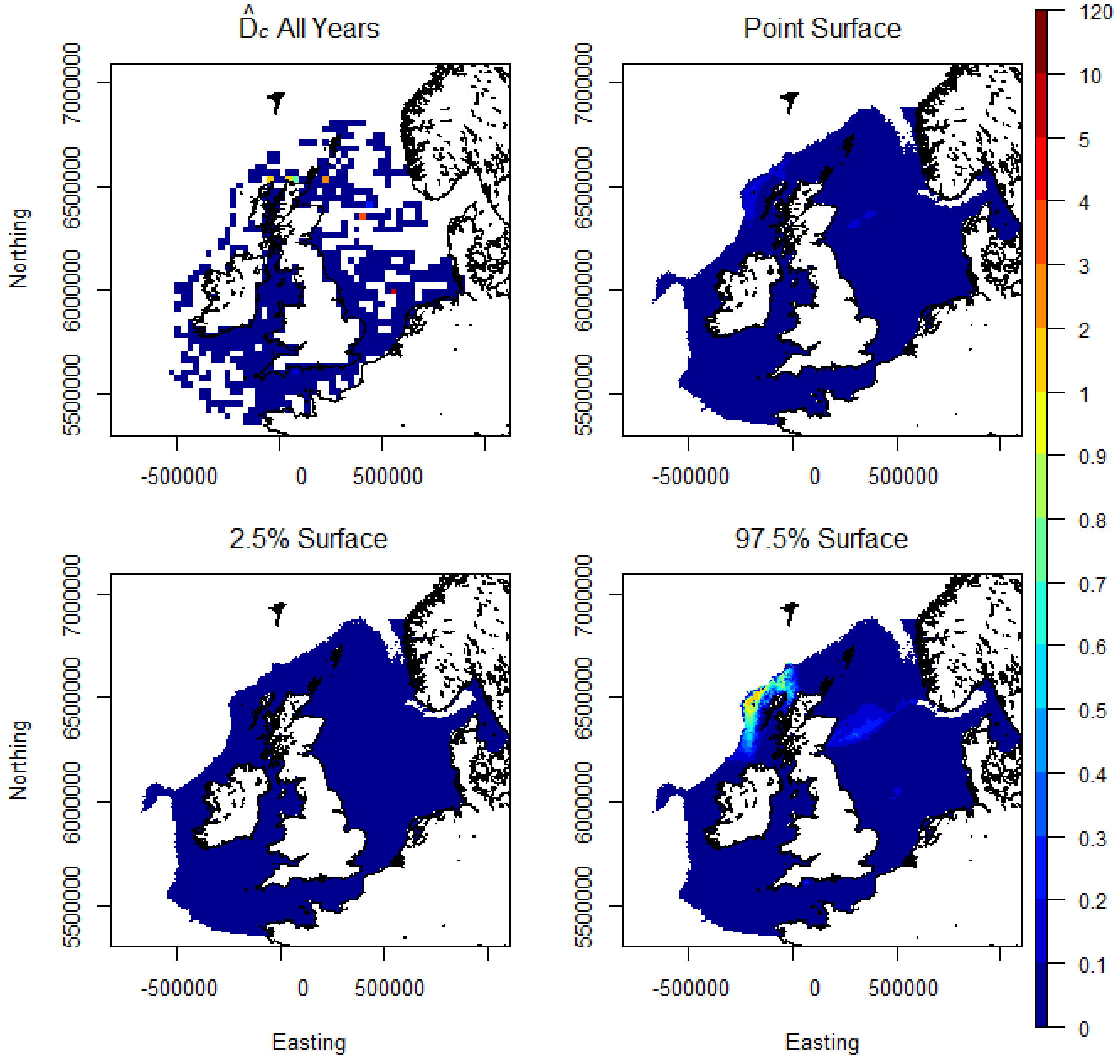

Figure A4.36. White-beaked dolphins densities (animals/km^2) autumn 2010. Top left: mean $\widehat{D}_c$ 2010 combined. Top right: predicted densities autumn 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

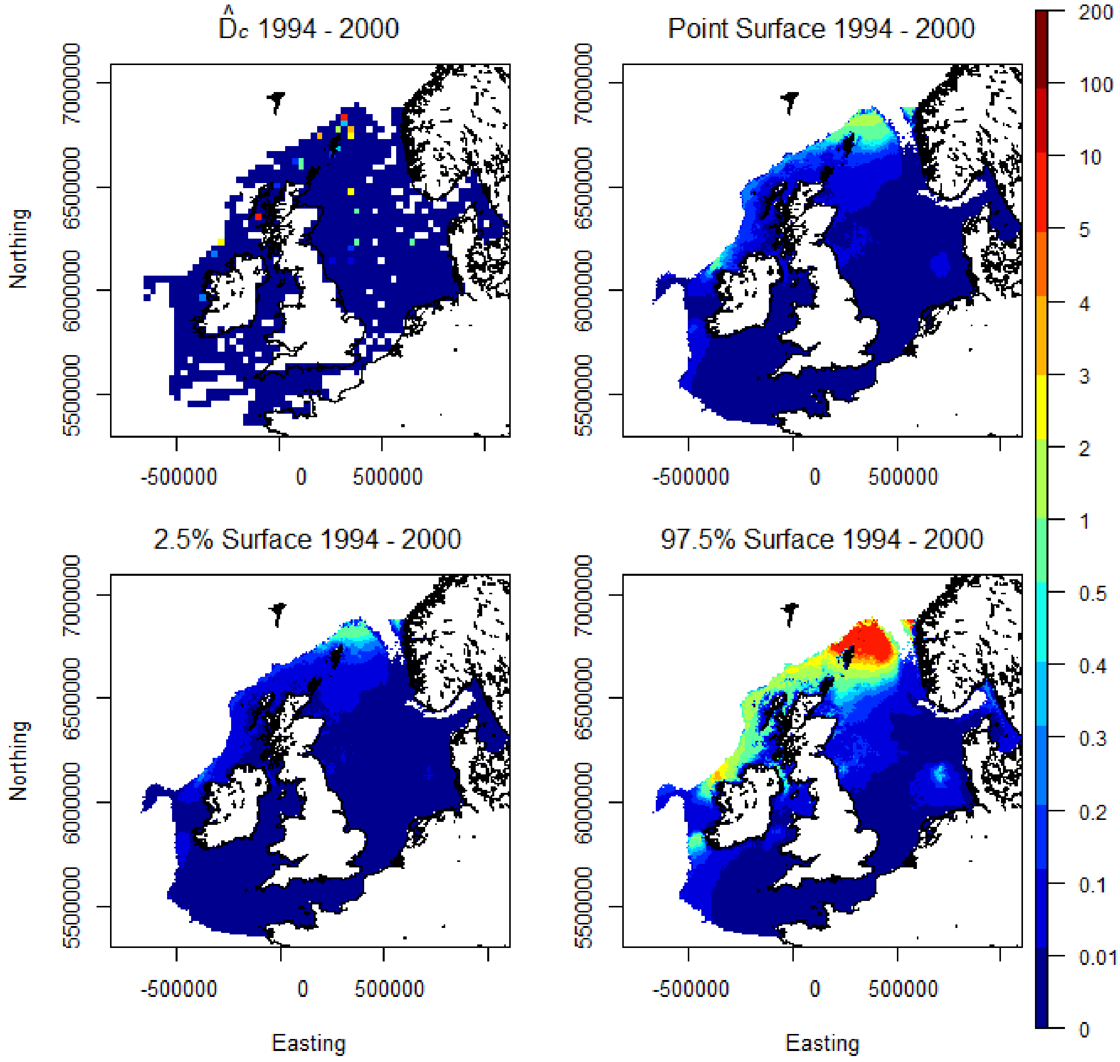

Figure A4.37. White-sided dolphins densities (animals/km^2) 1994 – 2000. Top left: mean $\widehat{D}_c$ summers 1994 – 2000 combined. Top right: predicted mean summer densities 1994 – 2000, bottom left: lower bound (2.5%) of confidence interval of individual cell densities, bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

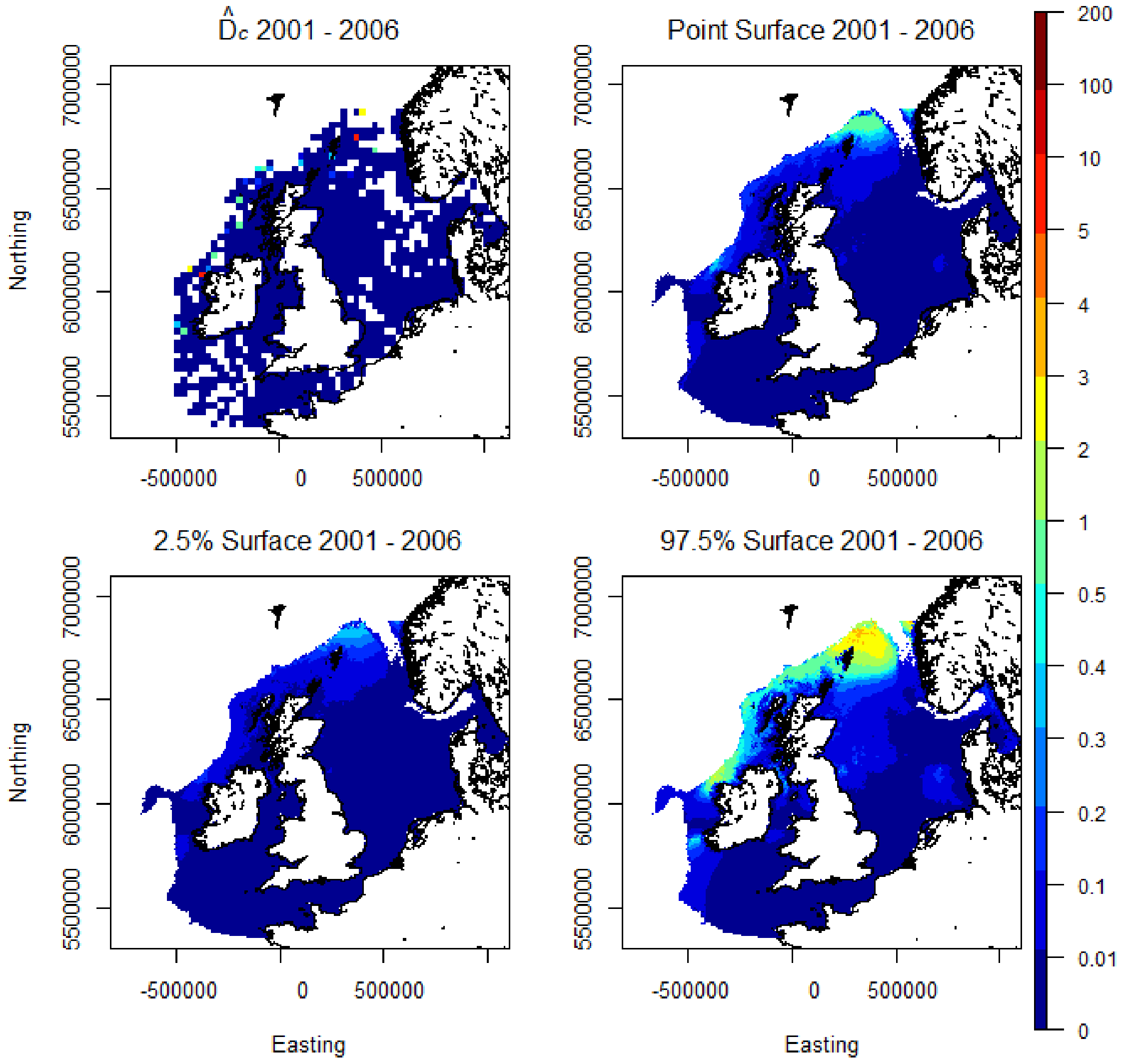

Figure A4.38. White-sided dolphins densities (animals/km^2) 2001 – 2006. Top left: mean $\widehat{D}_c$ summers 2001 – 2006 combined. Top right: predicted mean summer densities 2001 – 2006, bottom left: lower bound (2.5%) of confidence interval of individual cell densities, bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

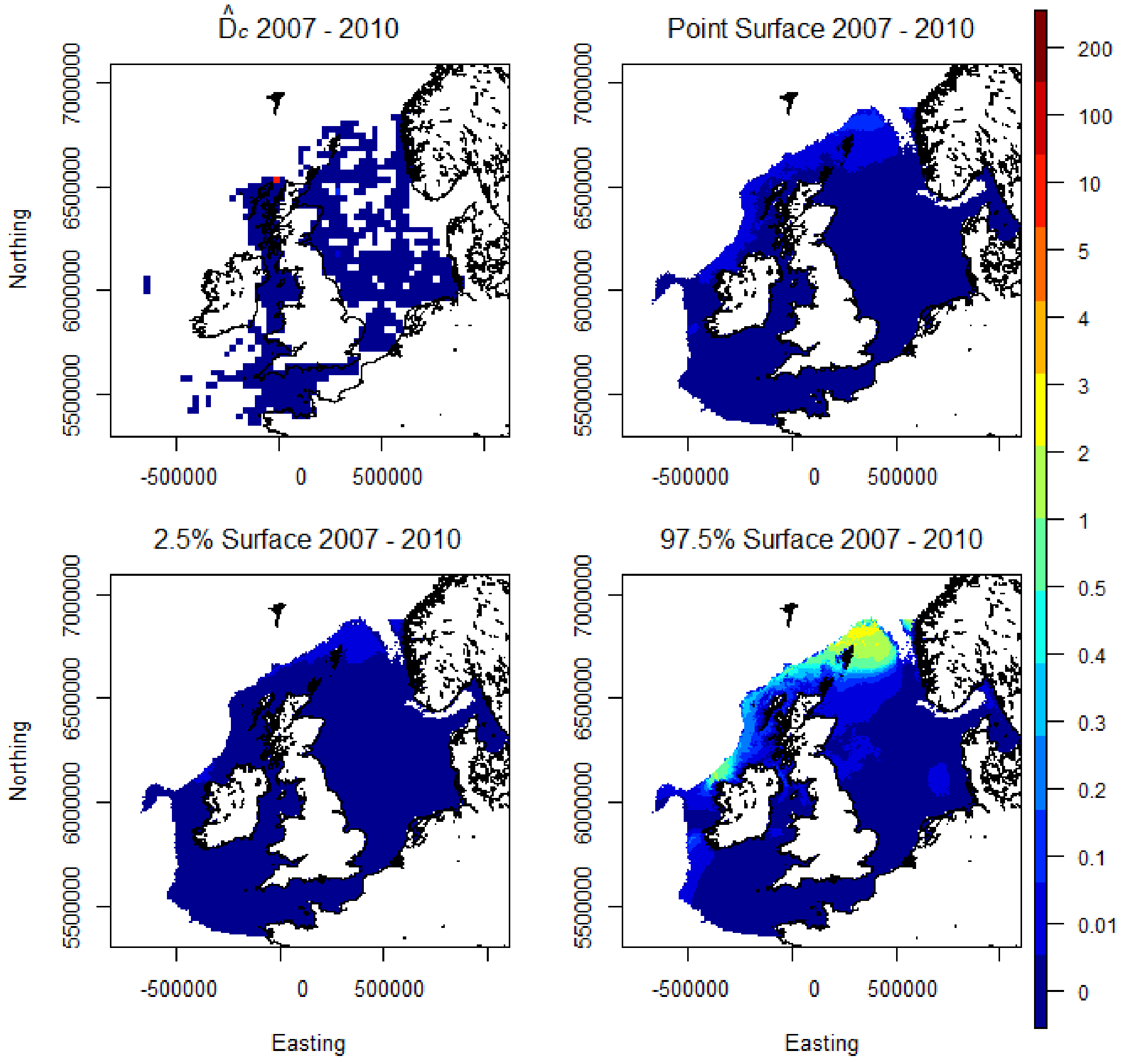

Figure A4.39. White-sided dolphins densities (animals/km^2) 2007 – 2010. Top left: mean $\widehat{D}_c$ summers 2007 – 2010 combined. Top right: predicted mean summer densities 2007 – 2010, bottom left: lower bound (2.5%) of confidence interval of individual cell densities, bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that top left plot exaggerates the spatial coverage of the relevant effort.

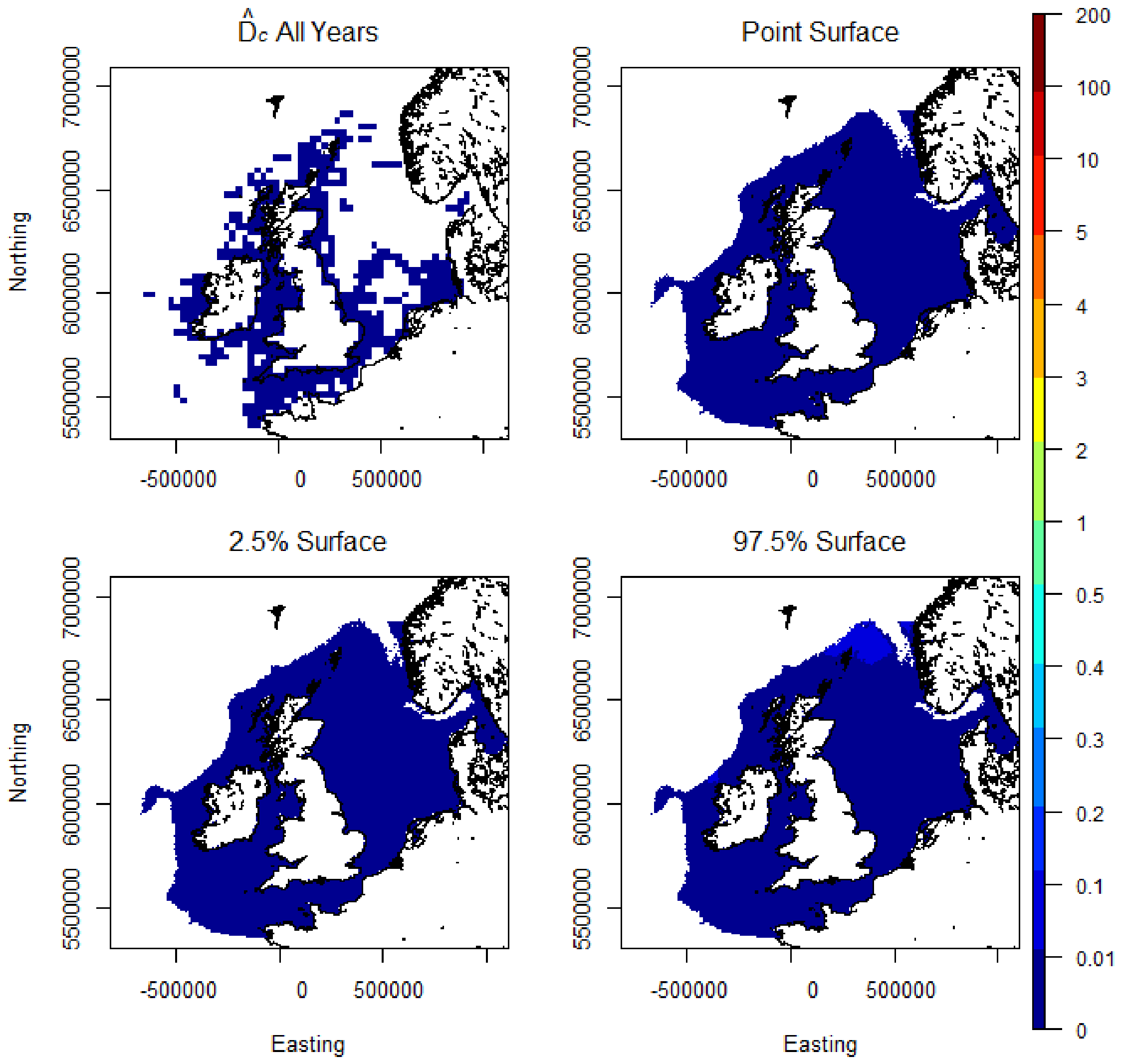

Figure A4.40. White-sided dolphins winter densities (animals/km^2) 2010. Top left: mean $\widehat{D}_c$ winters all years. Top right: predicted densities winter 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

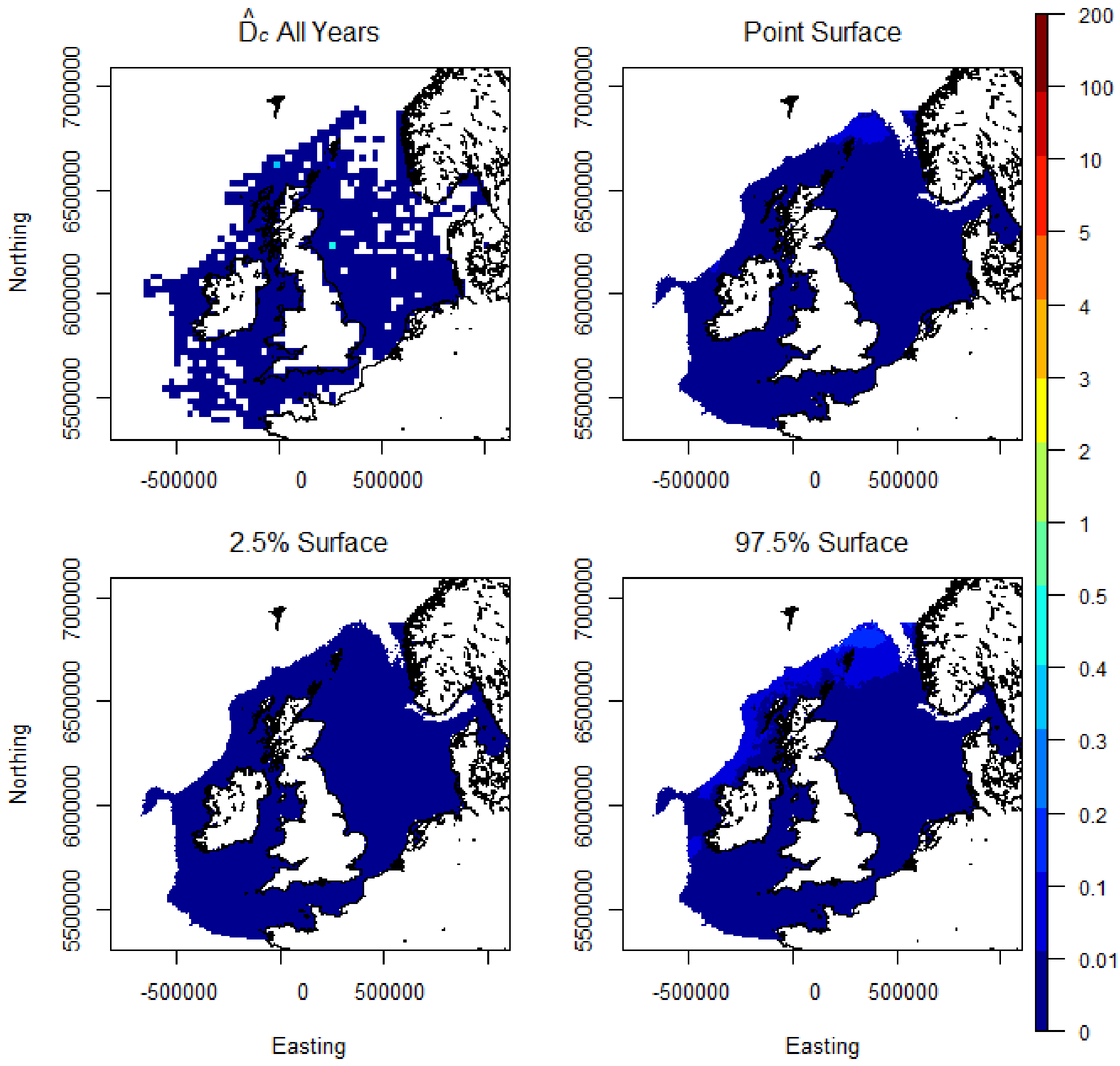

Figure A4.41. White-sided dolphins densities (animals/km^2) spring 2010. Top left: mean $\widehat{D}_c$ spring all years. Top right: predicted densities spring 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

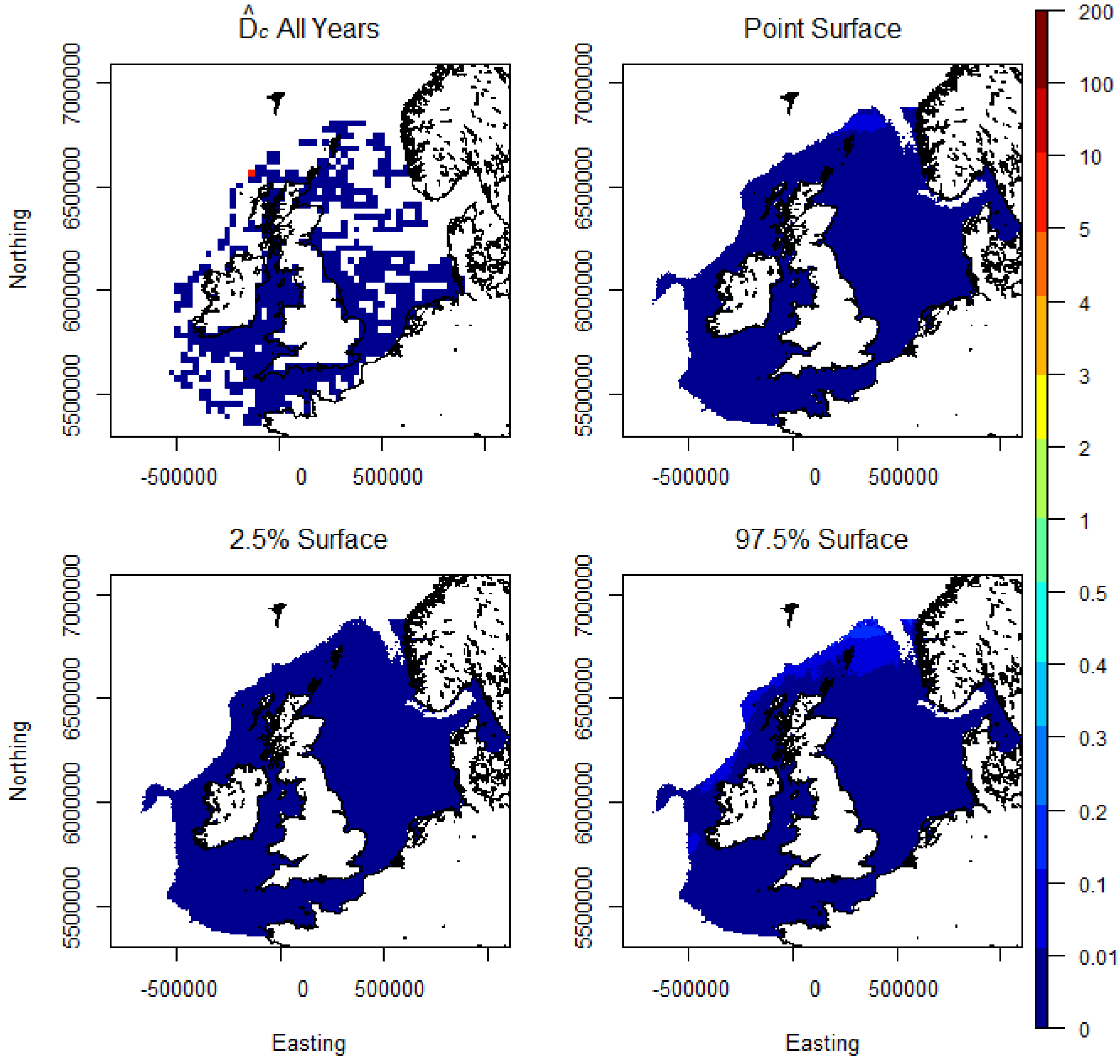

Figure A4.42. White-sided dolphins densities (animals/km^2) autumn 2010. Top left: mean $\hat{D}_c$ autumns all years. Top right: predicted densities autumn 2010. Bottom left: lower bound (2.5%) of confidence interval of individual cell densities. Bottom right: upper bound (97.5%) of confidence interval of individual cell densities. Note that the top left plot exaggerates the spatial coverage of the relevant effort.

Appendix 5. Accuracy as a Function of Area

This analysis of the results of the JCP considered how the accuracy of predictions fluctuated as a function of area for two of the species under consideration: harbour porpoise and Risso's dolphin (as examples of models with relatively low and high uncertainty). By means of the point estimates, accuracy was measured by analysing the change in magnitude of the residuals collected from different sized areas. The method was as follows:

1. For each size of area of interest (25, 100, 150, 225, 400, 625, 900, 1225, 1600, 2025, 2500, 3025 km^2), 60 random locations of that area were selected.
2. Within each random location, all available data points were identified.
3. For each data point, a simple residual (i.e. a model fitted value less the observed $\widehat{N}_c$) was calculated which was subsequently converted to a residual of density by dividing by the relevant segment area associated with the point.
4. These residuals were then averaged across each random sample and each size of area. Both absolute value and actual value was considered.
5. The results were relativized by considering them as a percentage of the mean density for summer 2010.

The whole process was repeated 500 times to provide a 95% confidence interval.

Results

The results are plotted in Figures A5.1-4. Because the localities were randomly located, the actual realized area was sometimes less than the target area (as some selected areas overlapped the edges of the prediction region). The realized areas were approximately 25, 96, 143, 215, 379, 589, 840, 1133, 1460, 1833, 2238, and 2682 km^2.

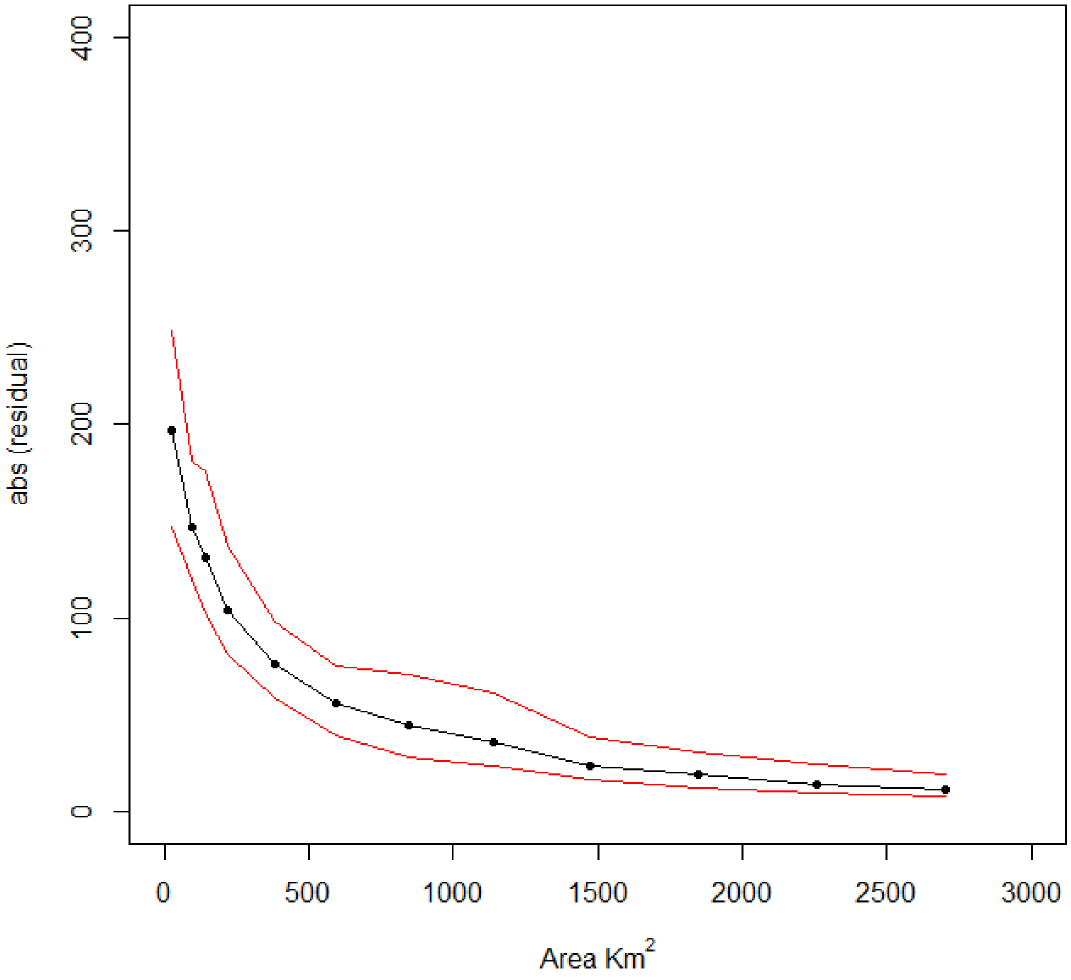

Figure A5.1. Percent absolute residuals of density as a function of area (harbour porpoise). Results are given as a percentage, scaled by the mean predicted density from summer 2010 (0.213 animals/km^2). Black line and points: mean values, red lines lower and upper 95% confidence bounds.

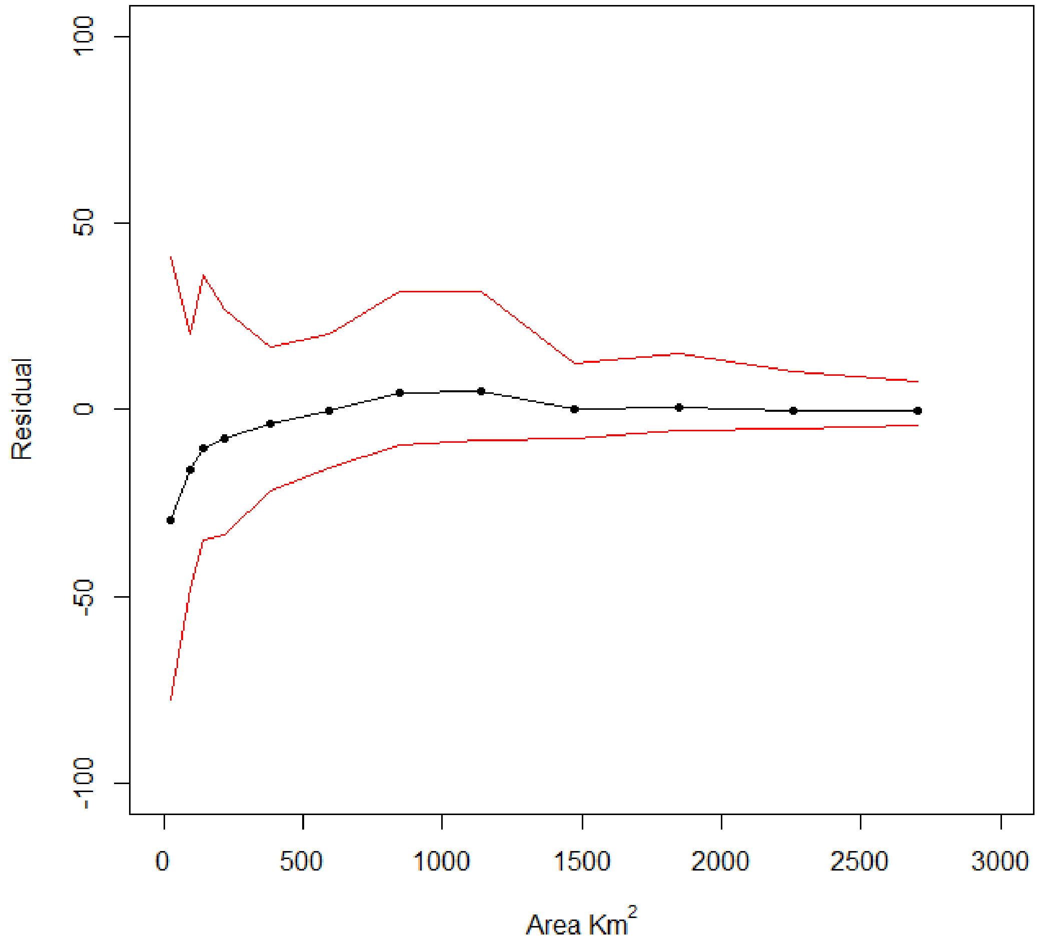

Figure A5.2. Percent residuals as a function of area (harbour porpoise). Results are given as a percentage, scaled by the mean predicted density from summer 2010 (0.213 animals/km^2). Black line and points: mean values, red lines lower and upper 95% confidence bounds.

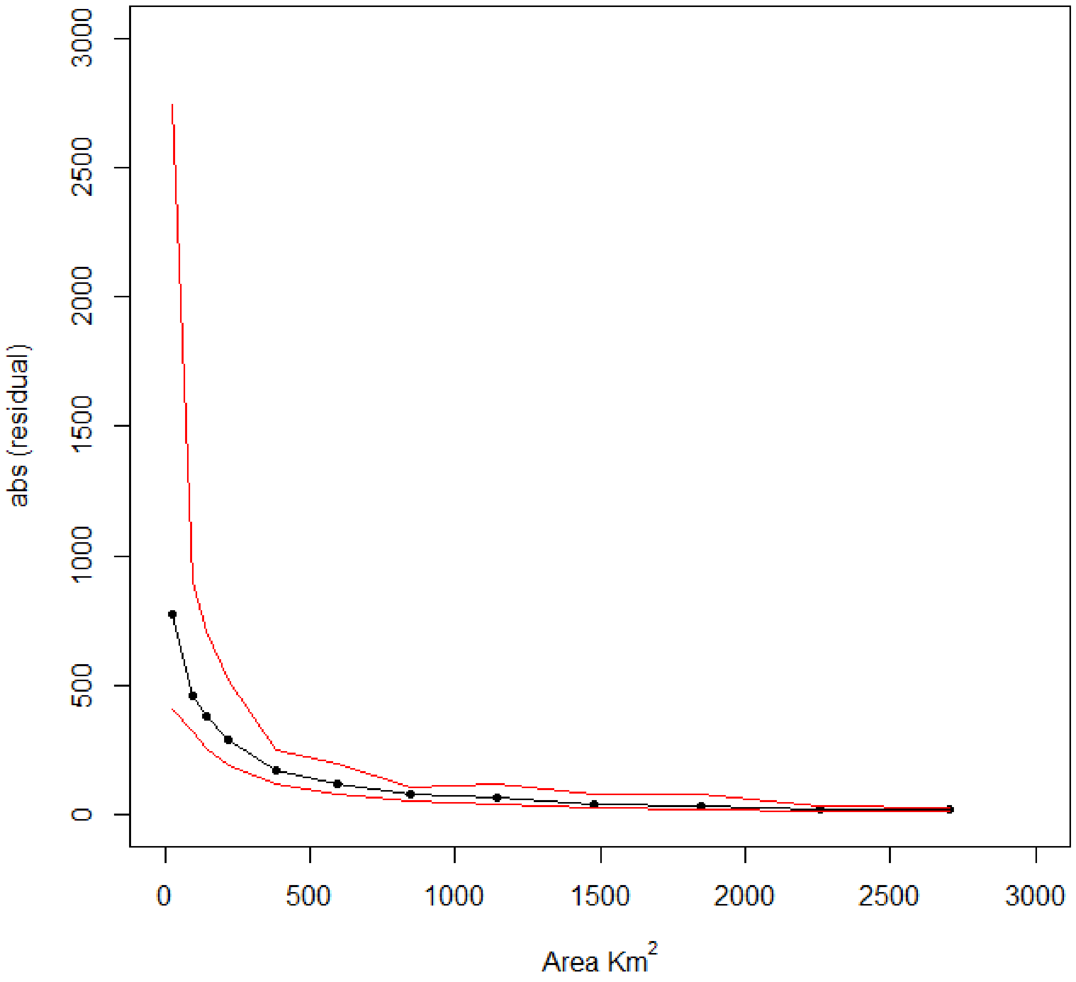

Figure A5.3. Percent absolute residuals as a function of area (Risso's dolphin). Results are given as a percentage, scaled by the mean predicted density from summer 2010 (0.001 animals/km^2). Black line and points: mean values, red lines lower and upper 95% confidence bounds.

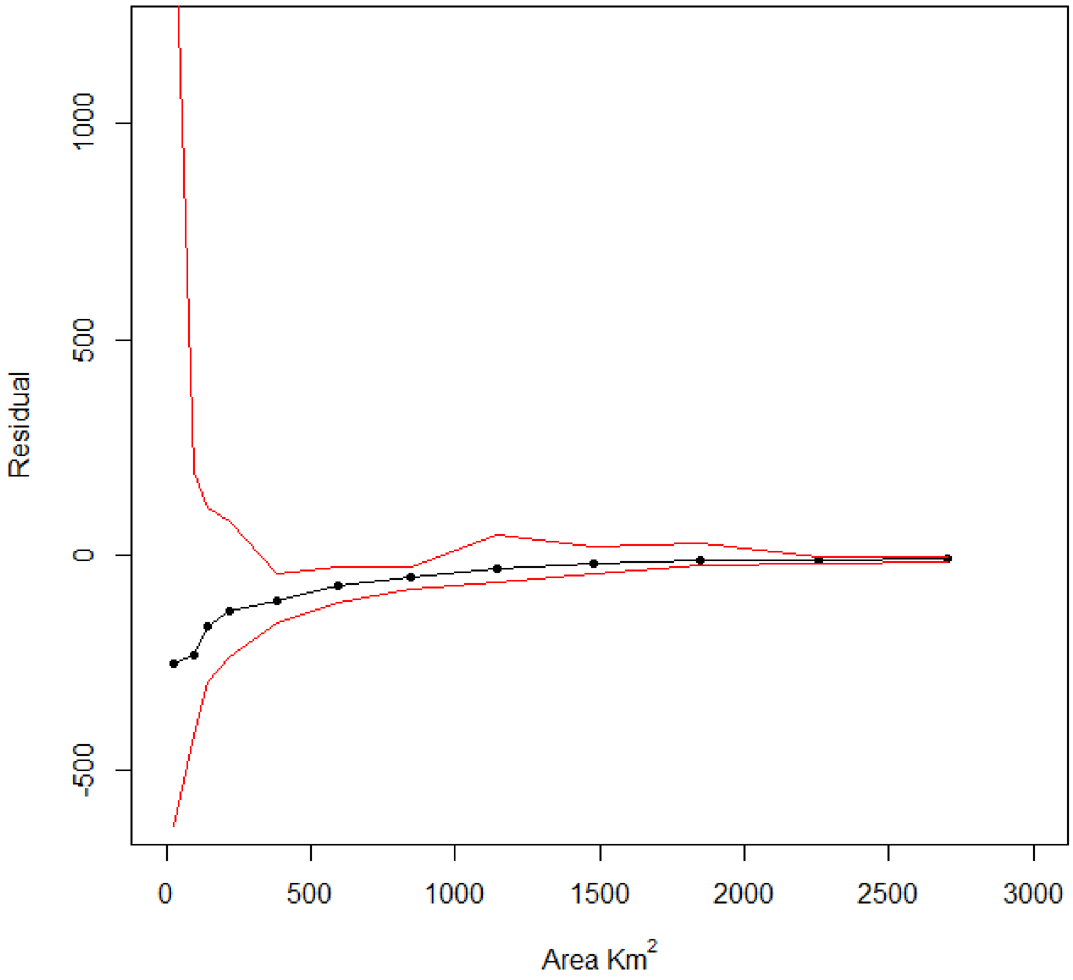

Figure A5.4. Percent residuals as a function of area (Risso's dolphin). Results are given as a percentage, scaled by the mean predicted density from summer 2010 (0.001 animals/km^2). Black line and points: mean values, red lines lower and upper 95% confidence bounds.

It is readily apparent that the inaccuracy associated with Risso's dolphin over small areas is greater than that of harbour porpoise. Nevertheless, for both species, inaccuracy, at least compared to the overall area, declines rapidly once a few hundred square kilometres is considered.

Appendix 6. JCP Abundances for particular areas of interest

The United Kingdom's Inter-Agency Marine Mammal Working Group (IAMMWG) has delineated species specific management unit areas (Table A6.1) for the commoner species of cetacean on the continental shelf (IAMMWG, 2015). These areas are illustrated in Figures A6.1., A6.2. and A6.3. for harbour porpoise, bottlenose dolphin and all other species considered here respectively. To provide context in relation to the wider cetacean populations on the European shelf, bootstrap estimates of each developer area's abundance were calculated as a percentage of the wider management unit JCP abundance for each species. For each bootstrap a developer area abundance estimate was made along with a management unit area abundance estimate (averaged over the summers of 2007 – 2010, the most recent period). The quotient was then taken of the developer area abundance and the management unit abundance. This constituted one bootstrap replicate of the desired percentage. Bootstrap confidence intervals were then calculated from the distribution of percentages. Developer areas were chosen based on best available information at this time of the areas of interest for renewable energy developments. Anyone wishing to estimate abundance for any particular area not shown here should consult the JNCC website where details can be found on how to obtain those estimates. For harbour porpoise and bottlenose dolphin only, no estimates were made in large developer areas that substantially overlapped different management areas (e.g. the developer area north of Caithness/Sutherland). *Note that in the JCP Phase III analyses, the abundance estimates apply only to areas from 0 to 300m depth, and for harbour porpoise, areas to the east of easting 820000 were excluded, due to the great uncertainty in the estimates arising from the small amount of data available for those areas.*

Table A6.1. Cetacean management units (IAMMWG, 2015) see also figures A6.1., A6.2. & A6.3).

Species	**Management Unit**	**Description**	**Area (Km^2) up to 300 m contour***
Harbour porpoise	CIS	*Celtic and Irish Seas.* Southeastern continental shelf. Irish sea to 55° Lat. north English channel to Cotentin peninsula (Normandy)	333656
	WS	*West Scotland*. Northwestern continental shelf	114892
	NS	*North Sea*. The North Sea between 62° Lat. north and to Cotentin peninsula (Normandy)	586869**
Bottlenose dolphin	IS	*Irish Sea*. Irish Sea between 52 and 55° Lat. North, excluding Isle of Man territorial waters.	44779
	OCSW	*Offshore Channel, Celtic Sea and Southwest England.*	200084
	CWC	*Coastal West Channel.*	9676
	OW	*Offshore Waters.* Continental shelf outside of coastal waters, westward from -9° E in the south	151126
	WCI***	*West Coast of Ireland.* Western Irish coastal waters	27327
	CWSH	*Coastal West Scotland and Hebrides.* Western coastal waters of Scotland	41635
	CES	*Coastal East Scotland*. Eastern coastal waters of Scotland (includes Orkneys)	20989
	GNS	*Greater North Sea*	540805
Minke whale, common dolphin, Risso's dolphin, white-beaked dolphin and white-sided dolphin	CGNS	*Celtic and Greater North Sea*. All of the European shelf from 48° Lat. north.	1088939 (harbour porpoise: 1035700**)

*The JCP Phase III region does not include areas deeper than 300m given that no data was included in the analyses for those areas.

**The area to the east of easting 820000 was not included in the abundance estimates for the harbour porpoise, as the density maps show great uncertainty deriving from the few data available for this area.

***The Shannon estuary management unit is not considered here, as no data from this area were included in the analyses.

Figure A6.1. Developer areas (red) and harbour porpoise management units (green). Please note that few data were included for areas to the east of easting 820000, and therefore, to avoid the inevitable great uncertainty in the eastern predictions, these areas were not included in the estimates.

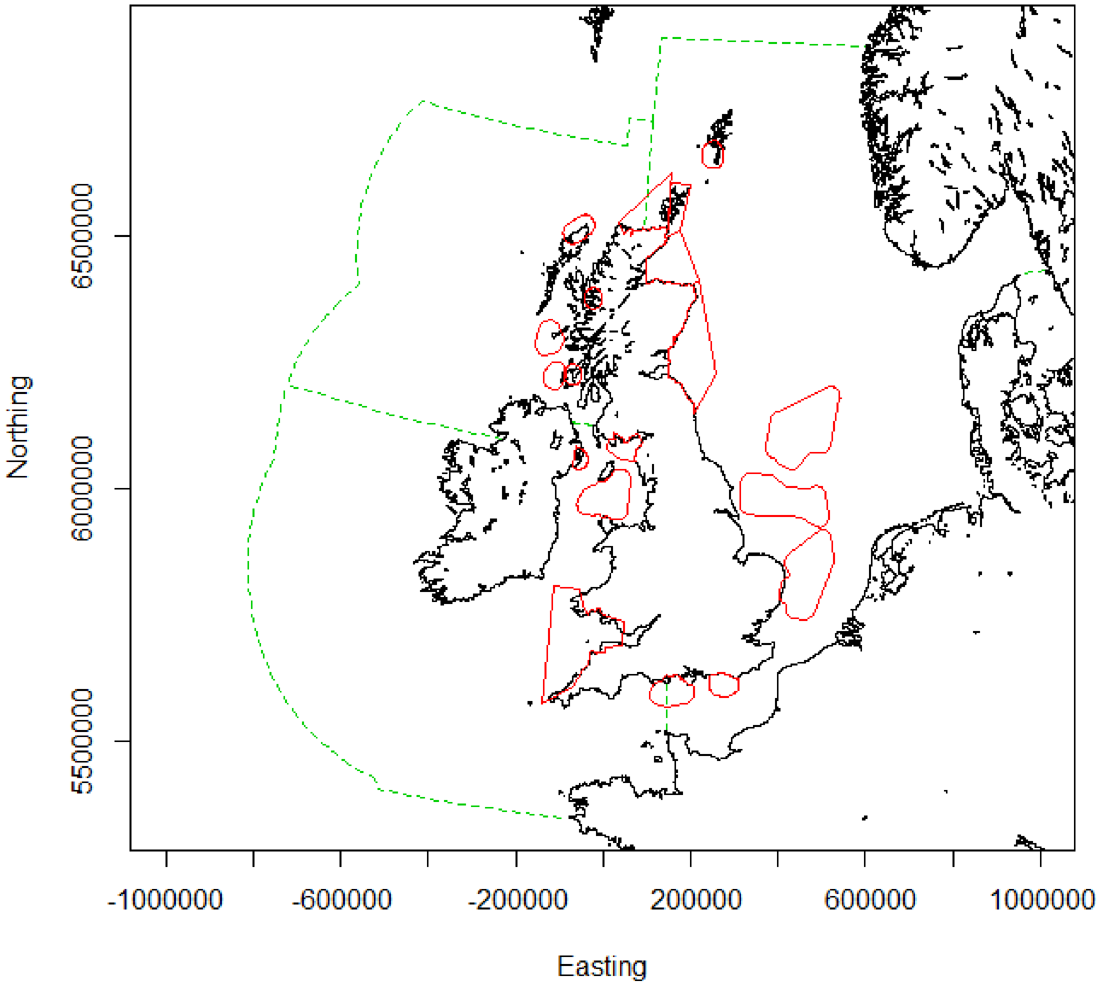

Figure A6.2. Developer areas (red) and bottlenose dolphin management units (green).

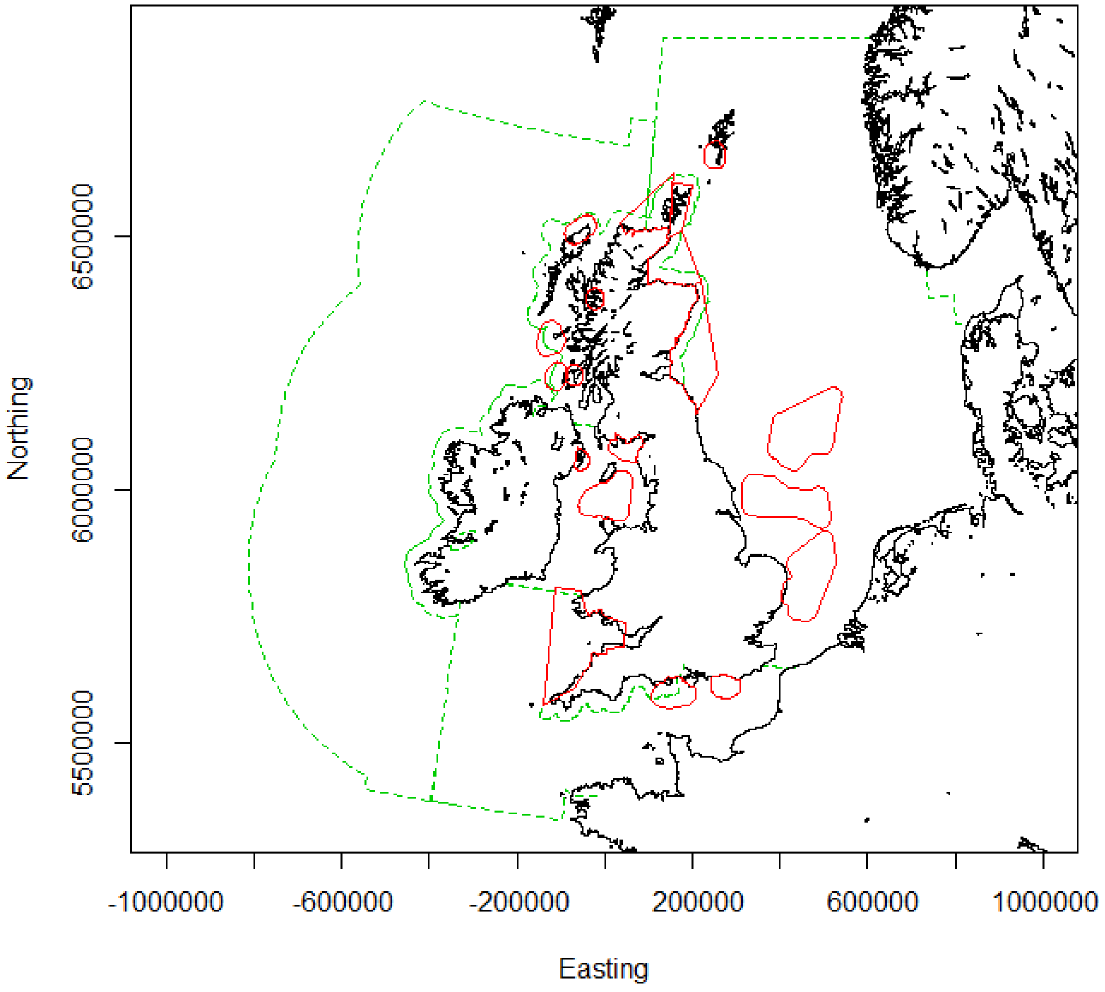

Figure A6.3. Developer areas (red) and the minke whale, common dolphin, Risso's dolphin, white-beaked dolphin and white-sided dolphin management unit (green).

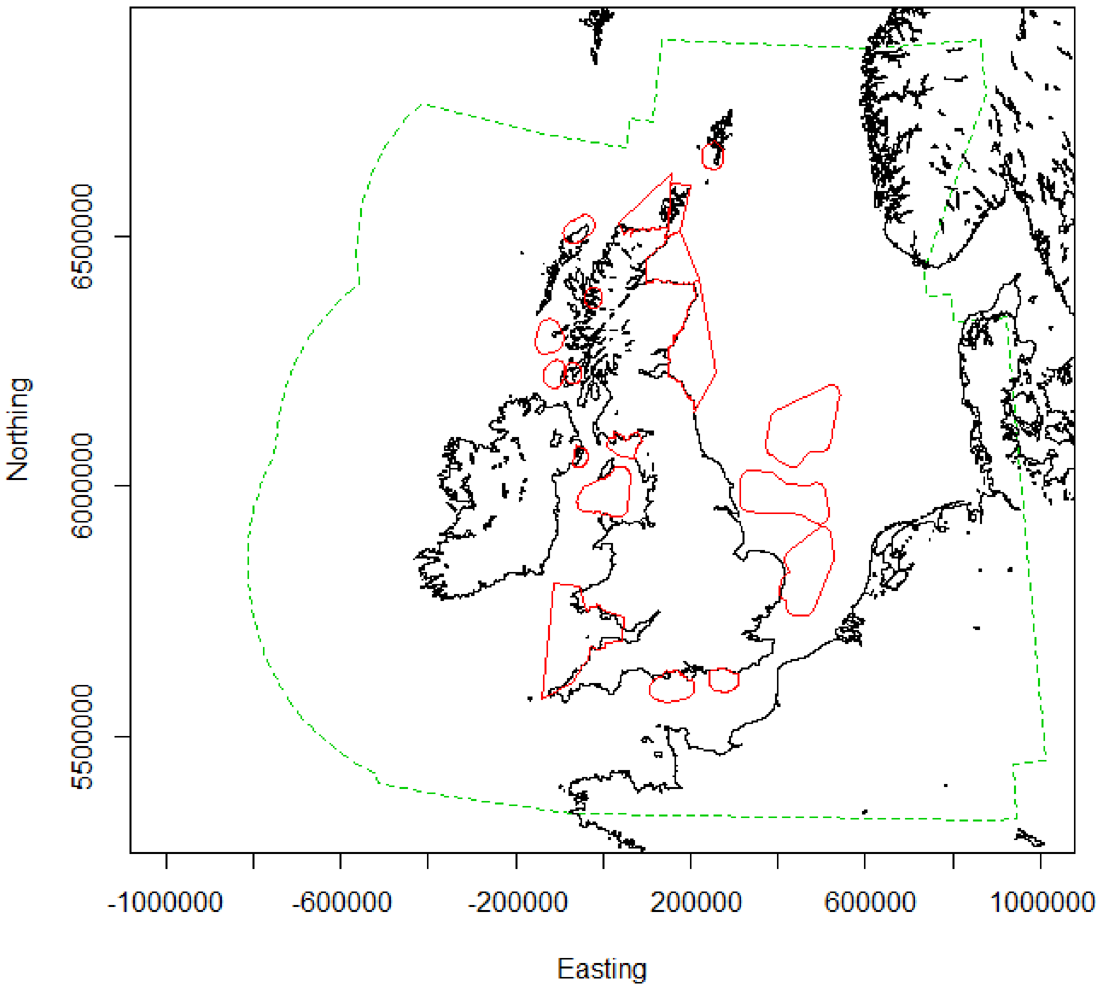

Table A6.2. Predicted harbour porpoise abundance in each developer area as a percentage of the predicted numbers from the relevant management unit (%). All figures are based on estimates for summers 2007 – 2010 (averaged). North Scotland region not included as this straddles two management areas.

	Point	**Lower**	**Upper**	**% Area of relevant management unit**
Moray Firth	1.4	0.5	3.0	1.3
Firth of Forth	1.4	0.6	2.3	2.4
Atlantic Array	15.4	7.9	18.6	5.9
Islay	1.1	0.7	1.6	1.8
Solway Firth	1.6	0.8	2.2	0.7
East Orkney	0.6	0.3	1.1	0.5
Lewis	3.9	3.1	4.9	1.6
Kyle Rhea	0.5	0.3	0.8	0.3
Sound of Islay	0.4	0.2	0.6	0.5
Argyll Array	3.2	2.4	3.9	2.9
Hastings	0.4	0.2	0.6	0.7
IOW	0.5	0.2	0.8	1.3
Dogger Bank	7.6	4.6	9.1	3.0
South Dogger Bank	13.9	8.1	18.3	2.4
Norfolk Bank	13.9	8.9	19.2	2.4
Irish Sea	4.7	2.2	5.9	2.5
Strangford Lough	0.5	0.2	0.7	0.2
Shetlands	0.2	0.1	0.4	0.3

Table A6.3. Predicted minke whale abundance in each developer area as a percentage of the predicted numbers from the relevant management unit. All figures are based on estimates for summers 2007 – 2010 (averaged).

	Point	***Lower***	***Upper***	***% Area of relevant management unit***
Moray Firth	1.3	0.7	1.6	0.7
Firth of Forth	2.2	1.1	3	1.3
Atlantic Array	1.7	1.0	2.4	1.8
Islay	0.6	0.4	0.8	0.2
Solway Firth	0.2	0.1	0.4	0.2
North	1	0.6	1.4	0.6
East Orkney	0.5	0.3	0.8	0.3
Lewis	0.6	0.4	0.8	0.2
Kyle Rhea	0.1	0	0.1	0
Sound of Islay	0.1	0.1	0.1	0.1
Argyll Array	1.9	1.1	2.8	0.3
Hastings	0	0	0	0.2
IOW	0	0	0.1	0.4
Dogger Bank	3.2	1.7	5.0	1.6
South Dogger Bank	1.9	0.8	4.9	1.3
Norfolk Bank	0.2	0	0.6	1.3
Irish Sea	1.1	0.6	2.8	0.8
Strangford Lough	0.2	0.1	0.6	0.1
Shetlands	0.2	0.1	0.5	0.2

Table A6.4. Predicted bottlenose dolphin abundance in each developer area as a percentage of the predicted numbers from the relevant management unit. All figures are based on estimates for summers 2007 – 2010 (averaged).

	Point	***Lower***	***Upper***	***% Area of relevant management unit***
Atlantic Array	1.6	0.9	3.6	9.8
Islay	1.1	0.8	2.0	5.0
Solway Firth	0.6	0.3	1	5.1
East Orkney	3.6	1.9	5.0	14.4
Lewis	4.5	2.5	5.2	4.4
Kyle Rhea	1.9	1.3	2.6	0.7
Sound of Islay	0.7	0.6	1.2	1.4
Argyll Array	3.9	3.0	5.7	7.9
Hastings	0.1	0	0.2	1.2
Dogger Bank	2.6	1.5	4.4	3.3
South Dogger Bank	0.9	0.5	1.7	2.6
Norfolk Bank	0.6	0.2	1.1	2.6
Irish Sea	1.2	0.6	2.2	18.4
Strangford Lough	0.2	0.1	0.3	1.4
Shetlands	0.3	0.1	0.5	0.3

Table A6.5. Predicted common dolphin abundance in each developer area as a percentage of the predicted numbers from the relevant management unit. All figures are based on estimates for summers 2007 – 2010 (averaged).

	Point	***Lower***	***Upper***	***% Area of relevant management unit***
Moray Firth	0.1	0	0.2	0.7
Firth of Forth	0.1	0	0.1	1.3
Atlantic Array	8	5.8	12.7	1.8
Islay	0.1	0	0.1	0.2
Solway Firth	0	0	0	0.2
North	0.4	0.2	1.1	0.6
East Orkney	0	0	0	0.3
Lewis	0.2	0.1	0.4	0.2
Kyle Rhea	0	0	0	0
Sound of Islay	0	0	0	0.1
Argyll Array	0.7	0.5	1.6	0.3
Hastings	0	0	0	0.2
IOW	0	0	0	0.4
Dogger Bank	0	0	0	1.6
South Dogger Bank	0	0	0	1.3
Norfolk Bank	0	0	0	1.3
Irish Sea	0.1	0	0.1	0.8
Strangford Lough	0	0	0.1	0.1
Shetlands	0	0	0	0.2

Table A6.6. Predicted Risso's dolphin abundance in each developer area as a percentage of the predicted numbers from the relevant management unit up to the 300m depth contour. All figures are based on estimates for summers 2007 – 2010 (averaged).

	Point	**Lower**	**Upper**	**% Area of relevant management unit**
Moray Firth	0.2	0.1	0.3	0.7
Firth of Forth	0.3	0.1	0.5	1.3
Atlantic Array	0.2	0	1	1.8
Islay	0.1	0	0.2	0.2
Solway Firth	0.3	0.1	0.6	0.2
North	1.1	0.8	1.6	0.6
East Orkney	0.1	0.1	0.2	0.3
Lewis	0.8	0.6	1	0.2
Kyle Rhea	0.1	0.1	0.1	0
Sound of Islay	0	0	0	0.1
Argyll Array	1	0.6	1.9	0.3
Hastings	0	0	0	0.2
IOW	0	0	0	0.4
Dogger Bank	0.1	0	0.3	1.6
South Dogger Bank	0.1	0	0.2	1.3
Norfolk Bank	0	0	0.1	1.3
Irish Sea	3	1.5	5.3	0.8
Strangford Lough	0.2	0.1	0.4	0.1
Shetlands	0	0	0.1	0.2

Table A6.7. Predicted white-beaked dolphin abundance in each developer area as a percentage of the predicted numbers from the relevant management unit. All figures are based on estimates for summers 2007 – 2010 (averaged).

	Point	***Lower***	***Upper***	***% Area of relevant management unit***
Moray Firth	0.5	00	0.6	0.7
Firth of Forth	4.6	0.1	5.9	1.3
Atlantic Array	0	0	0	1.8
Islay	0	0	0	0.2
Solway Firth	0	0	0	0.2
North	0.1	0	0.2	0.6
East Orkney	0	0	0.1	0.3
Lewis	0.9	0	1.5	0.2
Kyle Rhea	0	0	0	0
Sound of Islay	0	0	0	0.1
Argyll Array	0.2	0	0.3	0.3
Hastings	0	0	0	0.2
IOW	0	0	0	0.4
Dogger Bank	3.2	0.1	4.4	1.6
South Dogger Bank	1.9	0	2.7	1.3
Norfolk Bank	0.2	0	0.4	1.3
Irish Sea	0	0	0	0.8
Strangford Lough	0	0	0	0.1
Shetlands	0	0	0	0.2

Table A6.8. Predicted white-sided dolphin abundance in each developer area as a percentage of the predicted numbers from the relevant management unit. All figures are based on estimates for summers 2007 – 2010 (averaged).

	Point	**Lower**	**Upper**	**% Area of relevant management unit**
Moray Firth	0.2	0.1	0.3	0.7
Firth of Forth	0.2	0.1	0.7	1.3
Atlantic Array	0	0	0	1.8
Islay	0	0	0.1	0.2
Solway Firth	0	0	0.1	0.2
North	0.4	0.2	0.7	0.6
East Orkney	0.1	0	0.1	0.3
Lewis	0.1	0.1	0.2	0.2
Kyle Rhea	0	0	0	0
Sound of Islay	0	0	0	0.1
Argyll Array	0.2	0.1	0.4	0.3
Hastings	0	0	0	0.2
IOW	0	0	0	0.4
Dogger Bank	0	0	0.1	1.6
South Dogger Bank	0	0	0	1.3
Norfolk Bank	0	0	0	1.3
Irish Sea	0	0	0.1	0.8
Strangford Lough	0	0	0	0.1
Shetlands	0.3	0.2	0.4	0.2

Appendix 7. The Joint Cetacean Protocol Project: current status, applications and limitations

March 2016

Paper written by Tim Dunn and Sónia Mendes on behalf of the JCP Steering Group

1. Summary

The JCP project has to date provided a platform for the integration of cetacean sightings, which comprise the largest collation ever attempted in Europe. The data derive from many different surveys and from several areas and time periods. They were standardised and corrected for several inherent biases by making a series of assumptions. However, the validity of some of these assumptions and of other potential sources of bias were not examined in detail. This may, therefore, need to be taken into consideration in any future analyses. In addition, the temporal and spatial paucity of the data has meant abundance estimates for many areas and time periods have wide confidence intervals. Notwithstanding these limitations, Phase III has allowed the production of maps that provide an indicative illustration of the average distribution and abundance of the most common cetacean species occurring in NW European waters between 1994 and 2010. Some of these could potentially be used to derive average densities and therefore numbers of animals of a particular species in an area. This could for example be applied when informing the relative importance of areas during Environmental Impact Assessment (EIAs) and when identifying areas which require survey effort to ensure appropriate marine spatial planning.

The JCP Phase III analysis indicated that trends in population size cannot be detected, even for species for which we have reasonably good time-series of data, except perhaps for dramatic changes, i.e. over 50% in 12 years. It is, therefore, unable to meet EC cetacean monitoring expectations and unable to provide the early warning signal necessary to allow timely management measures to be developed and implemented. It is however possible that, with the inclusion of additional sightings data-sets which increase spatial and temporal coverage, the power to detect trends in abundance could be improved. The continued collection and input of high quality data is therefore crucial to increase the applicability of the JCP project for conservation and management needs.

Despite its caveats and limitations, the JCP project and its outputs currently represent the best available data resource and illustration of the broad scale distribution of cetaceans in NW Europe. This project has provided the foundation for further statistical improvements and better standardisation of data collection and integration. It has supplied a common currency in terms of species abundance that will improve future environmental assessment, marine spatial planning and associated marine management.

DOS	**DON'TS**
• JCP distribution maps are an indicative illustration of species distribution and spatio-temporal variability. • JCP derived density/abundance estimates can be used in impact assessments when scaled to the total abundance of reference populations[1,2].	• JCP Phase III data are not suitable for detecting all but dramatic changes in population size. • JCP abundance estimates are not suitable to be used directly in EIAs. • Inferences made from the JCP analyses are unlikely to be reliable at scales of less than approximately 1000 km^2

2. Introduction

The Joint Cetacean Protocol (JCP) project collated and analysed data from both systematic visual surveys and other effort-related cetacean observations in the waters around the United Kingdom. Its aims were to:

1. identify relevant north-west (NW) European cetacean sightings data available;
2. investigate what power these data had to assess trends in abundance and changes in distribution and how to improve that power;
3. define standards for the collection and storage of sightings data;
4. facilitate sharing of standardised cetacean datasets via a web portal; and
5. develop methods for the production of cetacean distribution maps and estimates of both abundance and changes in abundance.

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[1] IAMMWG. 2015. Management Units for cetaceans in UK waters (January 2015). JNCC Report No. 547, JNCC Peterborough.

[2] For harbour porpoise, more recent analyses have produced density surfaces that should be used instead of JCP ones.

The results from the JCP project were expected to be used to:

1. inform reports to various European legislation especially the EU Habitats Directive and Marine Strategy Framework Directive;
2. assist with marine industry Environmental Impact Assessments (EIAs) by characterising the distribution and abundance of cetaceans in UK waters; and
3. initiate a rolling programme of data assimilation, analysis and sharing that would maintain and enhance the ability to address 1 and 2.

This paper summarises the project so far; assesses how the work carried out has addressed the project's objectives; and explores the applications and limitations of the outputs.

3. Project history and current status

Effort-related sightings data covering the period 1973 to 1999, from the Sea Watch Foundation (SWF), European Seabirds at Sea (ESAS) and Small Cetaceans in the European Atlantic and North Sea (SCANS) project of July 1994 were used to produce the *Atlas of Cetacean Distribution in North-West European Waters* (Reid *et al.* 2003). This represented the most up-to-date statement on the distribution of the 28 cetacean species occurring in NW European waters and also provided relative abundance estimates (animals per hour) for the ten most common species. The *Atlas* has been cited extensively and maps from it reproduced in many publications ranging from environmental assessments to scientific papers and reports. Data underlying the distribution and abundance maps have also been used by several organisations for further analysis and mapping.

In 2006, under JNCC's coordination, a working group was established to identify ways to update the Atlas (Annex 1). This led to the initiation of the JCP with the aim of delivering information on the distribution, abundance and population trends of cetacean species in European waters. This initiative was welcomed at the 2009 meeting of the Parties to the Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas (ASCOBANS) as potentially providing a means to realising international/ transboundary Favourable Conservation Status (FCS) reporting. In 2010, the JCP project acquired the additional aim of assessing whether these data could help with marine EIAs. Consequently, the Crown Estate joined the Project

Steering group in 2010 and marine renewable industry representatives joined in 2011 (see Annex I for membership of Steering Group and funders).

The JCP assembled datasets from all major UK sources, e.g. SCANS & SCANS II; the Cetacean Offshore Distribution and Abundance (CODA) survey; ESAS; SWF; and Atlantic Research Coalition (ARC). Data from other non-governmental and marine renewable industry sources were also included. This comprises the largest (both in time and space) cetacean sightings collation ever attempted in Europe, with 38 data sources from 542 distinct survey platforms (ship and aircraft) collected between 1994 and 2010 (full list of data sources in JCP Phase III report).

Thomas (2009) reviewed the potential of using the JCP data resource for detection of trends in species abundance and concluded that, whilst the European Commission's indicative threshold for a *large population decline* to be equivalent to a loss of more than 1% per year (ETCBD, 2011) was too small to be detectable for cetaceans, trends over longer periods may be detectable. The JCP analyses were subsequently conducted in three phases. **JCP Phase I** (Paxton & Thomas, 2010) aimed to standardise and combine a subset of datasets from the Irish Sea. It fitted regression models to estimate densities and identified spatial and temporal trends in abundance estimates. The models developed in Phase I were further refined and expanded to include the Scottish west coast in **Phase II** (Paxton *et al*, 2011). Density (animals/km^2) surfaces were generated for five species; however, some of the confidence intervals were very wide, especially in areas/times of low effort. The **Phase III** analysis corrected the data for under-detection and availability at the sea surface and went on to produce density surface maps over a large portion of UK waters (0-300m depth, see Phase III report) and period of data collection for seven species. In addition, statistically significant changes in abundance larger than 9% per year over 11.5 years (equivalent to two FCS reporting periods) were detected for some species. The final Phase III draft report was submitted in October 2012 and underwent an international peer review in early 2013. There were substantial comments which the contractors CREEM addressed to the Steering Group members satisfaction and which led to some changes to the report. The final version (Paxton et al., 2013) was submitted in May 2013 and is being published as a JNCC Report in March 2016[3].

4. Final output, uses and limitations

This section describes how the JCP project objectives were addressed and summarises the potential uses for the outputs as well as their limitations.

4.1. The objectives

Objective 1: identify all relevant NW European cetacean sightings data available.

A review of all current and planned cetacean survey effort undertaken by non-governmental organisations in UK waters was commissioned by JNCC in 2007 (Anderwald et al. 2007) and a further JNCC in-house review of cetacean survey effort in NW Europe then conducted. These were used to identify organisations which held data that were likely to meet a set of pre-defined JCP standards (see Annex 2). These organisations were contacted to request permission to use their data for JCP analysis and, if granted, that they convert their data to JCP standard format and run a series of data validation checks. Out of a total of 60 organisations contacted, 38 had data which achieved the JCP standard and were willing for them to be used in the analyses. Unfortunately, some additional datasets were not included due to the inability to reach a data sharing agreement, or insufficient resources to standardise the data, in time for inclusion in the analysis. Further work is now necessary to develop a strategic process which will ensure that the majority of high quality data, from NW Europe, are automatically standardised to JCP format and made available for specific analyses through long term agreements.

Objective 2: investigate what power these data have to assess trends in abundance and changes in distribution and how to improve that power.

Although a preliminary power analysis was performed in Phase I, it was based on only a subset of the data, and required further refinements. In Phase III it was estimated that

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[3] The unfortunate delay in publication was caused by a temporary decrease in staff resources at JNCC and other work taking priority.

the harbour porpoise population would need to experience a reduction of 90% or more over a period of two FCS reporting intervals (i.e. ~ 12 years) before a decline was detectable with a power of 80%. For the short-beaked common dolphin the detectable reduction would be 51% or more and somewhere in between for the other five species looked at (minke whale, bottlenose dolphin, risso's dolphin, white-beaked dolphin and Atlantic white sided dolphin). This corresponds to an average annual decline of between 6% (common dolphin) and 18% (harbour porpoise). Contrary to what might otherwise be expected due to their abundance, the ability to detect trends was worse for harbour porpoise. This is because the data were able to support a more complex and realistic model, incorporating spatio-temporal interactions, which allowed temporal trends to vary over space. However this resulted in relatively high uncertainty in the predictions and therefore low power. It is clear from Phase III results that the existing JCP dataset does not allow for the detection, with good statistical power, of any but the most dramatic changes in abundance.

In order to improve the power to detect trends, more data would be required that would improve temporal and spatial coverage. It is possible that with the addition of more existing datasets smaller changes could become detectable. The JCP Phase III report authors recommended that an alternative approach to the one used in Phase III might render better results. Rather than setting the power to detect at 0.8 (with $\alpha = 0.05$) and then asking what is the minimum average annual change detectable, one would choose the rate of population change that is considered to be biologically relevant. The power to detect such a rate could then be explored and it could be determined whether this would allow for useful conservation assessment within defined timescales and satisfy requirement for FCS reporting.

Objective 3: define standards for the collection and storage of sightings data;

Annex 2 sets out the current data standard that must be met for data to be included in the JCP. This standard accompanies a set of spreadsheets of ideal data structure (available on request from JNCC) which were also used for submission of data for JCP analysis. An online data validation tool (http://www.ruwpa.st-and.ac.uk/dpwebi/jcp/) was used to check all data-sets prior to their inclusion in the analyses. The production and adoption of standardised methods for the collection of cetacean sightings data is also essential to improving the future power of any JCP data resource.

Objective 4: facilitate sharing of standardised cetacean datasets via the www;
The JCP project was successful in unlocking the potential strength of cetacean sightings data from a wide range of organisations for a specific set of analyses. It demonstrated that there was a willingness to share cetacean data for certain purposes and that a long-term sharing agreement and combined data resource could become a reality. JNCC intend to garner the opinions of existing and potential contributing organisations to determine whether such an agreement can be established.

Objective 5: develop methods for the production of cetacean distribution maps and estimates of both abundance and changes in abundance.

Species distribution

The JCP data resource covers a variety of survey types and methods. By correcting for under detection (not all animals at the sea surface are detected) and availability at the sea surface (not all animals are at the surface) in Phase III, the different data-sets were standardised as much as possible and used collectively in the density modelling that followed. The JCP report authors acknowledge that there are sources of potential bias that it was not possible to take into account in the analyses (e.g. differences in individual observers' ability to detect cetaceans). If these are influential sources of bias then some of the spatial/temporal patterns observed could be driven by them rather than real differences in animal density. Further research is required on possible ways to address this. JCP Phase III results should therefore be considered indicative rather than an accurate representation of distribution.

The modelled distribution predictions illustrated in JCP Phase III maps faithfully reflected the patterns in the input data for all species. The output maps that will be of most interest (Appendix 4) show the predicted mean summer densities from 2007 to 2010, the most recent time period of the data resource. These represent averages of distribution predicted for those years and an indication of the range of abundances that can be expected for particular areas. Despite the huge size of the dataset, survey effort is distributed very patchily with significant gaps in both space and time. Therefore, any inferences should only be made in the context of the available survey effort for the area (e.g. in areas of low effort only limited conclusions can be derived from patterns shown); the concordance of the modelled density patterns to the input observed densities; and the confidence intervals (all of which can be assessed by comparing the four maps provided for each species in Appendix 4). The JCP Phase III report authors also warn of the fact that any inferences made from the analyses are unlikely to be reliable at scales of less than approximately 1000 km^2 (e.g. an area of 31.6 x 31.6 or 10 x 100 km^2).

Abundance estimates

The models generally produced abundance estimates over large areas that were broadly similar, although usually higher and of greater uncertainty, than estimates from SCANS-II. The JCP report authors highlight that, given the patchy distribution of the JCP data resource and assumptions that had to be made to render its datasets comparable, the estimates of abundance from Phase III are less reliable than those from well-designed dedicated abundance surveys such as SCANS or CODA. The SCANS II and CODA estimates were used by the UK Interagency Marine Mammal Working Group when agreeing on cetacean Management Units[4] (MUs). The estimated abundance of animals in these MUs are currently considered as the reference populations for relevant cetacean species.

It is, therefore, imperative that JCP Phase III abundance outputs are used in the context of the currently agreed species-specific MU reference populations and are not used directly. JCP abundance outputs had therefore to be adjusted so that their total contained within each MU is equal to the respective reference population estimate. To facilitate this for areas of conservation and marine development interest, an appendix to the JCP Phase III report (Appendix 6) was produced by CREEM during 2014/2015, with the estimates of the proportions of total JCP abundances for each species for each of those areas[5]. These proportions can then be multiplied by the scaled reference population abundance of each MU which will be provided on the JNCC website. The JCP main outputs (adjusted average summer density surfaces for

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[4] IAMMWG. 2015. Management Units for cetaceans in UK waters (January 2015). JNCC Report No. 547, JNCC Peterborough.

[5] In the JCP Phase III analyses the abundance estimates apply only to areas from 0-300m depth, and for harbour porpoise, areas to the east of easting 820000 were excluded, due to the great uncertainty in the estimates arising from the small amount of data available for those areas.

years 2007-2010) will also be made available on the JNCC website to allow extraction of abundances (with confidence intervals) for any given area.

Harbour porpoise density surfaces

In 2013, the majority of the standardised JCP data resource (with a couple of additional datasets) was used in an analysis, carried out by DHI Water Environment Ltd. under contract to JNCC, to help identify discrete and persistent areas of high harbour porpoise density in the UK marine area[6]. This work underpinned the identification of potential harbour porpoise Special Areas of Conservation. The DHI modelling approach differed from JCP Phase III in that models were fitted to each Management Unit, allow their different harbour porpoise habitats to be taken into consideration by using several habitat predictors (e.g. features thought to influence prey availability such as oceanographic fronts and substrate type) as explanatory variables. This is in contrast with the JCP Phase III modelling approach which used a small number of explanatory variables, mainly time and latitude/longitude, to predict cetacean abundance and distribution.

The harbour porpoise density surfaces generated by the Phase III analysis and those generated by DHI work are broadly similar although in some areas there are localised differences, particularly in areas where survey effort was low. In these areas, the JCP predictions are influenced by the number of porpoises seen during limited survey effort areas. In contrast, DHI predictions are driven more by associations between observed abundances and habitat characteristics within the respective MU. DHI predictions are, therefore, more likely to provide a realistic picture of abundance in areas where effort is low. JNCC has considered the two outputs in the context of their future potential use for environmental impact assessments and concluded that the DHI density surfaces better represent the expected distribution and abundance of harbour porpoise for any given area of interest and should, therefore, be used preferentially.

JNCC must now seek agreement from data providers to make the DHI density surfaces available for wider use.

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[6] Heinänen, S. & Skov, H. 2015. The identification of discrete and persistent areas of relatively high harbour porpoise density in the wider UK marine area, JNCC Report No. 544, JNCC, Peterborough. 108pp. Available from http://jncc.defra.gov.uk/page-6991

4.2. JCP outputs: uses and limitations

1. To assist with status reporting, especially for the Habitats Directive and Marine Strategy Framework Directive;

The European Commission's guidelines for conservation status reporting (ETCBD, 2011) suggest an indicative threshold for a *large population decline* would be equivalent to a loss of more than 1% per year (within a period specified by the Member State, e.g. FCS reporting period) in combination with abundance below 'favourable reference population' or equivalent to abundance more than 25% below the 'favourable reference population'[7]. These describe very significant and unacceptable levels of decline if occurring beyond natural variation and the long-term consequences of these will vary between species.

Precise measures of abundance, particularly for cetacean species, are very expensive to obtain, hence the appeal of using existing data from multiple sources and funding streams such as those collated under the JCP. However, the relatively poor spatial/temporal coverage of the current JCP Phase III data resource means that it only has the power to detect large-scale trends in population. Previous assessments of monitoring schemes (e.g. Taylor *et al*. 2007) concluded that to reliably detect even precipitous declines in cetacean population abundance, much greater amounts of survey effort than are routinely undertaken were required. So, rather than this being a JCP specific problem, this is considered a general issue with cetacean abundance monitoring. Nevertheless, the JCP Phase III approach may allow large-scale trends to be detected in the absence of a more intensive programme of monitoring. However, it cannot currently provide an effective early warning trigger for conservation action (i.e. implement measures to reduce a threat or halt a population decline), nor is it likely to meet EC expectations regarding surveillance in its current form. Further advice provided by JNCC on monitoring cetacean options is currently being discussed with governments.

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[7] The 'favourable reference population' is at least the size of the population when the Directive came into force, i.e. 1994.

2. To assist with Marine Industry EIAs in UK waters by characterising the distribution and abundance of cetaceans around proposed developments;

The JCP Phase III outputs can add value to industry impact assessments (e.g.offshore renewables and oil & gas developments) and assist with government processes (e.g. marine planning) as these may need to characterise the distribution and abundance of cetaceans in order to predict potential impacts from proposed developments on species. The JCP species' maps, which depict long-term average spatial distribution patterns, are likely to provide a more realistic 'picture' of the range of densities that can be expected in any given area, at a medium to large scale, than those obtained from surveys carried out during one or even two years at a smaller scale[8]. Although the modelled spatial distributions may not be accurate for any particular moment in time, it is appropriate to use an average and representative range of possible densities in impact assessments, given the large natural temporal and spatial variability in cetacean abundance.

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[8] Although see discussion above on harbour porpoise density surfaces.

In the context of environmental impact assessments, particularly cumulative assessments, the Phase III outputs *can* be used in the following ways:

- To assess species' distribution patterns at large spatial scales (no less than 1000km^2) and place the predicted relative abundance of a particular area in the context of their wider abundance and distribution.
- To provide (adjusted) abundance estimates with confidence intervals, at the scale of conservation/development interest areas in UK waters (see Appendix 6 and JNCC website). These figures should be used instead of/or in addition to the SCANS II block densities and to supplement and strengthen a developer's own survey work and assessment.
- To explore how distribution patterns may have changed in the period analysed (1994 to 2010) for harbour porpoise only. These temporal changes should only be used to provide context and not for inferring impacts of particular activities.
- To assist in assessing whether there would be added value in carrying out site characterisation cetacean surveys for particular areas of interest. Areas of little effort are obvious candidates as well as areas where there is greater uncertainty in the estimates.
- To provide abundance estimates for assessments of potential cumulative impacts from offshore industries across a species range within a management units. This is of particular value because data from different regions are comparable whereas it would not be appropriate to use several developer estimates together in one cumulative impact assessment without first standardising the data and correcting for biases, as has been done for the JCP data resource.
- To provide an indication of which development areas are more likely to have higher abundances of particular species and therefore the most suitable locations for detecting the effects of certain activities through monitoring and surveillance.
- To assist in standardising and quality controlling the collection and storing of effort-based cetacean observation data. This facilitates the collation of data from multiple sources in order to be used collectively in any future analyses.

In the context of environmental impact assessments, the Phase III outputs *cannot* be used in the following ways:

- Without the reference population adjustment (see Appendix 6), the outputs should not be used to infer abundance at any scale. In standardizing the data from so many sources, strong assumptions were made about factors such as detection probability. This results in estimates of abundance that cannot be as reliable as those coming from a well-designed dedicated abundance survey.
- To infer adjusted abundance at a finer scale than 1000km^2. This is due to issues inherent to the data (assumptions that may not hold and spatio-temporal patchiness) and the spatial smoothing used, making reliable estimation at fine scales problematic.
- To infer abundance etc. in areas with little effort. The JCP extrapolation into regions of low or zero effort leads to great uncertainty in the estimates.
- To infer temporal changes in abundance for the smaller regions of conservation/development interest for any species but perhaps the harbour porpoise.
- To replace, for certain particularly sensitive areas or areas for which little survey effort exist, the need to carry out cetacean surveys.
- To provide baseline data for impact monitoring of short term change i.e. being able to establish cause and effect between for example, pile driving noise and trends in local abundance. To be able to detect change and establish cause and effect, surveys would need to be designed for that specific purpose.

3. To initiate a rolling programme of data assimilation, analysis and sharing that would maintain and enhance the ability to address the above requirements;

The JCP project has so far provided the foundation for a rolling programme of data assimilation, analysis and sharing to be designed in the future once agreement on the future of the JCP has been discussed amongst the data providers and stakeholders.

5. References

Anderwald, P., Ansamann, I., Baines, M., Baulch, S., Evans, P.G.H., Nuuttila, H., & Pesante, G. (2007). Cetacean monitoring effort carried out by voluntary NGOs in UK waters up to 2007. JNCC Report No 480. http://jncc.defra.gov.uk/pdf/480.web.pdf

European Topic Centre on Biological Diversity 2011. Assessment and reporting under Article 17 of the Habitats Directive: Reporting Formats for the period 2007-2012. https://circabc.europa.eu/sd/a/5c427756-166d-4cc8-a654-fca8bfae3968/Art17%20-%20Reporting-Formats%20-%20final.pdf

IAMMWG. 2015. Management Units for cetaceans in UK waters (January 2015). JNCC Report No. 547, JNCC Peterborough.

Paxton, C. G. M & Thomas, L. 2010. Phase One Data Analysis of Joint Cetacean Protocol Data. Report to Joint Nature Conservation Committee Contract number C09-0207-0216. http://jncc.defra.gov.uk/pdf/JCP_Phase_1_Analysis.pdf

Paxton, C. G.,M., Mackenzie, M., Burt, M. L., Rexstad, E and Thomas, L. 2011. Phase II Data Analysis of Joint Cetacean Protocol Data Resource. Report to Joint Nature Conservation Committee Contract number C11-0207-0421. http://jncc.defra.gov.uk/pdf/JCP_Phase_II_report.pdf

Paxton, C.G.M., Scott-Hayward, L., Mackenzie, M., Rexstad, E. & Thomas, L. 2013. Revised Phase III Data Analysis of Joint Cetacean Protocol Data Resource. Draft report to The Joint Nature Conservation Committee. Contract number C11-0207-0421.

Reid, J.B., Evans, P.G.H. and Northridge, S.P., 2003. Atlas of cetacean distribution in north-west European waters. Joint Nature Conservation Committee, Peterborough.

Taylor, B. L., Martinez M., Gerrodette, T., Barlow, J. and Hrovat, Y. N. (2007). "Lessons from monitoring trends in abundance of marine mammals." Marine Mammal Science **23**(1): 157-175.

Thomas, L. 2009. Potential Use of Joint Cetacean Protocol Data for Determining Changes in Species' Range and Abundance: Exploratory Analysis of Southern Irish Sea Data. Report to Joint Nature Conservation Committee Contract No: F90-01-1208. http://jncc.defra.gov.uk/pdf/JCP_Prelim_Analysis.pdf

Annex 1

Current Membership and funding of the JCP Steering Group

Organisation	**Members**	**Joined**	**Funding provided**
JNCC	Tim Dunn, Mark Lewis, Sonia Mendes, Eunice Pinn, Kelly MacLeod	2006	Yes
Department of Environment Northern Ireland	Gary Burrows (until 2014), Stephen Foster	2006	No
Natural Resources Wales	Tom Stringell	2006	Yes
Sea Mammal Research Unit	Simon Northridge	2006	No
Sea Watch Foundation	Peter Evans	2006	No
National Parks & Wildlife Service	Eamonn Kelly, Oliver O'Cadhla	2006	Yes
University College Cork	Emer Rogan	2006	No
Irish Whale and Dolphin Group	Simon Berrow, Dave Wall	2009	No
The Crown Estate	Jessica Campbell	2010	Yes
Source Low Carbon LLP	Zoe Crutchfield	2011	No
RES Offshore	Gero Vella	2011	No