# A first approach to deal with cumulative effects on birds and bats of offshore wind farms and other human activities in the Southern North Sea

Mardik F. Leopold (IMARES), Martijn Boonman (Bureau Waardenburg), Mark P. Collier (Bureau Waardenburg), Nara Davaasuren (IMARES), Ruben C. Fijn (Bureau Waardenburg), Abel Gyimesi (Bureau Waardenburg), Job de Jong (Bureau Waardenburg), Ruud H. Jongbloed (IMARES), Bob Jonge Poerink (The Fieldwork Company), Jonne C. Kleyheeg-Hartman (Bureau Waardenburg), Karen L. Krijgsveld (Bureau Waardenburg), Sander Lagerveld (IMARES), Rob Lensink (Bureau Waardenburg), Martin J.M. Poot (Bureau Waardenburg), Jan Tjalling van der Wal (IMARES) & Michaela Scholl (IMARES)

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IMARES Wageningen UR

(IMARES - Institute for Marine Resources & Ecosystem Studies)

Client: Rijkswaterstaat
WVL afdeling Waterkwaliteit en Natuurbeheer
Maarten Platteeuw
Postbus 17, 8200 AA Lelystad

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P.O. Box 68
1970 AB IJmuiden
Phone: +31 (0)317 48 09 00
Fax: +31 (0)317 48 73 26
E-Mail: imares@wur.nl
www.imares.wur.nl

P.O. Box 77
4400 AB Yerseke
Phone: +31 (0)317 48 09 00
Fax: +31 (0)317 48 73 59
E-Mail: imares@wur.nl
www.imares.wur.nl

P.O. Box 57
1780 AB Den Helder
Phone: +31 (0)317 48 09 00
Fax: +31 (0)223 63 06 87
E-Mail: imares@wur.nl
www.imares.wur.nl

P.O. Box 167
1790 AD Den Burg Texel
Phone: +31 (0)317 48 09 00
Fax: +31 (0)317 48 73 62
E-Mail: imares@wur.nl
www.imares.wur.nl

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## Contents

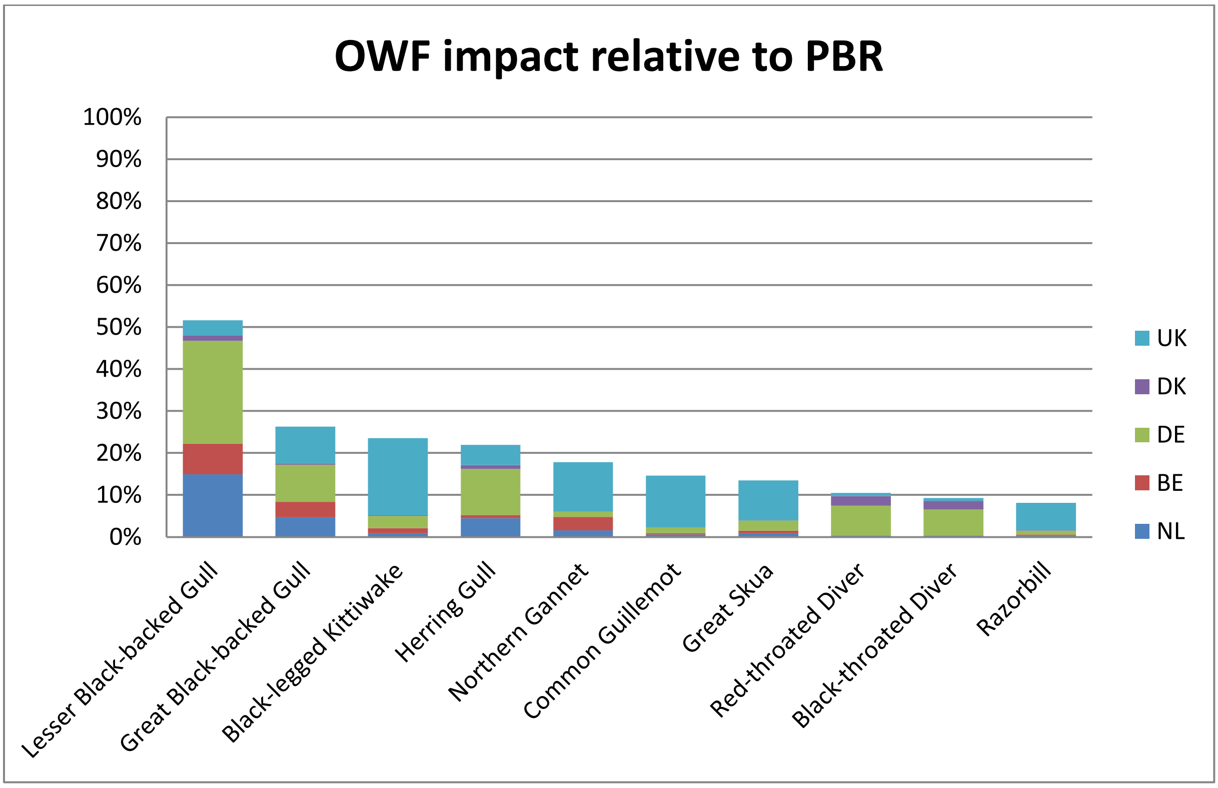
# 1 Summary

Renewable energy is an increasing demand, and governments of North Sea countries are looking at developing offshore wind farms to help meet sustainability demands. The first at-sea wind farms have become operational in several countries, or are under construction, but many more are on the drawing board. Altogether, around 100 offshore wind farms are scheduled to be operational by 2023 in the southern North Sea (51-56°N) alone. There may be two sides to this development in environmental terms: on the one hand this will help reduce $CO_2$ emissions, on the other hand protected North Sea biota may be negatively impacted. This report considers the cumulative impact of all projected wind farms in the southern North Sea (by 2023) on birds and bats.

Birds and bats have flight in common, and any animal flying over the North Sea may collide with rotor blades in (future) offshore wind farms. This will lead to increased mortality. Mortality rates will depend on the numbers of animals in the air, at rotor height, moving through the wind farms, and their behaviour while doing so. The impact of collisions on the population level will depend on the relative population size (to the number of casualties) and the regenerative power of the species concerned. North Sea seabirds also use the area as their habitat and may suffer additional mortality through habitat loss or habitat degradation. Space taken up by offshore wind farms may be avoided by seabirds, and this loss of habitat may lead to additional loss of fitness. It has been assumed that 10% of the seabirds that are displaced by offshore wind farms, will die. We note that loss of life through habitat loss may be structural, in that the carrying capacity of the southern North Sea will be permanently decreased, leading to higher stress on the seabirds that rely on this habitat.

The combined, cumulative effects of collisions and displacement, have been modelled for all wind farms considered operational in 2023 in the southern North Sea, using the method recently proposed by Bradbury *et al*. (2014). We have extended this method to be able to predict numbers of birds killed directly from collisions, and indirectly from displacement. Total numbers of birds estimated to die remain below PBR for all species of seabirds commonly occurring in the North Sea.

Predicted numbers of casualties, relative to these latter factors have been compared in Potential Biological Removal (PBR) models. It is assumed that if the cumulative number of casualties (of all wind farms) remain under PBR, the birds (or bats) killed will be replaced and the population size will not decrease because of offshore wind farm development. The results of this modelling exercise shows, that total predicted mortalities in all seabird species will remain within the limits of PBR. This would imply that no bird species will go extinct because of the development of offshore wind alone. Eight seabird species have predicted mortalities that are 10% or more of PBR: Lesser Black-backed Gull (52%), Great Black-backed gull (26%), Black-legged Kittiwake (24%), Herring Gull (22%), Northern Gannet (18%), Common Guillemot (15%), Great Skua (13%) and Red-throated Diver (10%), while two more come close to this figure: Black-throated Diver (9%) and Razorbill (8%). For all other species, predicted wind farm related mortality rates are below 5% of PBR (see Table 5.1 in this report).



*Cumulative effects of all projected wind farms in the southern North Sea (as envisaged by 2023) for the ten species, for which the wind farm mortality value is closest to their respective PBR values (scaled at 100%) and the origin of the involved national exclusive zones.*

The collision-part of the extended Bradbury-method was cross-checked. Cumulative numbers of collisions were also estimated by using the Band (2012) method. Model outcomes of this routine also largely predicted mortality rates below PBR. However, with this method, higher and in some cases much higher mortality rates were predicted, which in two species exceed PBR: Lesser Black-backed Gull (313% of PBR) and Great Black-backed Gull (131%), which would mean that collisions alone would lead to extinction of these species in the southern North Sea. Using the Band (2012) method also resulted in high predicted mortality rates from collisions in European Herring Gull (81% of PBR), Northern Gannet (50%), Black-legged Kittiwake (36%), and Great Skua (10%), while two more species had predicted mortality rates exceeding 5% of PBR: Common Eider (8%) and Sandwich Tern (6%; for the full list see Table 4.23 in this report).

Mortality rates, resulting from collisions with offshore wind farms were also estimated with the Band (2012) method for land- and waterbirds (from freshwater habitats) that commonly migrate across the southern North Sea (see Table 4.24 in this report), from known population sizes and migration routes, and compared to their specific PBR values. None of these species was predicted to suffer a cumulative mortality above PBR, but high values were found for Eurasian Curlew (60% of PBR), Black Tern (52%), and Tundra Swan (also known as Bewick's Swan: 44%), while notably high figures were also found for Sanderling (21%), Common Starling (12%), Red Knot (11%) and Bar-tailed Godwit (6%).

The Band model appears to be highly sensitive to the numbers of birds assumed to be flying through the wind farms. For some of the seabirds, unrealistically high numbers were possibly generated for some future wind farms, on the basis of at-sea count data. These had to be extrapolated over wider areas and peak counts, from e.g. concentrations of gulls and Northern Gannets around fishing vessels were a possible cause. Similarly a count of a flock of migrating Common Eiders over a spot that was chosen for a future wind farm, generated a high local density for this species in that particular future wind farm, leading to a high predicted mortality rate. This explanation, however, is not valid for migrant birds, such as Black Tern, Tundra Swan, the waders mentioned above and the Common Starling, for which no at-sea survey data were used as model input. We also note that wind farm related mortality should be seen in concert with other mortality factors, and high mortality rates from collisions alone, in comparison with PBR, are worrying.

All predicted mortality rates, at this stage, are only model predictions. The displacement part of predicted seabird mortality is still highly uncertain and could not be cross-checked by another model. We could only model displacement in relation to wind farm configuration in Common Murre, the species for which comparable data are available from several offshore wind farm impact studies. Displacement varied between wind farms, in relation to turbine density. For many future offshore wind farms, the turbine density is not yet known, and more data will be needed to explore this in full, also for other species of seabirds.

Our modelling exercise did, however, identify species that would seem to be at risk. Most of these are already closely monitored: the seabirds in their breeding colonies, the Tundra Swan at its wintering and staging quarters, and the waders in tidal basins around the North Sea. We recommend that populations remain closely followed, now also in the light of offshore wind farm development. Given that the circa 100 wind farms considered in this report will not be built overnight, population trends of the various bird species identified here as vulnerable can be followed and compared to increasing mortality rates from offshore wind farms, as progressively more projects become operational. Following developments closely would allow adjustment of the development of offshore wind, should mortality rates become unacceptably high.

For bats at sea, far less information is available than for birds, but the same general rules apply. We note that several species have been regularly identified flying over the North Sea, or may be expected to do so, by extrapolation. However, the sizes of populations likely to be impacted are very imperfectly known, as are bat numbers at sea and their offshore behaviour. Several species may be impacted negatively by offshore wind farm development in the southern North Sea, most notably the Nathusius’ Pipistrelle, while Particoloured Bat and Noctule would also seem to be vulnerable. For bats, increased monitoring, particularly at sea, is required to get to grips with the possible problem at hand.

## 2 Introduction

In the Dutch Energy Agreement for Sustainable Growth (also called *SER-akkoord*[1]), parties agreed to realize 4,450 MW of offshore wind energy by 2023, possibly increasing to 6 000 MW in subsequent years. To achieve this, the Dutch government wants to put to contract (as of 2015) a total of 3,450 MW of offshore wind, which corresponds to approximately 10 new offshore wind farms (OWF) on the Dutch Continental Shelf (DCS).

Environmental impact studies (EIS), carried out so far, concentrated on (possible) effects of individual wind farms. The cumulative effects of all existing and planned wind farms on the DCS, and wind farms in other parts of the southern North Sea together, are insufficiently investigated; let alone, all these effects in combination with impact factors other than wind farms. However, knowledge of cumulative effects is essential when considering the draw up of an overall plan that regulates the issuance of lots, i.e. individual tenders for certain prespeciefied locations, taking into account ecological interests.

To gain more insight into the subject of cumulative effects, the Ministry of Economic Affairs has asked Rijkswaterstaat to set up a project called 'Assessment framework ecology and accumulation of effects, 3rd Round Offshore Wind’. This project consists of several subprojects. IMARES has been commissioned to carry out the subproject on 'birds and bats’, called: Building blocks for dealing with cumulative effects on birds and bats of offshore wind farms and other human activities in the Southern North Sea.

## 3 Assignment and reading guide

The assignment was in accordance with IMARES’ quotation no. 14.43.072, covered by the RWS-framework agreement ‘*Specialistische adviezen Stichting Dienst Landbouwkundig Onderzoek ten behoeve van het Ministerie van Infrastructuur en Milieu’*.

To further define the requested ‘building blocks’ of the project, two meetings with the commissioner were organized. One on 20 June 2014 between Maarten Platteeuw (RWS) and Michaela Scholl, Mardik Leopold, Jan Tjalling van der Wal, Ruud Jongbloed and Sander Lagerveld (the IMARES team) on the island of Texel, and another one on 24 June 2014 with Maarten Platteeuw (RWS), Mardik Leopold and Ruud Jongbloed, and Martin Poot and Jan van der Winden (both Bureau Waardenburg) in Rijswijk, the Netherlands.

The points of discussion were the client’s requirements that the instruments to be developed should be in line with the concepts of other research groups working on the subject of cumulative effects, and that the IMARES-approach has the capability to take other offshore activities than offshore wind energy, for example shipping, into account. After weighing the pros and cons of different concepts such as Harmony and ODEMM, the CUMULEO framework (see Annex A) was chosen. This concept provides a high degree of flexibility on the input as well as the output side to link to other concepts and subjects.

To secure a project result that can stand the criticism of experts outside the Netherlands, it was decided at an early stage to adopt the approach of Bradbury et al. (2014) regarding seabird sensitivity for offshore windturbines and shipping, and customize it for the purpose of our project. Furthermore, we deliberately sought collaboration and informal collegial contact with third parties: Dr Alain Zuur (Highland Statistics Ltd, Newburgh, United Kingdom), an expert on the statistical treatment of seabirds at sea

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[1] https://www.ser.nl/en/publications/publications/2013/energy-agreement-sustainable-growth.aspx

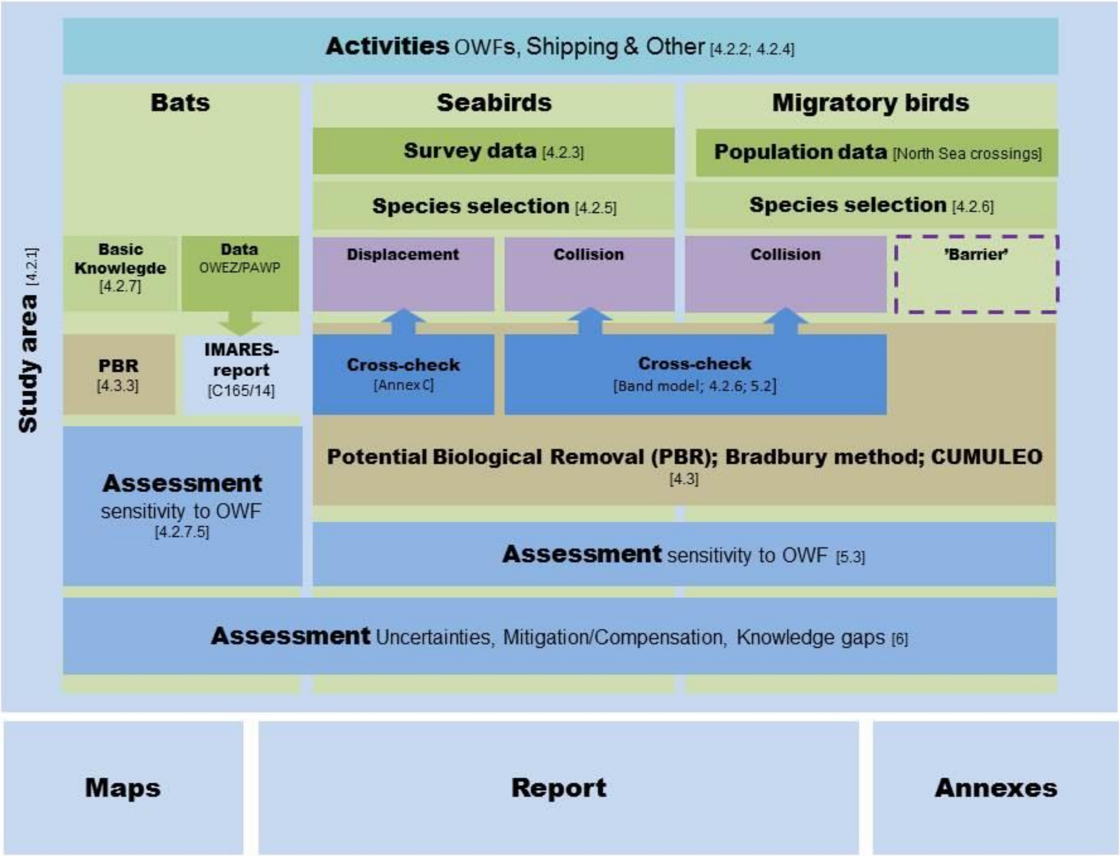
distributions in relation to offshore wind farms, and Lothar Bach (Bach Freilandforschung in Bremen, Germany), an expert on bats (at sea).

The approach concerning the type of data and the methods in our study can be summarised in the following steps:

1. The distribution and density of seabird species, offshore windfarms and shipping in the southern North Sea is contained in GIS databases.
2. The sensitivity of seabird species for OWF (first tier collision mortality and displacement mortality) and for shipping (first tier displacement mortality) is taken from the method and data developed by Bradbury *et al*. (2014) and subsequently modified.
3. The sensitivity (second tier collision mortality) of seabirds for OWF is calculated with the SOSS Band model (Band 2012).
4. The Potential Biological Removal criterion and the Ornis committee criterion for seabird populations are derived.
5. The cumulated impacts on seabird populations of OWF and OWF in combination with shipping is calculated with CUMULEO. For that purpose the input from points 1, 2 and 4 is used as input, combined and processed. More information about the CUMULEO framework can be found in Annex A.
6. The collision mortality of seabirds caused by OWF is independently estimated by using the SOSS Band model (Band 2012) for the purpose of validation of the collision mortality results of CUMULEO (point 5).

The next chapter (4) starts with an explanation of the general approach chosen and presents the so-called building blocks as shown in Figure 3.1 (quantitative and qualitative data/information, methods and validations) in more detail. These building blocks were partly brought together by applying the CUMULEO-framework. A general description of this framework is given in Annex A. In chapter 5 we show how we applied the extended Bradbury and PBR methods within the CUMULEO framework to quantitatively assess the wind farm sensitivity of the species selected by us. Also, the methods that we used for calculations related thereto are explained here. In chapter 6 we conduct an overall assessment, by indicating and discussing the uncertainties surrounding our results. We point out the possibilities of mitigation and compensation of the potential identified impact of offshore wind farms on birds and bats, and end this chapter with addressing the most important knowledge gaps. Chapter 7 summarizes the conclusions, some of which already emerge from the context in chapters 5 and 6.

*Figure 3.1. Blocks set: lay out of the subjects and issues covered in this project.The arrangement of the blocks indicates the steps taken.The attached numbers refer to the chapter(s) where the different 'blocks' are described.*



# 4 Building blocks

## 4.1 General approach

This report provides the first quantitative assessment of the cumulative effects of a great many offshore wind farms in the southern North Sea on birds and bats. Both seabirds, that use the southern North Sea as their habitat, and other species that only migrate through the area, may be impacted.

Several building blocks are needed to consider the impact, arising from offshore wind farms spread out over a large area, on this variety of bird and bat species. For this project, the following data were made available and then reprocessed.

Data/information

- Co-ordinates and GIS data to define the study area.
- Timeline: until 2023
- Data on offshore wind farms, both existing and planned: (intended) location, area covered, (intended) turbine types, including associated characteristics such as rotor diameter, hub height, and foundation type. Where available, data on positions of substations are included. In addition, data were received on development status (licensed, under construction, operational), and expected development schedule, including year of construction start, first power supply and fully operational status. N.B.: Specific offshore wind farm configurations (i.e. arrangement of wind turbines) are not considered in this study, since these are as yet unknown. Moreover, there is insufficient knowledge of the variation in effects of different configurations (Krijgsveld 2014). However, for one species, the Common Murre the effect of wind farm configuration has been explored (Appendix C).
- Information on the bird and bat species considered. Guiding questions were: which species occur in the study area in significant numbers, which species are (most) at risk? Taking into account occurrence, behaviour and ultimately the potential species specific impact of offshore wind turbines, not all species that occur in the area, or might occur, are equally relevant. Therefore, a selection of species to be considered in this report, is made and the rationale for this selection is given. For the selected species, existing information on their distribution over the study area is compiled, together with information on their (presumed) vulnerability/sensitivity to offshore wind farms. Note that only a small number of offshore wind farms have been built to date and that only few impacts have been studied. In particular, basic information is missing for bats. In many cases, information from the few studies had to be extrapolated to e.g. more offshore areas in the southern North Sea (e.g. the Dogger Bank area), where wind farms are as yet non-existent, but planned for the near-future.
- Data on shipping intensity, since it has impact on the presence of bird species.

Methods

- Risk/sensitivity assessment method (collision and displacement) following Bradbury *et al.* (2014) (called extended Bradbury method). A common risk to all animals that travel through air, is that wind turbines pose a risk of collision. Collision with the rotor blades will generally be lethal, so the risk is increased mortality. Seabirds that use the area are subject to a second risk: that of habitat loss, or habitat degradation, if they will not, or will to a lesser extent, enter offshore wind farms. Such (partial) habitat loss will equal direct mortality if, and only if, the displaced birds cannot find another place to go and survive there equally well. This will be the case if the total number of birds is at carrying capacity. As long as wind farms are relatively few in numbers, displaced birds will probably be able to move to other parts of the sea, and survive. However, around 100 offshore wind farms are on the drawing board, in the southern North Sea alone. In this report, we consider the stage of offshore wind development by the year 2023, with all planned wind farms built and operational. In this scenario, the effects of displacement are as yet unknown, and impossible to estimate with current knowledge, but probably above zero. In the analyses in this report, we follow the suggestion given by Bradbury *et al.* (2014) to use a factor of 0.1 (or 10%) for mortality of displaced birds. Bradbury *et al.* (2014) have recently published a method to estimate the combined effects of collision mortality and displacement mortality on seabirds confronted with offshore wind farms, albeit in a relative way. For the present study, we have extended their methods by introducing a scaling factor, which allows us to estimate absolute mortalities, per seabird species and per individual wind farm based on quantitative information on densities of seabirds. The Bradbury method is based on a number of factors considered influential, that are weighted by expert judgement, based on an extensive literature review. Their method is an elegant one, because of the underlying expertise and because it may evaluate the cumulative risks of collisions and displacement for all seabird species that commonly occur in the southern North Sea. The only input needed is an estimate for bird density, across the year, for the site to be considered and these data are availble from offshore seabirds surveys. Ideally, both components of the Bradbury estimate for offshore wind farm induced seabird mortality (caused by collision and by displacement) should be validated, or at least cross-checked against an alternative estimation method.

- Validation of risk assessment; extended Bradbury method versus Band model (collision risk). The mortality component from collisions can also be estimated with the SOSS Band model (Band 2012, see paragraph 4.2.6.4 of this report). This model is widely used to estimate the number of bird victims at offshore wind farms, and also is embedded in extensive expert judgement. The Band model uses a different approach than the Bradbury method and uses the rotor-swept area (i.e., turbine size) and the flux of birds moving through that area, in combination with estimates of how well birds are able to avoid being hit. Both methods have been applied to the available data and we compare the outcomes. We note, however, that neither model has been rigorously tested in a marine field situation, by good independent detections of birds being hit. The Band model has been rather extensively used in wind farm studies, while the Bradbury approach has only just been published (September 2014) and the necessary extension to this model to estimate absolute numbers of collision victims has been developed for the present study and has not yet been evaluated by others. The extended-Bradbury approach, as proposed in our study, has therefore received less peer scrutiny than the Band model. However, both models have been developed by experts in their fields and even without validating field data on actual numbers of collision victims, should be regarded valuable. Still, it might be expected a priori, that the outcomes of the two models will differ. Whether the differences will be structural or random across species and wind farms will be evaluated in this report. The Band model is the only approach possible to estimate mortalities for non-seabirds (migrants over the North Sea), and is also the most appropriate for these. Habitat loss from displacement is no issue for these birds. The same applies to migrating bats, should there be sufficient data.
- Validation of risk assessment (displacement); (Annex C). No comparison with results from an independent method is yet possible for the displacement component within the Bradbury mortality estimates. The few published studies on displacement provide different values for the displacement factor: the change of bird density within wind farm perimeters. Also, such factors are not yet known for all seabirds concerned. To explore this further, we have asked Dr Alain Zuur (Highland Statistics Ltd, Newburgh, United Kingdom) to statistically compare the results of available studies for one of the most abundant and wide-spread seabirds in the North Sea, that is also known to be displaced by offshore wind farms: the Common Murre (*Uria aalge*). We have explored the possibility that the density of turbines in a wind farm may influence the displacement factor: this may help us to understand the different results of various impact studies.
- Method to assess impact on population level of species; Potential Biological Removal (PBR). Weighing the mortality estimates will be done by comparing absolute numbers of casualties with the size of the impacted population, and its regenerative power. This is done by using Potential Biological Removal models (see: Watts 2010; Poot *et al.* 2011a; Sugimoto & Matsuda 2011; Bellebaum *et al.* 2013), that estimate the number of birds (or bats) that might be removed from a given population (per year), without impacting the size of the population in an unsustainable

manner. This provides a tool to evaluate the cumulative effects of all projected offshore wind farms on the populations involved.

- Method to assess cumulative effects; CUMULEO framework. With the model outcomes, we can progress to evaluate the contribution to the total estimated mortalities by different wind farms, and add up projected wind farm mortality to other man-made mortalities. This is done with the tool "Cumuleo", developed to visualize cumulative effects of multiple sources, and ranking individual contributions of individual sources, in this case particular offshore wind farms.

## 4.2 Basic knowledge and input data

### *4.2.1 Study area*

The study area comprises the southern North Sea, between 51°N and 56°N. Rivers flowing into the North Sea (e.g. the Humber and the Western Scheldt) and marginal seas (e.g. Limfjorden, the Wadden Sea and the Dutch Delta waters) are excluded, as are open, adjoining estuaries, such as the Thames and the Wash mouths (Figure 4.1). The rationale for excluding marginal areas is that there is insufficient coverage from seabird surveys and that these areas are mostly nature conservation areas where no wind farms have been and will be developed.

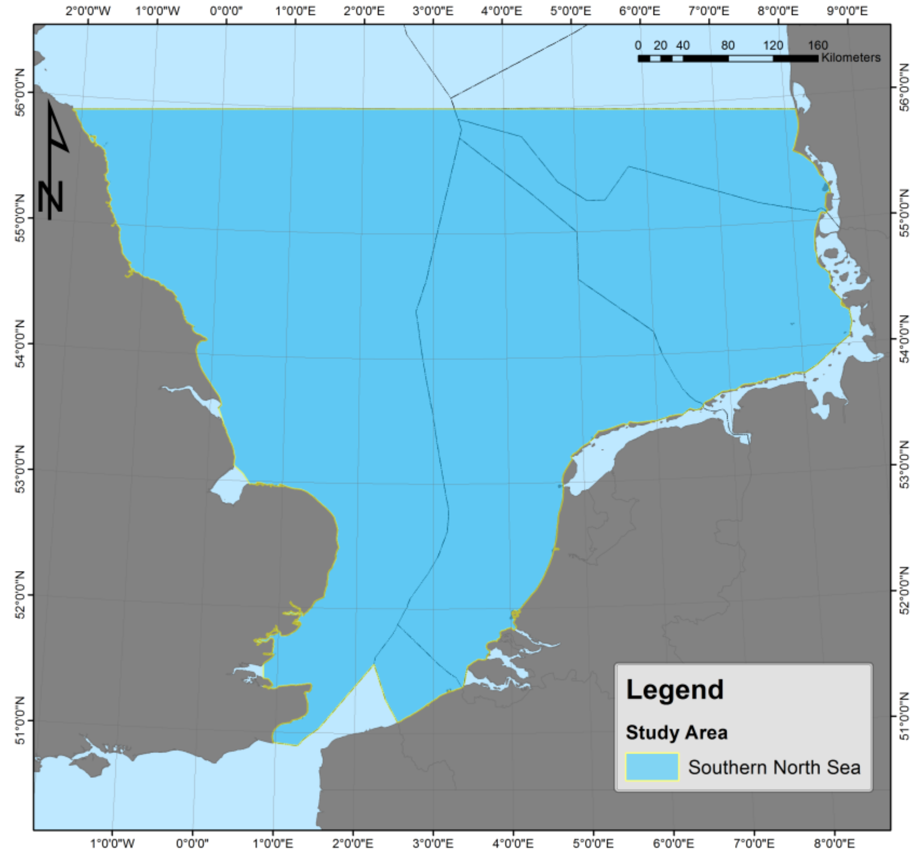


*Figure 4.1. Study area: Southern North Sea, including Belgian, English, Danish, and German waters, between 51 and 56 degrees North, excluding marginal seas and adjoining estuaries.*

Figure 4.2 shows the study area and the two human activities considered in this study: Offshore wind energy and major shipping routes. Existing and proposed MPAs (Marine Protected Areas) have been added for reference: note that these may, or may not overlap with offshore wind farms.

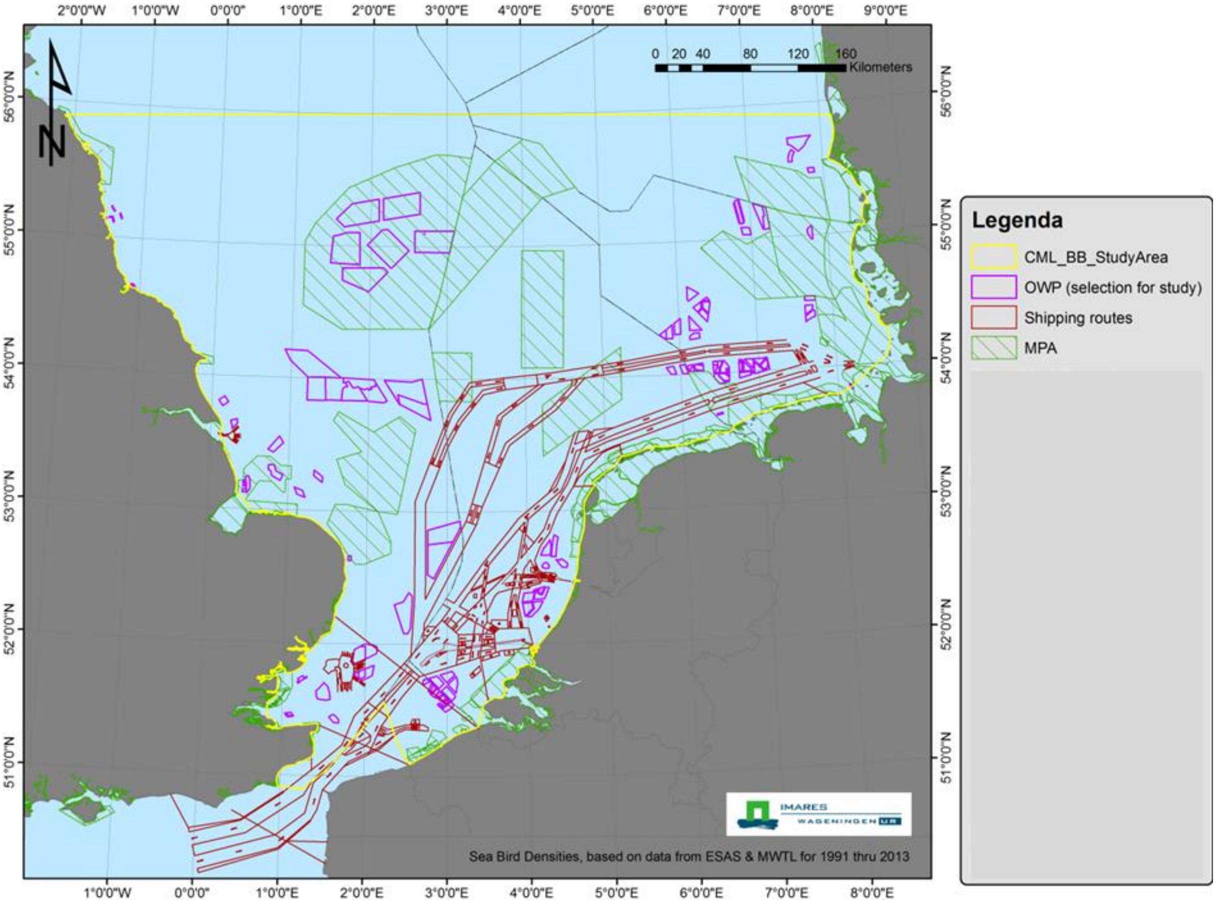


*Figure 4.2. Map of study area, showing the locations of (proposed) offshore wind farms in the study area and the major shipping lanes. Marine Protected Areas are also indicated.*

### *4.2.2 Wind farm data*

The geographical locations of the wind farms considered in this study were already known from a previous study (database Jan Tjalling van der Wal, IMARES). More recent attribute information, as supplied by the Commissioner (the Dutch Ministery of Infrastructure and the Environment, I&M) was checked against the existing information and cross-checked against other sources (e.g. the 4Coffshore-website: http://www.4coffshore.com/offshorewind/). This resulted in some minor edits, e.g. for Alpha Ventus, which has been built in two stages, each with a different type of wind turbine. As a result it was a requirement for this study to divide Alpha Ventus in two parts each with different turbine characteristics.

A series of ten new offshore wind farms has been added to the dataset: SER1 through SER10 (labeled Tender...2015-2019 in Table 4.1). These are intended for development of offshore wind capacity in the Dutch sector of the North Sea in the next decade. According to specifications given by the Ministry, the wind farms are to be located in the designated search areas, plus additional coastward extensions thereof, while being as square as possible and avoiding some known obstacles. These geographical contours were discussed with and approved by the Commissioner for use in this study during a project meeting. At that time the choice for using a 4MW generic turbine for these ten offshore wind farms was also made final.

The above mentioned ten SER-offshore wind farms are replacing most of the previously proposed offshore wind farms in The Netherlands. Also, as per specifications given by the Commissioner, both existing operational offshore wind farms: OWEZ and Prinses Amalia Windpark plus three others have been included. Of these three Eneco Luchterduinen is currently under construction and Gemini East & West have been licensed and are expected to be realised in the next years. All wind farms considered are depicted in Figure 4.3.

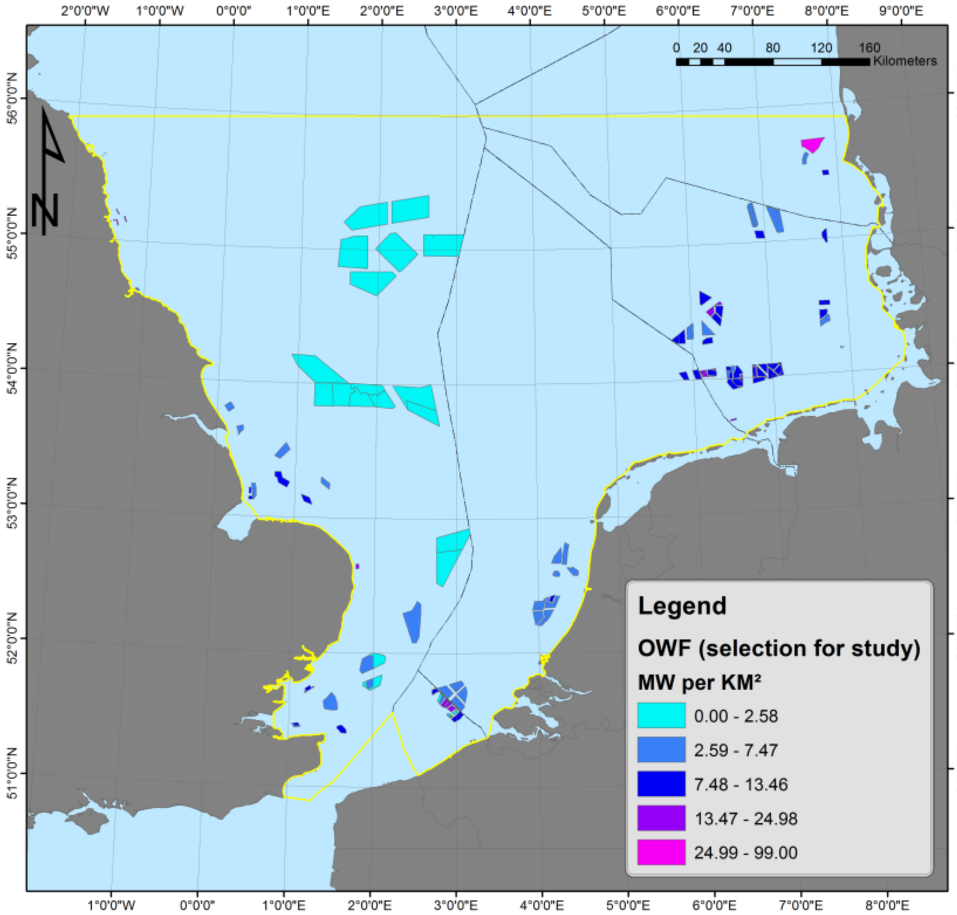


*Figure 4.3 Offshore wind farms as select for this study, shown in different colours, showing their power intensity (MW/km$^2$).*

Table 4.1 gives an overview of the main attributes that have been used for the offshore wind farms included in this study. A limited number of less pertinent (additional) attributes is available, but has been omitted from this table.

*Table 4.1. Offshore Wind Farm attributes. Some additional attributes are not shown due to space limitations on a piece of paper.*

| OFFSHORE WIND FARM NAME | OWP number | Country | Status | Yr StartCon | Yr FullOper | Yr FirstPow | Percent 4Result | Capacity_ 4Calc | TurbineModel | Turb MW | nTurbines | Tip Height M | Hub Height M | Rotor Diam M | Rotor Swept | MW/ km$^2$ | Foundation |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| OVP Albatros | 85 | DE | Licensed | 2018 | 2019 | 2018 | 100 | 553 | 6MW turbine (KEC) | 6 | 79 | 179 | 102 | 154 | 18626.5 | 14.2 | Piled: Jacket |
| Alpha Ventus Nord | 100 | DE | Operational | 2014 | 2014 | 2014 | 100 | 30 | REpower 5M | 5 | 6 | 155 | 92 | 126 | 12469 | 15.3 | Piled: Jacket |
| Alpha Ventus Süd | 10 | DE | Operational | 2014 | 2014 | 2014 | 100 | 30 | M5000-116 (Areva Wind) | 5 | 6 | 148 | 90 | 116 | 10568.3 | 15.3 | Piled: Tripod |
| Amrumbank West | 80 | DE | Construction | 2013 | 2015 | 2015 | 100 | 288 | SWT-3.6-120 (Siemens) | 3.6 | 80 | 150 | 90 | 120 | 11309.7 | 9 | Piled: Monopile |
| BARD Offshore 1 | 22 | DE | Operational | 2014 | 2014 | 2014 | 100 | 400 | Bard 5.0 (Bard) | 5 | 80 | 151 | 90 | 122 | 11689.9 | 6.8 | Piled: Tripile |
| Belwind Alstom Haliade Demonstration | 99 | BE | Construction | 2013 | 2014 | 2014 | 100 | 6 | Haliade 150-6MW (Alstom Power) | 6 | 1 | 175 | 100 | 150 | 17671.5 | 0 | Piled: Jacket |
| Belwind I | 94 | BE | Operational | 2014 | 2014 | 2014 | 100 | 165 | V90-3.0 MW Offshore (Vestas) | 3 | 117 | 117 | 72 | 90 | 6361.7 | 0 | Piled: Monopile |
| Belwind 2 | 95 | BE | Licensed | 2015 | 2017 | 2017 | 100 | 165 | V90-3.0 MW Offshore (Vestas) | 3 | 55 | 117 | 72 | 90 | 6361.7 | 7.5 | Piled: Monopile |
| Blyth | 57 | UK | Operational | 2014 | 2014 | 2014 | 100 | 4 | V66-2MW (Vestas) | 2 | 2 | 95 | 62 | 66 | 3421.2 | 0 | Piled: Monopile |
| Borkum Riffgrund I | 12 | DE | Construction | 2013 | 2015 | 2014 | 100 | 312 | SWT-4.0-120 (Siemens) | 4 | 78 | 145 | 85 | 120 | 11309.7 | 8.7 | Piled: Monopile |
| Borkum Riffgrund II | 88 | DE | Licensed | 2016 | 2018 | 2017 | 100 | 349 | SWT-3.6-120 (Siemens) | 3.6 | 97 | 145 | 85 | 120 | 11309.7 | 7.8 | Piled: Monopile |
| Borkum Riffgrund West I | 34 | DE | Licensed | 2021 | 2023 | 2022 | 100 | 270 | 6MW turbine (KEC) | 6 | 45 | 179 | 102 | 154 | 18626.5 | 9.1 | Piled: Monopile |
| Borkum Riffgrund West II | 83 | DE | Application | 2021 | 2023 | 2022 | 100 | 258 | 6MW turbine (KEC) | 6 | 43 | 179 | 102 | 154 | 18626.5 | 16.4 | Piled: Jacket |
| Trianel Windpark Borkum - phase 1 | 81 | DE | Construction | 2011 | 2014 | 2014 | 100 | 200 | M5000-116 (Areva Wind) | 5 | 40 | 148 | 90 | 116 | 10568.3 | 8.9 | Piled: Tripod |
| Trianel Windpark Borkum - phase 2 | 89 | DE | Licensed | 2015 | 2016 | 2016 | 100 | 200 | M5000-116 (Areva Wind) | 5 | 40 | 148 | 90 | 116 | 10568.3 | 6 | Piled: Tripod |
| Hornsea Project Two - Breesea | 63 | UK | EarlyPlanning | 2018 | 2022 | 2022 | 100 | 900 | 6MW turbine (KEC) | 6 | 150 | 179 | 102 | 154 | 18626.5 | 0 | Piled: Monopile |
| Butendiek | 14 | DE | Construction | 2014 | 2015 | 2014 | 100 | 288 | SWT-3.6-120 (Siemens) | 3.6 | 80 | 150 | 90 | 120 | 11309.7 | 8.7 | Piled: Monopile |
| Dogger Bank Creyke Beck A (Tranche A) | 66 | UK | Application | 2019 | 2022 | 2020 | 100 | 1200 | 6MW turbine (KEC) | 6 | 200 | 179 | 102 | 154 | 18626.5 | 2.3 | Piled: Monopile |
| Dogger Bank Creyke Beck B (Tranche A) | 69 | UK | Application | 2019 | 2022 | 2020 | 100 | 1200 | 6MW turbine (KEC) | 6 | 200 | 179 | 102 | 154 | 18626.5 | 2 | Piled: Monopile |
| DanTysk | 11 | DE | Construction | 2012 | 2015 | 2014 | 100 | 288 | SWT-3.6-120 (Siemens) | 3.6 | 80 | 148 | 88 | 120 | 11309.7 | 4.4 | Piled: Monopile |
| Delta Nordsee 1 | 16 | DE | Licensed | 2019 | 2021 | 2020 | 100 | 210 | 6MW turbine (KEC) | 6 | 35 | 179 | 102 | 154 | 18626.5 | 12.5 | Piled: Monopile |
| Delta Nordsee 2 | 23 | DE | Licensed | 2020 | 2022 | 2021 | 100 | 192 | 6MW turbine (KEC) | 6 | 32 | 179 | 102 | 154 | 18626.5 | 19.8 | Piled: Monopile |
| Deutsche Bucht | 45 | DE | Licensed | 2015 | 2017 | 2017 | 100 | 210 | M5000-116 (Areva Wind) | 5 | 42 | 148 | 90 | 116 | 10568.3 | 9.3 | Piled: Tripod |
| Dudgeon | 40 | UK | Licensed | 2016 | 2017 | 2017 | 100 | 402 | SWT-6.0-154 (Siemens) | 6 | 67 | 179 | 102 | 154 | 18626.5 | 7.3 | Piled: Monopile |

| OFFSHORE WIND FARM NAME | OWP number | Country | Status | Yr StartCon | Yr FullOper | Yr FirstPow | Percent 4Result | Capacity_ 4Calc | TurbineModel | Turb MW | nTurbines | Tip Height M | Hub Height M | Rotor Diam M | Rotor Swept | MW/ km² | Foundation |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| East Anglia Four | 64 | UK | EarlyPlanning | 2021 | 2024 | 2023 | 75 | 1200 | 6MW turbine (KEC) | 6 | 200 | 179 | 102 | 154 | 18626.5 | 0 | Piled: Monopile |
| East Anglia One | 59 | UK | Licensed | 2018 | 2020 | 2019 | 100 | 1200 | 6MW turbine (KEC) | 6 | 200 | 179 | 102 | 154 | 22698 | 4 | Piled: Jacket |
| East Anglia Three | 65 | UK | EarlyPlanning | 2019 | 2022 | 2021 | 100 | 1200 | 8 MW turbine (KEC) | 8 | 150 | 189 | 107 | 164 | 21124.1 | 0 | Piled: Jacket |
| EnBW He Dreiht | 47 | DE | Licensed | 2022 | 2024 | 2023 | 100 | 400 | 5 MW turbine (KEC) | 5 | 80 | 160 | 92.5 | 135 | 14313.9 | 6.4 | Piled: Jacket |
| EnBW Hohe See | 17 | DE | Licensed | 2018 | 2020 | 2019 | 100 | 492 | 6.2M 126 (Senvion) | 6.2 | 80 | 151 | 88 | 126 | 12469 | 11.8 | Piled: Monopile |
| Eneco Luchterduinen | 105 | NL | Construction | 2014 | 2015 | 2015 | 100 | 129 | V112-3.0 MW Offshore (Vestas) | 3 | 43 | 137 | 81 | 112 | 9852 | 8.1 | Piled: Monopile |
| Galloper Wind Farm | 73 | UK | Licensed | 2015 | 2018 | 2017 | 100 | 340 | M5000-135 (Areva Wind) | 5 | 68 | 160 | 92.5 | 135 | 14313.9 | 1.9 | Piled: Monopile |
| Gemini East | 103 | NL | Licensed | 2015 | 2016 | 2016 | 100 | 300 | SWT-4.0-130 (Siemens) | 4 | 75 | 153.5 | 88.5 | 130 | 13273.2 | 8.8 | Piled: Monopile |
| Gemini West | 104 | NL | Licensed | 2015 | 2016 | 2016 | 100 | 300 | SWT-4.0-130 (Siemens) | 4 | 75 | 153.5 | 88.5 | 130 | 13273.2 | 9 | Piled: Monopile |
| Global Tech I | 15 | DE | Construction | 2012 | 2015 | 2015 | 100 | 400 | M5000-116 (Areva Wind) | 5 | 80 | 148 | 90 | 116 | 10568.3 | 9.2 | Piled: Tripod |
| Global Tech II | 48 | DE | Application | 2021 | 2023 | 2022 | 100 | 553 | 6MW turbine (KEC) | 6 | 79 | 179 | 102 | 154 | 18626.5 | 11.3 | Piled: Jacket |
| Gode Wind 01 | 49 | DE | Licensed | 2015 | 2016 | 2016 | 100 | 330 | SWT-6.0-154 (Siemens) | 6 | 55 | 179 | 102 | 154 | 18626.5 | 7.9 | Piled: Monopile |
| Gode Wind 02 | 19 | DE | Licensed | 2015 | 2016 | 2016 | 100 | 252 | SWT-6.0-154 (Siemens) | 6 | 42 | 179 | 102 | 154 | 18626.5 | 8.7 | Piled: Monopile |
| Gode Wind 03 | 50 | DE | Application | 2020 | 2021 | 2020 | 100 | 90 | SWT-6.0-154 (Siemens) | 6 | 15 | 179 | 102 | 154 | 18626.5 | 7.7 | Piled: Monopile |
| Gode Wind 04 | 90 | DE | Licensed | 2020 | 2022 | 2021 | 100 | 252 | SWT-6.0-154 (Siemens) | 6 | 42 | 179 | 102 | 154 | 18626.5 | 8.6 | Piled: Monopile |
| Greater Gabbard | 29 | UK | Operational | 2014 | 2014 | 2014 | 100 | 504 | SWT-3.6-107 (Siemens) | 3.6 | 140 | 131 | 77.5 | 107 | 8992 | 3.4 | Piled: Monopile |
| Gunfleet Sands 3 - Demonstration Project | 58 | UK | Operational | 2014 | 2014 | 2014 | 100 | 12 | SWT-6.0-120 (Siemens) | 6 | 2 | 144 | 84 | 120 | 11309.7 | 4.7 | Piled: Monopile |
| Gunfleet Sands I + II | 28 | UK | Operational | 2014 | 2014 | 2014 | 100 | 172.8 | SWT-3.6-107 (Siemens) | 3.6 | 48 | 128.5 | 75 | 107 | 8992 | 10.9 | Piled: Monopile |
| Hornsea Project One - Heron Wind | 60 | UK | Application | 2017 | 2020 | 2018 | 100 | 498 | SWT-6.0-154 (Siemens) | 6 | 83 | 200 | 123 | 154 | 18626.5 | 2.3 | Piled: Monopile |
| Horns Rev 1 | 32 | DK | Operational | 2014 | 2014 | 2014 | 100 | 160 | V80-2.0 MW (Vestas) | 2 | 80 | 110 | 70 | 80 | 5026.5 | 7.7 | Piled: Monopile |
| Horns Rev 2 | 33 | DK | Operational | 2014 | 2014 | 2014 | 100 | 209.3 | SWT-2.3-93 (Siemens) | 2.3 | 91 | 114.5 | 68 | 93 | 6792.9 | 6.3 | Piled: Monopile |
| Horns Rev 3 | 93 | DK | EarlyPlanning | 2016 | 2020 | 2017 | 100 | 396 | 6 MW (KEC) | 6 | 66 | 179 | 102 | 154 | 18626.5 | 99 | Piled: Monopile |
| Hornsea Project Two - Optimus Wind | 62 | UK | EarlyPlanning | 2018 | 2022 | 2022 | 100 | 900 | 6MW turbine (KEC) | 6 | 150 | 179 | 102 | 154 | 18626.5 | 0 | Piled: Monopile |
| Hornsea Project Three SPC 5 | 74 | UK | EarlyPlanning | 2021 | 2025 | 2025 | 50 | 600 | 6MW turbine (KEC) | 6 | 100 | 179 | 102 | 154 | 18626.5 | 0 | Piled: Monopile |

| OFFSHORE WIND FARM NAME | OWP number | Country | Status | Yr StartCon | Yr FullOper | Yr FirstPow | Percent 4Result | Capacity_ 4Calc | TurbineModel | Turb MW | nTurbines | Tip Height M | Hub Height M | Rotor Diam M | Rotor Swept | MW/ km² | Foundation |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Hornsea Project Three SPC 6 | 75 | UK | EarlyPlanning | 2021 | 2025 | 2025 | 50 | 600 | 6MW turbine (KEC) | 6 | 100 | 179 | 102 | 154 | 18626.5 | 0 | Piled: Monopile |
| Hornsea Project Three SPC 7 | 76 | UK | EarlyPlanning | 2021 | 2025 | 2025 | 0 | 0 | 6MW turbine (KEC) | 6 | 0 | 179 | 102 | 154 | 18626.5 | 0 | Piled: Monopile |
| Hornsea Project Three SPC 8 | 77 | UK | EarlyPlanning | 2021 | 2025 | 2025 | 0 | 0 | 6MW turbine (KEC) | 6 | 0 | 179 | 102 | 154 | 18626.5 | 0 | Piled: Monopile |
| Humber Gateway | 72 | UK | Construction | 2013 | 2015 | 2014 | 100 | 219 | V112-3.0 MW Offshore (Vestas) | 3 | 73 | 136 | 80 | 112 | 9852 | 6.3 | Piled: Monopile |
| Inner Dowsing | 27 | UK | Operational | 2014 | 2014 | 2014 | 100 | 97.2 | SWT-3.6-107 (Siemens) | 3.6 | 27 | 133.5 | 80 | 107 | 8992 | 9.7 | Piled: Monopile |
| Nordsee One | 79 | DE | Licensed | 2015 | 2016 | 2016 | 100 | 332.1 | 6.2M 126 (Senvion) | 6.2 | 54 | 163 | 100 | 126 | 12469 | 10 | Piled: Monopile |
| Innogy Nordsee 2 | 86 | DE | Licensed | 2019 | 2021 | 2020 | 100 | 295.2 | 6.2M 126 (Senvion) | 6.2 | 48 | 163 | 100 | 126 | 12469 | 8.1 | Piled: Monopile |
| Innogy Nordsee 3 | 87 | DE | Licensed | 2020 | 2022 | 2021 | 100 | 369 | 6.2M 126 (Senvion) | 6.2 | 60 | 163 | 100 | 126 | 12469 | 12.7 | Piled: Monopile |
| Kaikas | 44 | DE | Licensed | 2017 | 2019 | 2018 | 100 | 581 | 6MW turbine (KEC) | 6 | 83 | 179 | 102 | 154 | 18626.5 | 9.3 | Piled: Jacket |
| Kentish Flats | 25 | UK | Operational | 2014 | 2014 | 2014 | 100 | 90 | V90-3.0 MW Offshore (Vestas) | 3 | 30 | 115 | 70 | 90 | 6361.7 | 9.1 | Piled: Monopile |
| Kentish Flats Extension | 53 | UK | Licensed | 2015 | 2015 | 2015 | 100 | 49.5 | V112-3.3 MW Offshore (MHI Vestas) | 3.3 | 15 | 139.6 | 83.6 | 112 | 9852 | 6 | Piled: Monopile |
| Lincs | 41 | UK | Operational | 2014 | 2014 | 2014 | 100 | 270 | SWT-3.6-120 (Siemens) | 3.6 | 75 | 160 | 100 | 120 | 11309.7 | 6.6 | Piled: Monopile |
| London Array Phase 1 | 54 | UK | Operational | 2014 | 2014 | 2014 | 100 | 630 | SWT-3.6-120 (Siemens) | 3.6 | 175 | 147 | 87 | 120 | 11309.7 | 6.3 | Piled: Monopile |
| Lynn | 26 | UK | Operational | 2014 | 2014 | 2014 | 100 | 97.2 | SWT-3.6-107 (Siemens) | 3.6 | 27 | 133.5 | 80 | 107 | 8992 | 9.7 | Piled: Monopile |
| Meerwind Ost/Sud | 51 | DE | Construction | 2012 | 2014 | 2014 | 100 | 288 | SWT-3.6-120 (Siemens) | 3.6 | 80 | 145 | 85 | 120 | 11309.7 | 7.3 | Piled: Monopile |
| MEG Offshore I | 82 | DE | Licensed | 2016 | 2017 | 2017 | 100 | 400 | M5000-116 (Areva Wind) | 5 | 80 | 148 | 90 | 116 | 10568.3 | 8.5 | Piled: Tripod |
| Blyth Offshore Wind Demonstration site | 57 | UK | Licensed | 2016 | 2018 | 2016 | 100 | 90 | 6MW turbine (KEC) | 6 | 15 | 179 | 102 | 154 | 18626.5 | 25 | Piled: Jacket |
| Hornsea Project One - Njord | 61 | UK | Application | 2017 | 2018 | 2018 | 100 | 498 | SWT-6.0-154 (Siemens) | 6 | 83 | 200 | 123 | 154 | 18626.5 | 2.6 | Piled: Monopile |
| Nordergründe | 20 | DE | Licensed | 2016 | 2017 | 2016 | 100 | 110.7 | 6.2M 126 (Senvion) | 6.2 | 18 | 147 | 84 | 126 | 12469 | 18.9 | Piled: Monopile |
| Nördlicher Grund | 91 | DE | Licensed | 2023 | 2025 | 2024 | 25 | 384 | 6MW turbine (KEC) | 6 | 64 | 179 | 102 | 154 | 12271.8 | 9.1 | Piled: Monopile |
| Nordpassage | 43 | DE | Application | 2023 | 2025 | 2024 | 25 | 400 | 5 MW turbine (KEC) | 5 | 80 | 160 | 92.5 | 135 | 14313.9 | 4.5 | Piled: Monopile |
| Nordsee Ost | 13 | DE | Construction | 2012 | 2015 | 2015 | 100 | 295.2 | 6.2M 126 (Senvion) | 6.2 | 48 | 153 | 90 | 126 | 12469 | 8.3 | Piled: Jacket |
| Norther | 38 | BE | Licensed | 2016 | 0 | 2017 | 100 | 468 | 6MW turbine (KEC) | 6 | 78 | 179 | 102 | 154 | 18626.5 | 12.3 | Piled: Monopile |
| Northwind | 35 | BE | Construction | 2013 | 2014 | 2014 | 100 | 216 | V112-3.0 MW Offshore (Vestas) | 3 | 72 | 127 | 71 | 112 | 9852 | 15.1 | Piled: Monopile |
| OWEZ, Offshore Windpark Egmond aan Zee | 102 | NL | Operational | 2014 | 2014 | 2014 | 100 | 108 | V90-3.0 MW Offshore (Vestas) | 3 | 36 | 115 | 70 | 90 | 6361.7 | 4.4 | Piled: Monopile |

| OFFSHORE WIND FARM NAME | OWP number | Country | Status | Yr StartCon | Yr FullOper | Yr FirstPow | Percent 4Result | Capacity_ 4Calc | TurbineModel | Turb MW | nTurbines | Tip Height M | Hub Height M | Rotor Diam M | Rotor Swept | MW/ km² | Foundation |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| OWP West | 84 | DE | Licensed | 2021 | 2022 | 2021 | 100 | 328 | 6MW turbine (KEC) | 6 | 41 | 179 | 102 | 154 | 18626.5 | 23 | Piled: Monopile |
| Prinses Amalia Windpark | 101 | NL | Operational | 2014 | 2014 | 2014 | 100 | 120 | V80-2.0 MW (Vestas) | 2 | 60 | 99 | 59 | 80 | 5026.5 | 7.2 | Piled: Monopile |
| Race Bank | 42 | UK | Licensed | 2016 | 2018 | 2018 | 100 | 546 | 6MW turbine (KEC) | 6 | 91 | 179 | 102 | 154 | 18869.2 | 9.4 | Piled: Monopile |
| RENTEL | 36 | BE | Licensed | 2016 | 2018 | 2017 | 100 | 468 | 6MW turbine (KEC) | 6 | 78 | 179 | 102 | 154 | 18626.5 | 20.2 | Piled: Monopile |
| Riffgat | 21 | DE | Operational | 2014 | 2014 | 2014 | 100 | 108 | SWT-3.6-120 (Siemens) | 3.6 | 30 | 150 | 90 | 120 | 11309.7 | 18.1 | Piled: Monopile |
| Sandbank | 18 | DE | Licensed | 2015 | 2016 | 2016 | 100 | 288 | SWT-4.0-130 (Siemens) | 4 | 72 | 145 | 80 | 130 | 13273.2 | 4.4 | Piled: Monopile |
| Sandbank extension | 52 | DE | Application | 2023 | 2025 | 2024 | 25 | 240 | 6MW turbine (KEC) | 6 | 40 | 179 | 102 | 154 | 18626.5 | 6.4 | Piled: Monopile |
| Scroby Sands | 24 | UK | Operational | 2014 | 2014 | 2014 | 100 | 60 | V80-2.0 MW (Vestas) | 2 | 30 | 100 | 60 | 80 | 5026.5 | 14.2 | Piled: Monopile |
| Seastar | 97 | BE | Licensed | 2017 | 2018 | 2018 | 100 | 372 | 6MW turbine (KEC) | 6 | 62 | 179 | 102 | 154 | 18626.5 | 18.4 | Piled: Monopile |
| Tender 2015 (1)- Borssele (SER1) | 0 | NL | EarlyPlanning | 2017 | 2019 | 2019 | 100 | 350 | 4MW basic | 4 | 88 | 155 | 90 | 130 | 13273.2 | 6 | Piled: Monopile |
| Tender 2019 (10) - Hollandse Kust Noord Holland | 7 | NL | EarlyPlanning | 2021 | 2023 | 2023 | 100 | 350 | 4MW basic | 4 | 88 | 155 | 90 | 130 | 13273.2 | 6 | Piled: Monopile |
| Tender 2015 (2)- Borssele (SER2) | 9 | NL | EarlyPlanning | 2017 | 2019 | 2019 | 100 | 350 | 4MW basic | 4 | 88 | 155 | 90 | 130 | 13273.2 | 6 | Piled: Monopile |
| Tender 2016 (3) - Borssele (SER3) | 1 | NL | EarlyPlanning | 2018 | 2020 | 2020 | 100 | 350 | 4MW basic | 4 | 88 | 155 | 90 | 130 | 13273.2 | 6 | Piled: Monopile |
| Tender 2016 (4) - Borssele (SER4) | 8 | NL | EarlyPlanning | 2018 | 2020 | 2020 | 100 | 350 | 4MW basic | 4 | 88 | 155 | 90 | 130 | 13273.2 | 6 | Piled: Monopile |
| Tender 2017 (5) - Hollandse Kust Zuid Holland | 2 | NL | EarlyPlanning | 2019 | 2021 | 2021 | 100 | 350 | 4MW basic | 4 | 88 | 155 | 90 | 130 | 13273.2 | 6 | Piled: Monopile |
| Tender 2017 (6) - HK Zuid Holland (SER6) | 3 | NL | EarlyPlanning | 2019 | 2021 | 2021 | 100 | 350 | 4MW basic | 4 | 88 | 155 | 90 | 130 | 13273.2 | 6 | Piled: Monopile |
| Tender 2018 (7)- HK Zuid Holland (SER7) | 4 | NL | EarlyPlanning | 2020 | 2022 | 2022 | 100 | 350 | 4MW basic | 4 | 88 | 155 | 90 | 130 | 13273.2 | 6 | Piled: Monopile |
| Tender 2018 (8)- HK Zuid Holland (SER8) | 5 | NL | EarlyPlanning | 2020 | 2022 | 2022 | 100 | 350 | 4MW basic | 4 | 88 | 155 | 90 | 130 | 13273.2 | 6 | Piled: Monopile |
| Tender 2019 (9) - HK Noord Holland (SER 9) | 6 | NL | EarlyPlanning | 2021 | 2023 | 2023 | 100 | 350 | 4MW basic | 4 | 88 | 155 | 90 | 130 | 13273.2 | 6 | Piled: Monopile |
| Sheringham Shoal | 31 | UK | Operational | 2014 | 2014 | 2014 | 100 | 316.8 | SWT-3.6-107 (Siemens) | 3.6 | 88 | 135.3 | 81.8 | 107 | 8992 | 9.1 | Piled: Monopile |
| Teesside | 56 | UK | Operational | 2014 | 2014 | 2014 | 100 | 62.1 | SWT-2.3-93 (Siemens) | 2.3 | 27 | 126.5 | 80 | 93 | 6792.9 | 6.3 | Piled: Monopile |
| Dogger Bank Teesside A (Tranche B) | 67 | UK | Application | 2020 | 2023 | 2021 | 100 | 1200 | 6MW turbine (KEC) | 6 | 200 | 179 | 102 | 154 | 18626.5 | 2.1 | Piled: Monopile |
| Dogger Bank Teesside B (Tranche B) | 68 | UK | Application | 2020 | 2023 | 2020 | 100 | 1200 | 6MW turbine (KEC) | 6 | 200 | 179 | 102 | 154 | 18626.5 | 2 | Piled: Monopile |
| Dogger Bank Teesside C (Tranche C) | 70 | UK | EarlyPlanning | 2021 | 2024 | 2022 | 75 | 1200 | 6MW turbine (KEC) | 6 | 200 | 179 | 102 | 154 | 18626.5 | 0 | Piled: Monopile |

| OFFSHORE WIND FARM NAME | OWP number | Country | Status | Yr StartCon | Yr FullOper | Yr FirstPow | Percent 4Result | Capacity_ 4Calc | TurbineModel | Turb MW | nTurbines | Tip Height M | Hub Height M | Rotor Diam M | Rotor Swept | MW/ km² | Foundation |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Dogger Bank Teesside D (Tranche C) | 71 | UK | EarlyPlanning | 2021 | 2024 | 2022 | 75 | 1200 | 6MW turbine (KEC) | 6 | 200 | 179 | 102 | 154 | 18626.5 | 0 | Piled: Monopile |
| Thanet | 30 | UK | Operational | 2014 | 2014 | 2014 | 100 | 300 | V90-3.0 MW Offshore (Vestas) | 3 | 100 | 115 | 70 | 90 | 6361.7 | 8.6 | Piled: Monopile |
| Thornton Bank phase I | 98 | BE | Operational | 2014 | 2014 | 2014 | 100 | 30 | 5M (Senvion) | 5.1 | 157 | 157 | 94 | 126 | 12469 | 0 | Gravitybased |
| Thornton Bank phase II | 37 | BE | Operational | 2014 | 2014 | 2014 | 100 | 184.5 | 6.2M 126 (Senvion) | 6.2 | 158 | 158 | 95 | 126 | 12469 | 0 | Piled: Jacket |
| Thornton Bank phase III | 96 | BE | Operational | 2014 | 2014 | 2014 | 100 | 110.7 | 6.2M 126 (Senvion) | 6.2 | 158 | 158 | 95 | 126 | 12469 | 0 | Piled: Jacket |
| THV Mermaid | 92 | BE | EarlyPlanning | 2017 | 2018 | 2018 | 100 | 234 | 6MW turbine (KEC) | 6 | 39 | 179 | 102 | 154 | 18626.5 | 13.5 | Piled: Monopile |
| Triton Knoll | 78 | UK | Licensed | 2019 | 2021 | 2019 | 100 | 900 | 6MW turbine (KEC) | 6 | 150 | 179 | 102 | 154 | 25446.9 | 6.7 | Piled: Monopile |
| Veja Mate | 46 | DE | Licensed | 2016 | 2017 | 2017 | 100 | 396 | 6MW turbine (KEC) | 6 | 66 | 179 | 102 | 154 | 18626.5 | 8 | Piled: Jacket |
| Westermost Rough | 39 | UK | Construction | 2013 | 2015 | 2014 | 100 | 210 | SWT-6.0-154 (Siemens) | 6 | 35 | 179 | 102 | 154 | 18626.5 | 6 | Piled: Monopile |

### *4.2.3 Seabird survey data*

For an assessment of the possible impact of offshore wind farms on biota, in this case seabirds, migrant birds and bats, the basic data required are the densities of the target organisms at the wind farm sites. In case of the seabirds, at sea survey data, both ship-based and aerial surveys, were used. Their on-site densities were then used to estimate wind farm mortality, both from collisions and from habitat loss (displacement), using methods developed by Bradbury *et al.* (2014), which were further developed in this report (see section 4.3.1). Migrant birds only face collision mortality. Their on-site numbers have been estimated using a combination of data on population sizes, estimated of proportion of these crossing the North Sea and the altitudes at which they do so. Note that at-sea collision studies have not yet been conducted and that all modelling relies mostly on studies done on land. For bats, we lack the most basic information on numbers at sea and we can only extrapolate mortility from land studies, for species known to, or considered to, fly over the North Sea.

To assess the impact of offshore wind park development on sea birds (updated) maps of their density and distribution across the study area were needed. The methodology followed is based on maps produced earlier, for the WindSpeed-project (Van der Wal *et al.* 2011b).

#### 4.2.3.1 Original data from two sources: ESAS and MWTL

The original data on seabird distributions and densities that were available for this study came from two sources, the ESAS-database and the MWTL North Sea Monitoring program.

The latest available ESAS-database (http://jncc.defra.gov.uk/page-4469 ) used was v5 ESAS, to which several Dutch ship-based surveying trips were added that were not yet included. After additional checks on the quality and integrity of the data in this updated version (v6) the data was released for use in producing the density maps of seabird distribution. ESAS contains both ship-based and aerial surveys, that were treated separately. To ease further processing, the MWTL aerial survey data, as supplied by the Commissioner of this project, were also kept separate. Therefore, three separate databases were created:

1) ESAS ship
2) ESAS fly
3) MWTL fly

From each database only observations were used from year>1990, i.e. data from 1991 to the most recent data available (March 2014).

#### 4.2.3.2 Combining the three sources into a single geographical dataset

For all three databases each observation came with a geographical location. For the combined result a 5 x 5 km vector grid has been prepared in a GIS (ESRI ArcGIS 10.2), and each rectangle was uniquely identifiable based on a GRID_ID. This GRID_ID was constructed from the geographical centrepoints of the grid cells by taking the first kilometer values of both the Easting and Northing values (WGS84 UTM31). Each observation in each of the three databases was plotted in GIS and joined on location to the vector grid, after which each record was enriched with a new field indicating the GRID_ID into which it falls. From this step onward most of the data were processed with regular database tools and the use of ArcGIS was no longer required. The geographical datasets were located in one single personal geodatabase (pGDB is an ESRI specific implementation of an MS-Access database). For manipulating and processing the data, a parallel Access-database was established that links to the tables in the pGDB. This is a work-around, as ArcGIS does not allow for queries, forms etc to exists inside the pGDB.
Season was added as a field to each of the three databases (Integer, [1..6]) where a season denotes a two-month period with Season=1 being August + September; 2 October + November etc. The existing field Month was used as the input for assigning the correct season to each record.

#### 4.2.3.3 Splitting of ‘combi-species’ for MWTL and ESAS fly

From an airplane with a survey altitude of 500 ft as conducted in the MWTL monitoring programme a number of sea bird species is indistinguishable from each other, e.g. ‘razormots’, i.e, Razorbills/Common Guillemots (=Common Murres). For these, the ship-based observations from the same season and area have been used to split these ‘combi-species’ into the respective single species. The same approach was used to separate juvenile Lesser Black-backed Gulls (Euring 5910) from juvenile European Herring gulls (Euring 5920) in the MWTL fly-database. Similar combi records for unidentified gulls within the ESAS ship-based data were split. The ESAS fly database did not have any combined observations (inside the study area) and therefore no splitting of species was needed here.

#### 4.2.3.4 Lumping of species

For a few species a choice has been made to lump them together in the density maps. This was done for Arctic and Common Terns and for small loons that were often too difficult to distinguish from each other reliably (even ship-based). “Commic Tern” (Euring 6159) is a combination of Arctic Tern (Euring 6160) and Common Tern (Euring 6150). “Loon spec.” (Euring 59) is a combination of Red-throated Loon (Euring 20) and Black-throated Loon (Euring 30).

#### 4.2.3.5 Density sampling applies to ESAS ship

The ship-based observations in ESAS are made using a strip-count with series of strips on one or both sides of the ship. Based on density sampling theory and on the assumption that the birds were evenly distributed before the observing ship entered the area, and that equal densities should be present at all distances from the ship’s track line, species-specific correction factors were derived to compensate for birds missed at greater perpendicular distances (Table 4.2).

*Table 4.2. Density sampling correction factors for bird counts (ESAS ship).*

| ESAScode | NameEN | Correction factor |
|---|---|---|
| 220 | Northern Fulmar | 1.220 |
| 5780 | Little Gull | 1.220 |
| 5820 | Black-headed Gull | 1.097 |
| 5900 | Mew Gull | 1.170 |
| 5910 | Lesser Black-backed Gull | 1.153 |
| 5920 | European Herring Gull | 1.086 |
| 6000 | Great Black-backed Gull | 1.172 |
| 6020 | Black-legged Kittiwake | 1.204 |
| 430 | Sooty Shearwater | 1 000 |
| 460 | Manx Shearwater | 1.255 |
| 520 | European Storm-petrel | 1.407 |
| 550 | Leach's Storm-petrel | 1.392 |
| 710 | Northern Gannet | 1 000 |
| 5670 | Parasitic Jaeger | 1.224 |
| 5690 | Great Skua | 1.216 |
| 6110 | Sandwich Tern | 1.106 |
| 6150 | Common Tern | 1.099 |
| 6160 | Arctic Tern | 1.099 |
| 6169 | "Commic Tern" | 1.099 |

| ESAScode | NameEN | Correction factor |
|---|---|---|
| 6240 | Little Tern | 1.280 |
| 6340 | Common Murre | 1.422 |
| 6360 | Razorbill | 1.441 |
| 6470 | Little Auk | 1.375 |
| 6540 | Atlantic Puffin | 1.073 |
| 2060 | Common Eider | 0.998 |
| 2130 | Common Scoter | 1.017 |
| 2150 | Velvet Scoter | 1.029 |
| 20 | Red-throated Loon | 1.160 |
| 30 | Black-throated Loon | 1.160 |
| 59 | Loon spec. | 1.160 |
| 90 | Great Crested Grebe | 1 000 |
| 720 | Great Cormorant | 1.083 |
| 800 | European Shag | 1.231 |

#### 4.2.3.6 Extracting survey effort and bird counts

The next step was to extract survey effort from each of the three databases. Effort may differ between species. This is because not all bird species were included in all counts (Table 4.3). All valid observations were combined with the appropriate survey effort, yielding a bird density (birds/km²) per species, per count. From the ESAS database the correct effort and its geographical lay-out were determined, using seven different trip profiles (Table 4.4).

During ESAS seabirds counts, both birds inside the counting strip, and in case of flying birds, individuals present over the counting strip at appropriate snap shot moments and distances: see Tasker *et al*. (1984) for a full description of the methods used) are noted, and birds at greater distances or outside snapshot moments are omitted. Only the first category of birds is used in density estimates. These are coded “2” in the column {Transect indicator} in ESAS table ‘Species’; all other birds are coded “1”. The latter birds were not used in density estimates or for estimating numbers of wind farm victims, but have been included in dot maps for rare species in this report.

*Table 4.3. ESAS codes and their interpretation for including different bird species.*

| Code | Meaning |
|---|---|
| 1 | All species recorded |
| 2 | All species except *Larus* Gulls |
| 3 | All species except Fulmars |
| 4 | All species except *Larus* Gulls, Fulmars and Kittiwakes |
| 5 | Auks only |
| 6 | Auks and Seaduck only |
| 7 | All species except Eiders and Gulls |
| 8 | All species except Gannets |
| 9 | Other |
| 10 | Cetaceans only |

*Table 4.4. ESAS codes and their interpretation for different counting modes.*

| | |
|---|---|
| 1 | Full transect method with snapshot for flying birds |
| 2 | On water transect, no snapshot for flying birds |
| 3 | All observations, but no transect operated |
| 4 | Presence / absence data |
| 5 | Full transect, but no scan data for outside the transect |

# from the European Seabirds at Sea Database Coding Manual (version) a.o. available from http://jncc.defra.gov.uk/pdf/ESAS%20Coding%20Manual%20v%204.pdf

The combination of {Counting methods} and {Species observed} (both fields from ‘Trip’) defines the TRIPselProfile as shown in Table 4.5.

*Table 4.5. ESAS (ship) trip selection profile as valid for the various species of sea bird included in this study.*

| ESAS code | Counting methods | Species observed | TRIPsel-Profile | Transect = | Name EN | Scientific Name |
|---|---|---|---|---|---|---|
| 20 | With 30, as 59 | 1-2-3-4-6-7 | #N/A | 2 | Red-throated Loon | *Gavia stellata* |
| 30 | With 20, as 59 | 1-2-3-4-6-7 | #N/A | 2 | Black-throated Loon | *Gavia arctica* |
| 40 | Dot map | All incl. Transect=1 | 7 | 1,2 | Great Northern Loon | *Gavia immer* |
| 50 | Dot map | All incl. Transect=1 | 7 | 1,2 | White-billed Loon | *Gavia adamsii* |
| 59 | 1-2-3-5-7 | 1-2-3-4-6-7 | 6 | 2 | Loon spec. | *Gavia spec.* |
| 90 | 1-2-3-5-7 | 1-2-3-4-6-7 | 6 | 2 | Great Crested Grebe | *Podiceps cristatus* |
| 220 | 1-5-7 | 1-2-7 | 1 | 2 | Northern Fulmar | *Fulmarus glacialis* |
| 430 | 1-5-7 | 1-2-3-4-7 | 3 | 2 | Sooty Shearwater | *Puffinus griseus* |
| 460 | 1-5-7 | 1-2-3-4-7 | 3 | 2 | Manx Shearwater | *Puffinus puffinus* |
| 462 | Dot map | All incl. Transect=1 | 7 | 1,2 | Balearic Shearwater | *Puffinus mauretanicus* |
| 520 | 1-5-7 | 1-2-3-4-7 | 3 | 2 | European Storm-petrel | *Hydrobates pelagicus* |
| 550 | 1-5-7 | 1-2-3-4-7 | 3 | 2 | Leach's Storm-petrel | *Oceanodroma leucorhoa* |
| 710 | 1-5-7 | 1-2-3-4-7 | 3 | 2 | Northern Gannet | *Morus bassanus* |

| ESAS code | Counting methods | Species observed | TRIPsel-Profile | Tran-sect = | Name EN | Scientific Name |
|---|---|---|---|---|---|---|
| 720 | 1-2-3-5-7 | 1-2-3-4-6-7 | 6 | 2 | Great Cormorant | *Phalacrocorax carbo* |
| 800 | 1-2-3-5-7 | 1-2-3-4-6-7 | 6 | 2 | European Shag | *Phalacrocorax aristotelis* |
| 2060 | 1-2-3-5-7 | 1-2-3-4-6 | 5 | 2 | Common Eider | *Somateria mollissima* |
| 2130 | 1-2-3-5-7 | 1-2-3-4-6 | 5 | 2 | Common Scoter | *Melanitta nigra* |
| 2150 | 1-2-3-5-7 | 1-2-3-4-6 | 5 | 2 | Velvet Scoter | *Melanitta fusca* |
| 5670 | 1-5-7 | 1-2-3-4-7 | 3 | 2 | Parasitic Jaeger | *Stercorarius parasiticus* |
| 5690 | 1-5-7 | 1-2-3-4-7 | 3 | 2 | Great Skua | *Stercorarius skua* |
| 5780 | 1-5-7 | 1-3 | 2 | 2 | Little Gull | *Larus minutus* |
| 5820 | 1-5-7 | 1-3 | 2 | 2 | Black-headed Gull | *Larus ridibundus* |
| 5900 | 1-5-7 | 1-3 | 2 | 2 | Mew Gull | *Larus canus* |
| 5910 | 1-5-7 | 1-3 | 2 | 2 | Lesser Black-backed Gull | *Larus fuscus* |
| 5920 | 1-5-7 | 1-3 | 2 | 2 | European Herring Gull | *Larus argentatus* |
| 6000 | 1-5-7 | 1-3 | 2 | 2 | Great Black-backed Gull | *Larus marinus* |
| 6020 | 1-5-7 | 1-3 | 2 | 2 | Black-legged Kittiwake | *Rissa tridactyla* |
| 6110 | 1-5-7 | 1-2-3-4-7 | 3 | 2 | Sandwich Tern | *Sterna sandvicensis* |
| 6140 | Dot map | All incl. Transect=1 | 7 | 1,2 | Roseate Tern | *Sterna dougallii* |
| 6270 | Dot map | All incl. Transect=1 | 7 | 1,2 | Black Tern | *Chlidonias niger* |
| 6150 | With 6160, as 6169 | 1-2-3-4-7 | #N/A | 2 | Common Tern | *Sterna hirundo* |
| 6160 | With 6150, as 6169 | 1-2-3-4-7 | #N/A | 2 | Arctic Tern | *Sterna paradisaea* |
| 6169 | 1-5-7 | 1-2-3-4-7 | 3 | 2 | "Commic Tern" | *S. hirundo/paradisaea* |
| 6240 | 1-5-7 | 1-2-3-4-7 | 3 | 2 | Little Tern | *Sterna albifrons* |
| 6340 | 1-2-3-5-7 | 1-2-3-4-5-6-7 | 4 | 2 | Common Murre | *Uria aalge* |
| 6360 | 1-2-3-5-7 | 1-2-3-4-5-6-7 | 4 | 2 | Razorbill | *Alca torda* |
| 6380 | Dot map | All incl. Transect=1 | 7 | 1,2 | Black Guillemot | *Cepphus grylle* |
| 6470 | 1-2-3-5-7 | 1-2-3-4-5-6-7 | 4 | 2 | Little Auk | *Alle alle* |
| 6540 | 1-2-3-5-7 | 1-2-3-4-5-6-7 | 4 | 2 | Atlantic Puffin | *Fratercula arctica* |

For aerial survey data no trip profiles were required, as in all cases the objective was to count all species of birds that are observed (MWTL and ESAS fly). With the trip profiles and the mapping and counting of the effort, while disregarding whether or not birds have been counted, a number of complications are circumvented. On some stretches of a survey not a single bird may be observed, this is recorded with Euring=0 and is a valid observation for any bird species: no birds seen in the area covered. Also any bird species that is observed (but no others), also constitutes a valid effort for all other bird species. Again the main thing is that it yields valid effort. Determining valid effort gives us the geographical base onto which the observations of a bird can be added. For ESAS ship, the effort map and the count need to respect the trip profiles. After the correct amounts of effort and the correct counts of each bird species were determined for each of the three separate databases, these were combined into a single unified table. Both the effort and counts of birds in transect were summed for each bird species, season and grid_id (location). From this we calculated the density by dividing ∑count by ∑effort (bird/km$^2$).

#### 4.2.3.7 Effort maps

The available survey effort greatly determines what is known about bird distributions over the area, per season. Effort maps (Figure 4.4), in which the joint effort of all ship-based surveys and aerial surveys is combined, are given in this section (disregarding species-specific differences as explained above). Note that coverage has been unequal, both between seasons and between various regions within the southern North Sea. In August/September, the whole area has been covered best (but also note that effort of all survey years has been combined). In the other seasons, the eastern parts of the southern North Sea: the Belgian and Dutch Continental shelves and the inner German Bight tend to have been covered (much) better than UK waters, particularly the waters off Norfolk. These UK waters have been covered mainly by aerial surveys (Bradbury *et al.* 2014), but these data were not available within ESAS and have not been used in this study.

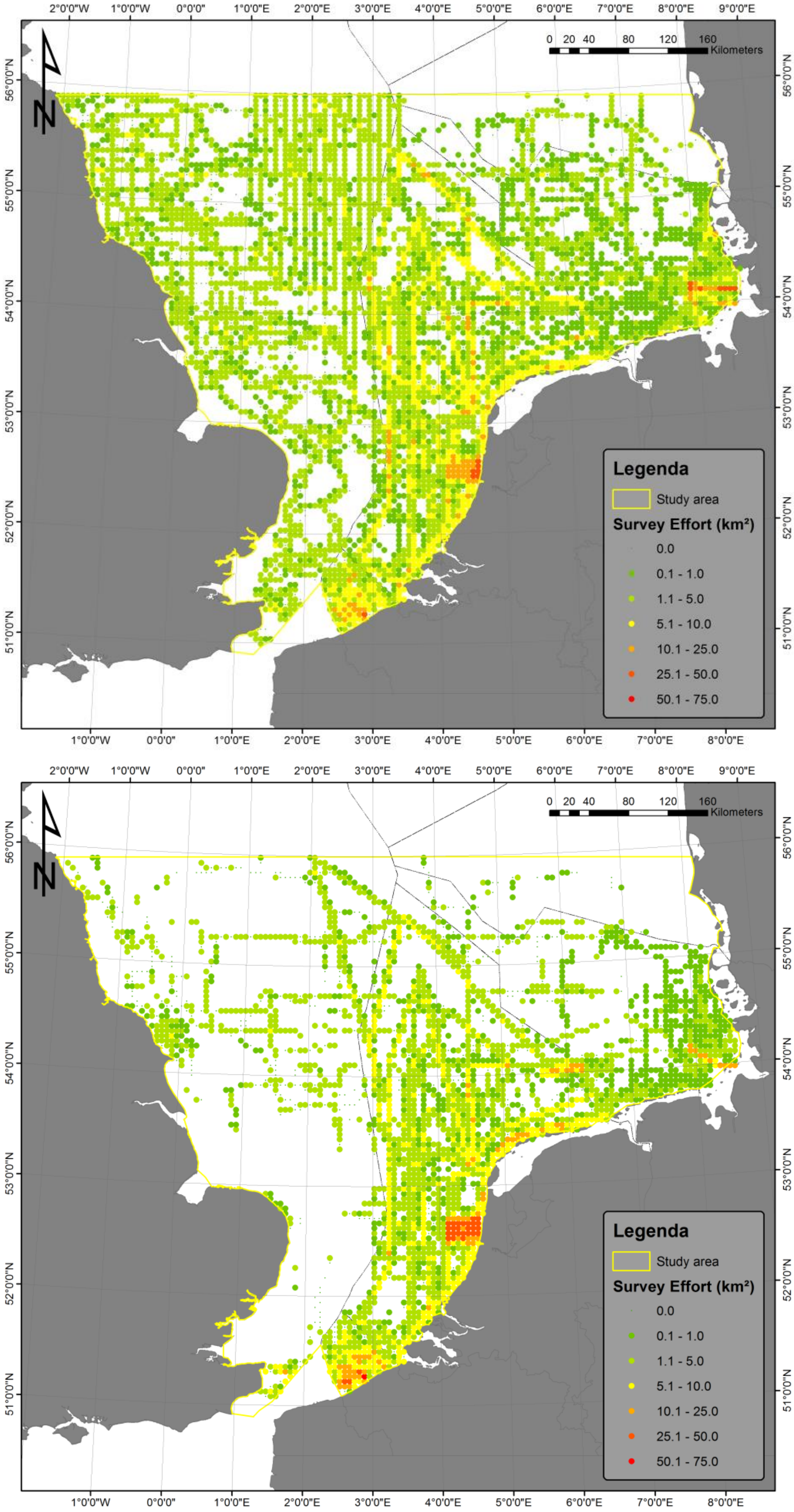


*Figure 4.4: Effort maps for August/September (Top) and for October/November (bottom).*

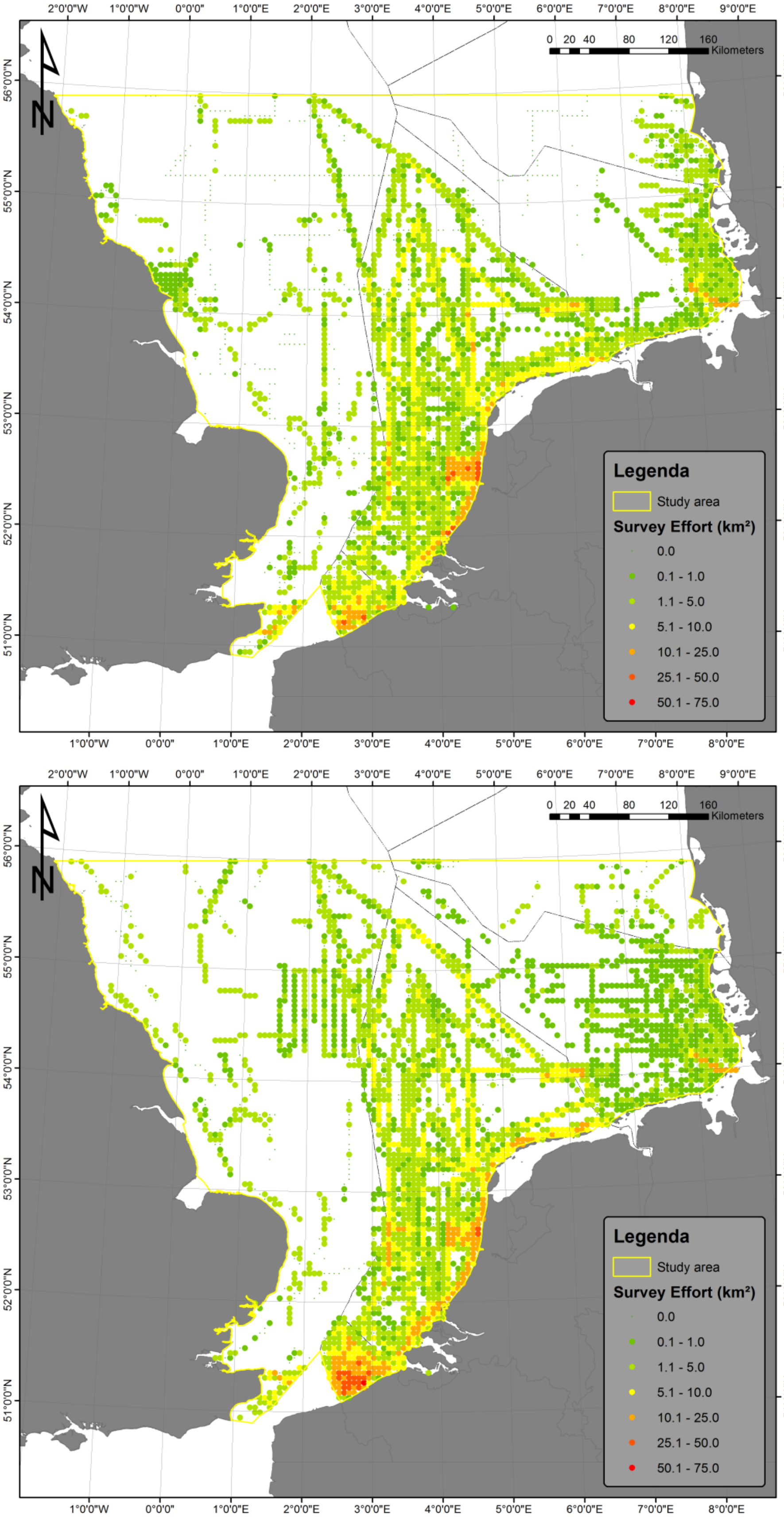
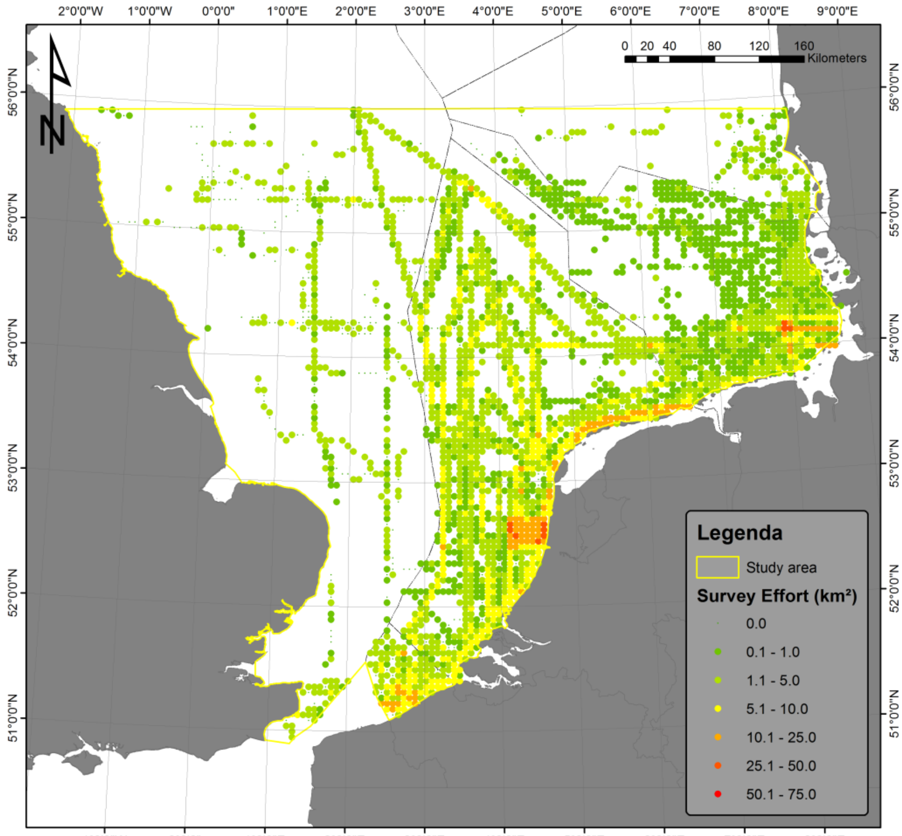


*Figure 4.4 (cont.): Effort maps for December/January (Top) and for February/March (bottom).*



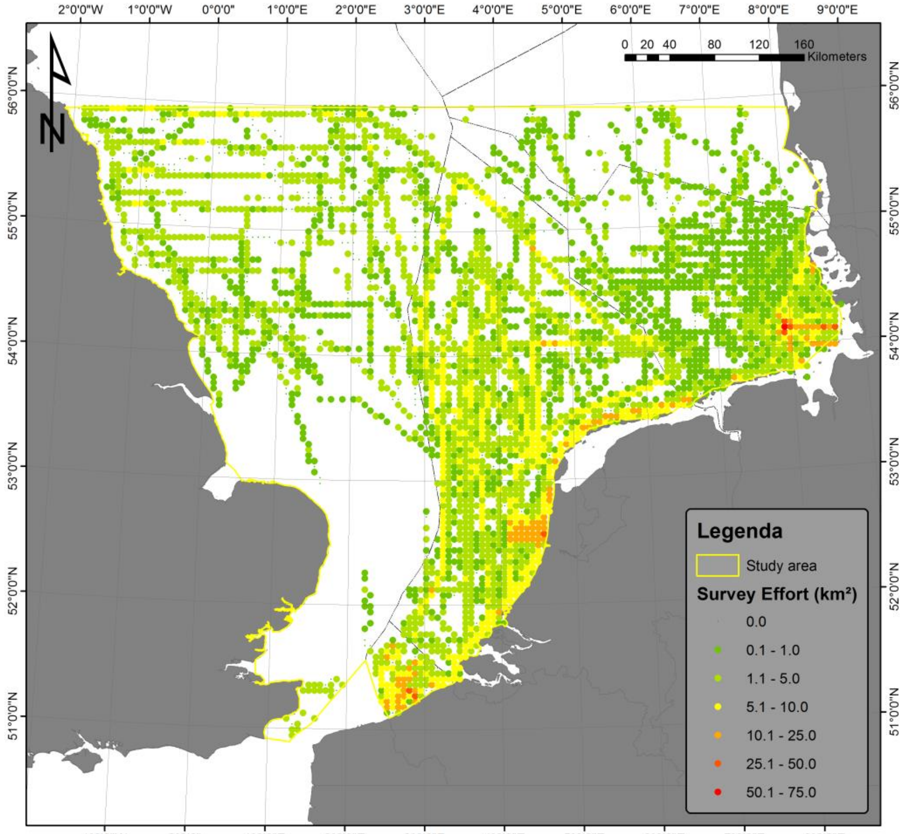


*Figure 4.4 (cont.): Effort maps for April/May (Top) and for June/July (bottom).*

#### 4.2.3.8 Interpolating the bird densities using GIS

After completion of the database operations, the resulting the bird densities were linked to the point dataset of the 5x5 km grid centerpoints in ArcMap, using the GRID_ID as the link. Using Python-script and the extension GeoStatistical Analyst, an inverse distance weighed (IDW) interpolation of the data points was done to improve geographical coverage. With the script this was automatically done for all 29 species of seabirds and 6 seasons (174 combinations). In order to conserve the results the geostatistical results where extracted to (a copy of) the grid centerpoint dataset. We also explored using Kriging as the interpolation method, but the results were often visually unsatisfactory, particularly because non-surveyed areas were all too easily filled in from either side, over distances not backed by species-specific semi-variograms, which show over which distances the data are spatially autocorrelated. Inverse distance weighting simply smoothed the grid cell values by averaging these with the distance from surrounding cells, given progressively less weight to cells at increasing distance.

#### 4.2.3.9 Final presentation

For final presentation the grid centerpoints with the bird densities where linked to the vector grid. Again using Python-script, layers were created for each of the 29 sea bird species and all 6 seasons all within a single Map Document. This Map Document also has layers for the coastline, OWF, shipping routes, marine protected areas, and EEZs. Using this Map Document a GeoPDF has been created. Also a PNG-file has been generated for each seabird/season for inclusion in a report.

### *4.2.4 Mapping shipping intensity*

A map of shipping intensity (or shipping density) that covers the study area was required. No existing and available map was fit for this purpose. The necessity of an updated map largely stems from a recent change to the shipping lanes in Dutch waters (per 1 August 2013). This change was made to free up space for OWF development relatively close to shore by moving shipping lanes. IMARES requested updated shipping density data from the Commissioner.

The shipping intensity map as documented by MARIN (2014) accurately documents the current distribution of shipping and intensity as observed in the period August 2013 through July 2014. From the GIS-datasets that were also made available, it became clear that the spatial cover of this dataset was insufficient for use across the study area, being the complete Southern North Sea (SNS). Shipping intensities were too low near the English East coast but also in German and Danish waters.

To remedy this, the following solution was developed: an available map produced earlier with better coverage across the southern North Sea (Van der Wal *et al.*, 2011a, 2011b) was updated using the new information from Dutch waters. Anatec (2008) has further details on the methodology behind their initial map. This ‘WindSpeed/Anatec map’ was updated to reflect the increased shipping intensity in places where the revived shipping lanes had moved and decreased shipping intensity in the old locations (where planned offshore wind farms are to be located).

The WindSpeed/Anatec-map had BIN=4 for the area overlapping with the unchanged part of the shipping lanes in Dutch waters. This value was also set for the new shipping lanes. In the areas freed up from shipping (but to be used for offshore wind farms) the BIN value was set to 2 where it was 3 or 4 and decreased to 1 where it was 2. The updated map was used to estimate how many birds might be displaced by shipping. Known relative shipping intensities and interpretation of the BIN/Rank in the WindSpeed/Anatec map are given in Table 4.6.

*Table 4.6. Relative intensity of shipping between bins in the WindSpeed/Anatec-map.*

| BIN | Rank | Interpretation | Relative intensity |
|---|---|---|---|
| 1 | <12 | less than one per month | 0.0019 |
| 2 | 13-52 | one per month to one per week | 0.0145 |
| 3 | 53-365 | one per week to one per day | 0.0777 |
| 4 | 366-8760 | one per day to one per hour | 1 0000 |
| 5 | >8760 | more than one ship per hour | 4.8928 |

A relative intensity of 1.0 was set for BIN=4 as this matches closely with the intensity in the Vessel Separation Scheme or VSS (the IMO system of shipping lanes, in the Southern North Sea), and an absolute estimate for this density can be taken from the current MARIN-map: 17.5 ships/1000 $km^2$.

The BIN values in the updated map are therefore taken to relate to shipping intensity as shown in Table 4.7.

*Table 4.7 Shipping intensities used in this study.*

| BINCMLbb | Shipping Intensity (ships / 1000 $km^2$) |
|---|---|
| 1 | 0.03 |
| 2 | 0.25 |
| 3 | 1.36 |
| 4 | 17.5 |
| 5 | 85.6 |

In estimating the number of birds avoiding an area due to shipping a final adjustment is made in relation to wind energy turbines. Across all projected offshore wind farms in the southern North Sea, average turbine density was calculated to be 1.66 turbine /$km^2$. Compensating for the difference in units (turbines/$km^2$) and shipping intensity (n/1000$km^2$) an initial relative impact factor of a ship vs. a wind energy turbine (WET) has been used of 0.000602.

Shipping induced displacement by birds has been calculated using the following formula:

BAtemp = bird density * shipping density * impact rel. to WET* 25

Where:

- bird density is the density for a sea bird species in a season (29 * 6 = 174 combinations).
- shipping density from the updated map as described here
- impact rel. to WET = 0.000602 as described above
- 25 $km^2$ is the size of a grid cell
- WET = wind energy turbine

The result (Figure 4.5) has been designated BAtemp because it does not yet include some of the aspects that are included in the approach chosen for this study which closely follows Bradbury *et al.* (2014).

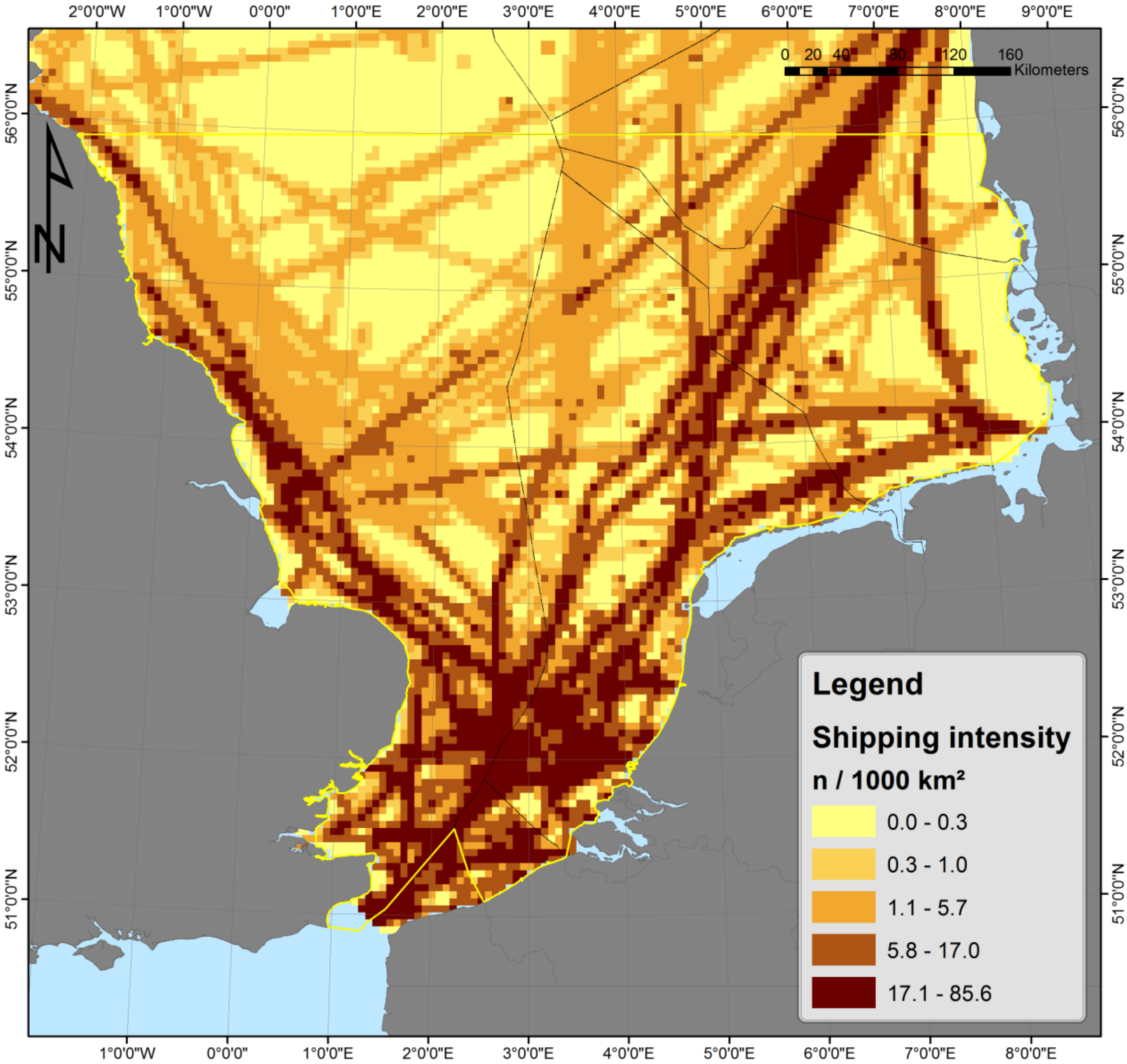


*Figure 4.5. Shipping intensity in the southern North Sea. Note that this figure gives the density of ships (n/1000km$^2$) per grid cell. How these values were substituted into ships/time is explained in section 4.2.4.*

Note that we consider a wind turbine equally deterring as a ship. They are however clearly different: ships pass by and thus only have a temporary effect on any given location, but ships travel and thus exert their pressure over greater distances than (stationary) wind turbines and their disturbing effect extends over much larger areas. There are, in fact, very few studies that substantiate the displacing effect of ships on seabirds. Schwemmer *et al*. (2011) found reduced densities of loons and seaduck within the major shipping lanes in the North and Baltic Seas. Likewise, Poot *et al.* (2011b) found reduced densities of auks in a busy North Sea shipping lane. Both studies show that ships displace seabirds more or less permanently from busy shipping lanes. The effect will be smaller of a single ship passing by, therefore we assumed a return time of one hour for each passing ship to take ship density into account, cf. Poot *et al*. (2014) and Zuur *et al.* (2014).

### *4.2.5 Seabird species*

Survey data from the southern North Sea have been compiled, using both ship-based surveys from all border states, as amalgamated in the European Seabirds at Sea (ESAS) database, kept at JNCC, Aberbeen, and aerial surveys (ESAS and Rijkswaterstaat, The Netherlands, for the Dutch Continental Shelf) have been used to describe seabird distribution patterns. The selected species are briefly covered in the section below. Rare seabirds are treated first, with all available sightings records plotted on a single map, using differently coloured symbols for six different “seasons”: August/September, October/November, December/January, February/March, May/April and June/July. The same seasons are used for the more common seabirds but these get a distribution map, showing densities (birds/km$^2$) for each season. In the accompanying texts, the current knowledge on relevant population size and wind farm vulnerability are highlighted. In this report, we consider the seabird species that commonly occur in the southern North Sea (sections 4.2.5.1 and 4.2.5.2). These are the major species occurring in the North Sea at large (Stone *et al.* 1995) and are important here, for numerical or conservation reasons. However, not all species are equally important regarding the risks of collision or displacement, as not all species are equally likely to occur in significant numbers in areas where offshore wind farms are projected. Therefore, we deal with individual species[2] in different ways:

1. Species, that are (rarely) reported in the southern North Sea (e.g. during sea watching: Camphuysen & Van Dijk 1983; Platteeuw *et al.* 1994) but that do not occur in the sightings databases used for this report, are not treated here. The list includes, among others, Great, Cory’s and Little Shearwater, (very) rare species of petrels and storm-petrels, Red-necked and Grey Phalaropes, and Roseate Tern.
2. The five European grebe species, Little Grebe, Great Crested Grebe, Red-necked Grebe, Horned Grebe, and Black-necked Grebe, all occur in North Sea coastal waters (on either side of the North Sea), and are unlikely to occur in offshore wind farms, because they all mostly live very closely nearshore, i.e. in waters where wind farms are not currently projected, spend most of their time swimming (rather than flying) and when they do fly, tend to do so below rotor height. Four out of these five grebe species are, therefore, not treated here. Great Crested Grebes are the most numerous grebes in the North Sea, mostly so in Dutch nearshore waters, and are the only grebe species listed in the appropriate tables and used in the modelling of usage of different parts of the southern North Sea. Likewise, waterfowl (other than Common Eider, Common Scoter and Velvet Scoter); waders; gulls that are either mostly land birds (Mediterranean Gull) or species that are rare and often missed by observers (e.g., Caspian and Yellow-legged Gull,) or that are even rarer (e.g., Sabine's Gull, Glaucous Gull and Iceland Gull); rare tern species (e.g., Gull-billed Tern, Caspian Tern, and all even rarer species); and other seabirds that are very rare in the North Sea (e.g., Brünnich's Murre) are not treated.
3. Species that are rare in the southern North Sea, i.e. have very few records in the sightings databases, but are of great conservation concern (e.g., are listed in Annex I of the EU Bird Directive), are treated briefly below. Dotmaps with positive records, with a short species account are presented for: Great Northern Loon, White-billed Loon, Balearic Shearwater, European Storm-petrel, Leach's Storm-petrel, Little Tern, Black Tern, Black Guillemot.
4. Bi-monthly maps are presented with a species account for the remaining species: Northern Fulmar, Sooty Shearwater, Manx Shearwater, Northern Gannet, Great Cormorant, European Shag, Common Eider, Common Scoter, Velvet Scoter, Arctic Jaeger, Great Skua, Little Gull, Black-headed Gull, Mew Gull. Lesser Black-backed Gull, European Herring Gull, Great Black-backed Gull, Black-legged Kittiwake, Sandwich Tern, Common Tern, Arctic Tern, Common Murre, Razorbill.

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[2] English bird names according to: http://www.worldbirdnames.org: Gill F. & Donsker D. (Eds) 2014. IOC World Bird List (v 4.4). doi : 10.14344/IOC.ML.4.4.

### 4.2.5.1 Rare seabirds

#### Great Northern Loon *Gavia immer*

Great Northern Loons are mostly nearctic breeders (Baffin Island, Greenland, Iceland). In Europe, the species winters mainly in nearshore waters off rocky shores, in Iceland, The British Isles, particularly Orkney and Shetland, northern Scandinavia, and the Atlantic coastline from Normandy to Iberia (Voous 1960; Del Hoyo *et al.* 1992). The southern North Sea, therefore, is not a staging area for this species, and birds seen here are mostly passing migrants. Single birds were seen along the Dutch and Belgian coastlines, and into the Channel, mostly in autumn and winter. Offshore records are rare and offshore wind farms are not considered a major problem for this species (Figure 4.6).

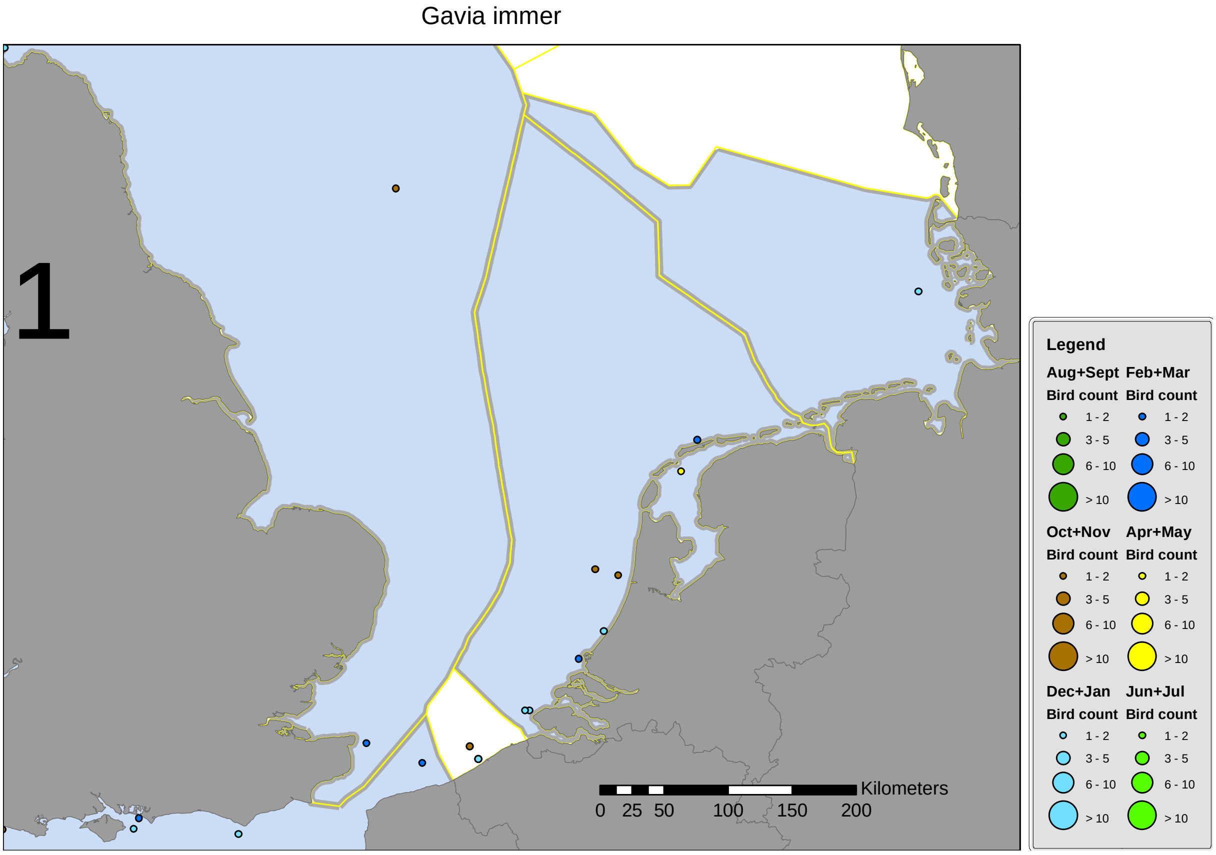


*Figure 4.6. Observations of Great Northern Divers in the southern North Sea (ESAS & MWTL databases).*

White-billed Loon *Gavia adamsii*

White-billed Loons are among the rarest seabirds in the North Sea and even world-wide. The species has a holarctic breeding distribution, showing, however, a conspicuous discontinuity in the North Atlantic (Voous 1960; Del Hoyo *et al.* 1992). Wintering grounds are poorly known; many birds supposedly remain at high latitudes year-round. The only known wintering ground of any importance in Europe was, until recently, the northern coastline of Norway, including Varangerfjord (>1500 individuals [Byrkjedal *et al.* 2000; Bell and Håland 2008]). Recent observations of migrating birds in spring through the German part of the Baltic Sea suggested either circum-Scandinavia migration, or the existence of unknown wintering grounds in the North Sea or further west or south (Bellebaum *et al.* 2010). This site may have been found in the central North Sea, where recent T-0 offshore wind farm seabirds surveys revealed the presence of an estimated 67 (Feb/Mar) to 157 birds (October) in the Dogger Bank area (Burton *et al.* 2013). These survey data have not been forwarded to ESAS; ESAS just holds two offshore sightings records, both in the Dogger Bank area (Figure 4.7), i.e. conform the findings of Burton *et al.* (2013). This newly discovered wintering ground is highly relevant to offshore wind development, given the supposedly high vulnerability of the species and the small relevant biogeographical population (10 000 birds, Burton *et al.* 2013).

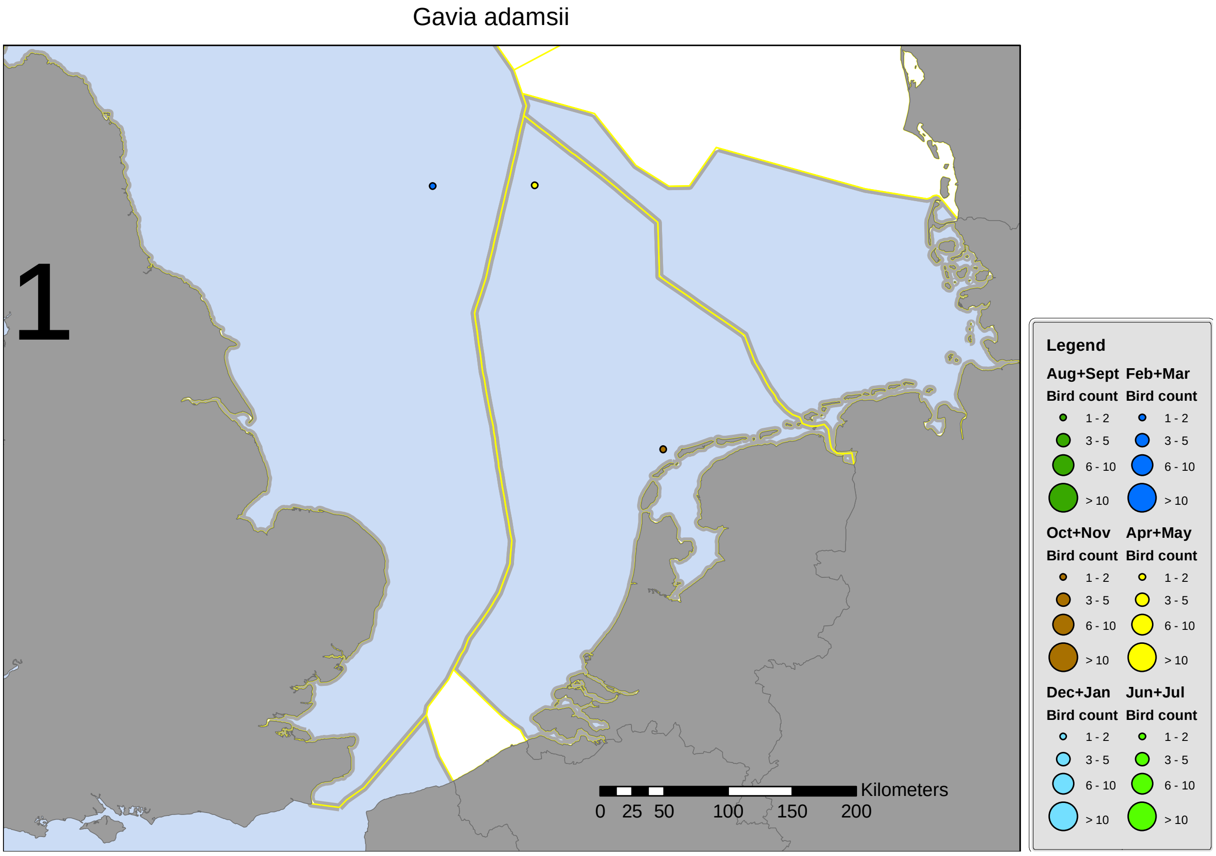


*Figure 4.7. Observations of White-billed Divers in the southern North Sea (ESAS & MWTL databases).*

Balearic Shearwater *Puffinus mauretanicus*

Until recently, the Balearic Shearwater was considered a vagrant in the North Sea, but in recent years the species is seen each year, both from the shore and further offshore, in steadily increasing numbers (Bos *et al.* 2012). The species is mostly recorded from July through October with autumn records and even winter records becoming more numerous (e.g. Van Dijk 2009). The increase in numbers of recorded birds is probably genuine (Wynn *et al.* 2007; Votier *et al.* 2008). The species is a rare endemic to the western Mediterranean with a very small and decreasing population size [(2200 breeding pairs in the Balearic Archipelago, (BirdLife International 2008)]. The at-sea biology and migration patterns are poorly known. The species seems to be increasingly using fisheries discards as food in the Mediterranean (Navarro *et al.* 2009), but may be mainly self-feeding in the Atlantic off Portugal, after breeding (Poot 2005). The species appears to be moving ever further north along the Atlantic seaboard from Iberia, and into the North Sea, possibly in reaction to climate change (Wynn *et al.* 2007; Votier *et al.* 2008) and this rare southern European endemic may meet offshore wind farms increasingly often in the years to come. At-sea records of Balearic Shearwaters have been made throughout the study area, particularly in Dutch waters (Figure 4.8), for which relatively much recent survey effort is available. This suggests that the species may occur anywhere in the study area. Larger numbers have been seen further southwest, in the Approaches to the English Channel and adjoining waters (Stone *et al.* 1995), indicating a 'reservoir' of birds just south of the study area that is relevant given the developments in distribution related to climate change.

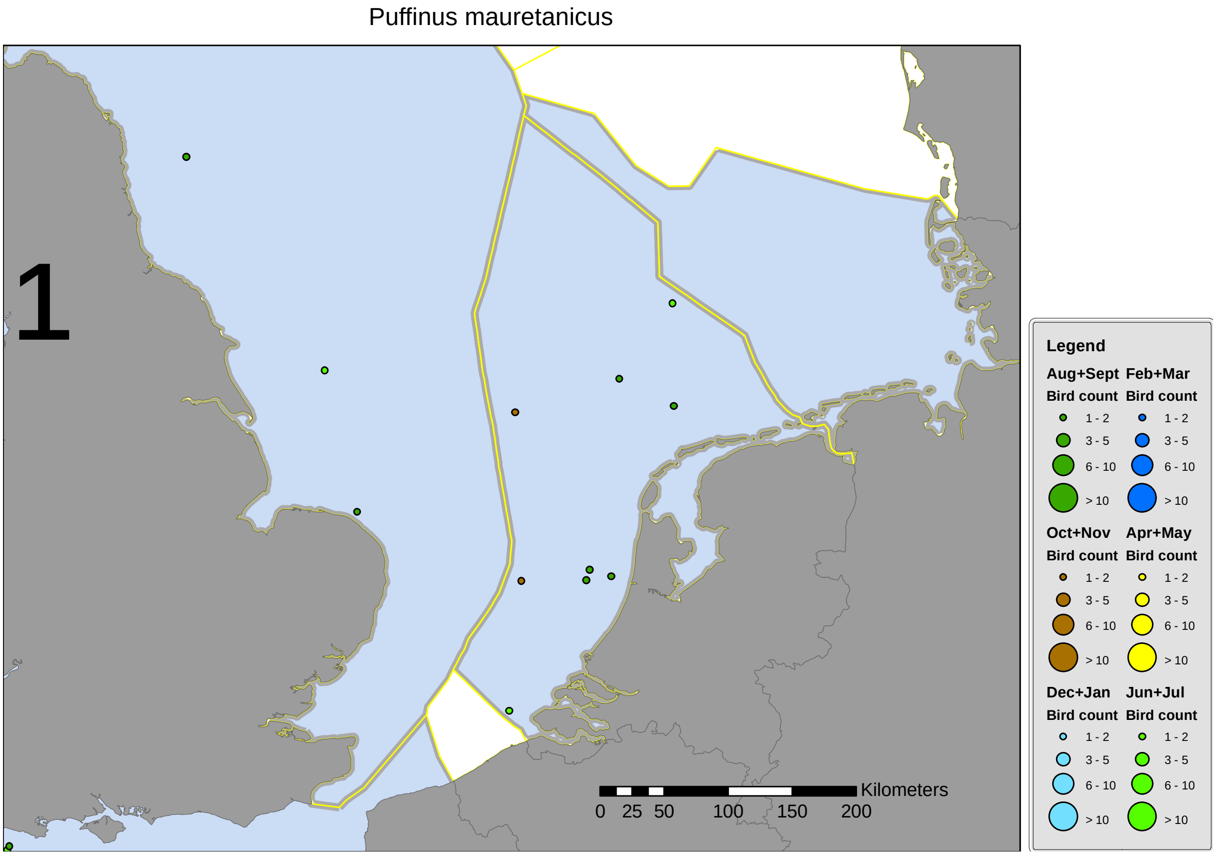


*Figure 4.8. Observations of Balearic Shearwaters in the southern North Sea (ESAS & MWTL databases).*

### European Storm-petrel *Hydrobates pelagicus*

European Storm-petrels breed on rocky shores and islands, with roughly 100 000 pairs in Britain and Ireland, mostly along the northern and western shores. No colonies are known in the southern North Sea (Mitchell *et al.* 2004). Large numbers are also found in the Faeroe Islands (150-400 000 pairs), Iceland (50-100 000), Norway (numbers not well known, distributed up to the Barents Sea) and France/Iberia also have breeding populations. In the North Sea, the species is concentrated in the northwest (Stone *et al.* 1995), spilling over into the northwest of the study area (Figure 4.9). Summer records may be tied to Scottish colonies, autumn records probably concern migrants from Scotland and northern Europe, en route to the wintering quarters off Africa.

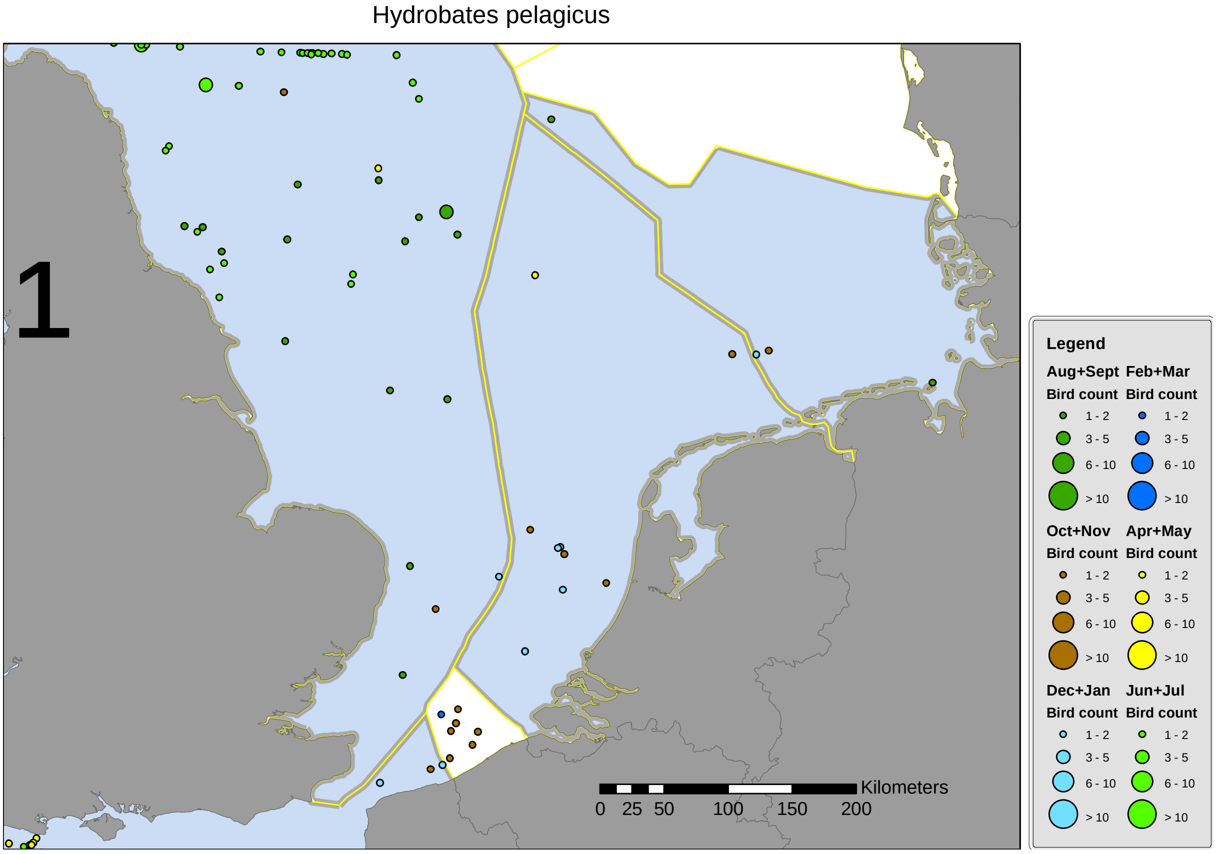


*Figure 4.9. Observations of European Storm-petrels in the southern North Sea (ESAS & MWTL databases).*

### Leach's Storm-petrel *Oceanodroma leucorhoa*

Leach's Storm-petrels are generally more ocean-going than European Storm-petrels. Breeding colonies in the British Isles are further to the northwest (as compared to those of their smaller relatives). The majority of the European birds probably breed in the Westmann Islands (Iceland: 80-150 000 pairs) and in the Faeroe Islands and Norway, but numbers are not well known (Mitchell *et al*. 2004). Birds entering the North Sea are migrants and mainly seen in autumn (Camphuysen & Van Dijk 1983; Platteeuw *et al*. 1994; Stone *et al*. 1995). The probability of sightings in any given locations seems to be strongly related to survey effort, with clusters of records in well-surveyed areas such as the well-studied OWEZ/PAWP sites in The Netherlands and the OWF sites on the Belgian part of the North Sea (Figure 4.10).

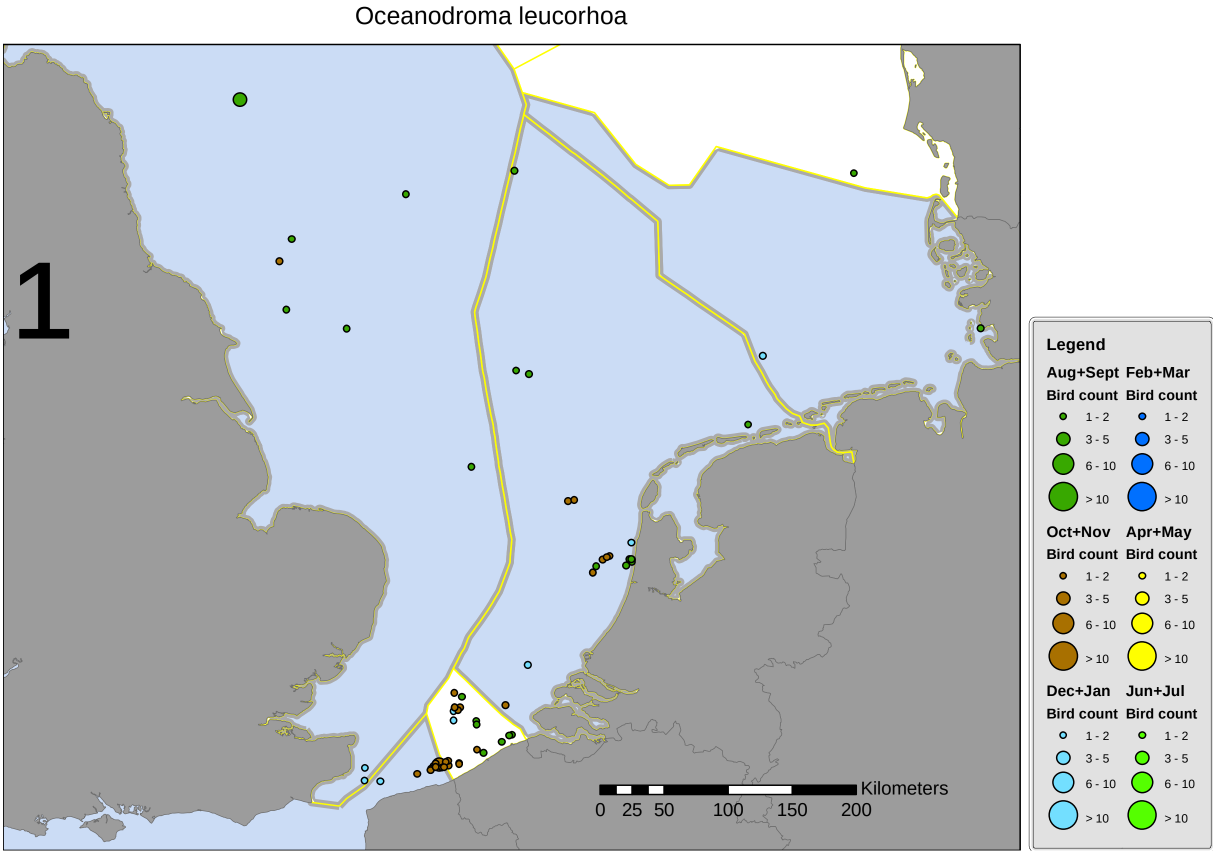


*Figure 4.10. Observations of Leach's Storm-petrels in the southern North Sea (ESAS & MWTL databases).*

### European Shag *Phalacrocorax aristotelis*

European Shags are, as breeders in the North Sea, confined to the northwest, the southernmost (large) colonies just touching the study area. Moreover, Shags do not normally venture far out to sea and find most of their food in coastal waters (Bogdanova *et al.* 2014). Most birds were therefore seen in the vicinity of the colonies (Figure 4.11), but some (mostly young) birds do cross the North Sea, as evidenced by at-sea records and sightings of colour-ringed birds along the European mainland (Harris 2001). A recent sighting of a bird colour-ringed in France in the Marsdiep area (western Wadden Sea; e.g., http://waarneming.nl/waarneming/view/87413613) demonstrated that birds may also reach the study area from further south.

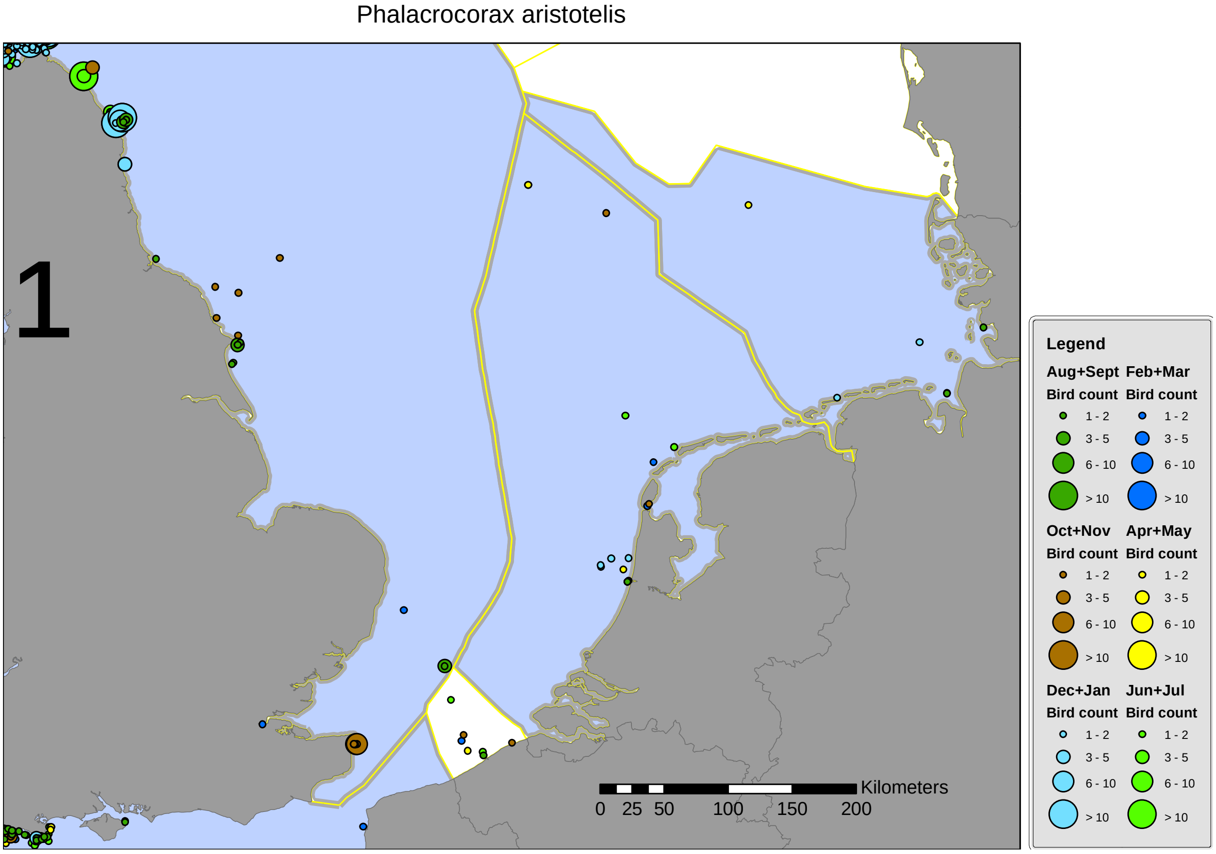


*Figure 4.11. Observations of European Shags in the southern North Sea (ESAS & MWTL databases).*

Pomarine Skua *Stercorarius pomarinus*

Pomarine Skuas breed in the tundra zone of Russia, Canada and Alaska and winter in coastal waters in the tropics (Del Hoyo *et al.* 1996). On migration, most bird moving to wintering areas in the Altantic take a route west of the British Isles, thus avoiding the North Sea. However, there is also weak yearly passage through the North Sea and in some years their occurrence here shows characteristics of an invasion (Camphuysen & Van IJzendoorn 1988a,b; Van den Berg & Bosman 1999). Invasions involve juvenile birds, and seem related to cyclic good breeding success in the Arctic, in response to the lemming-cycle (Camphuysen 1987). At sea, Pomarine Skuas were found scattered over the study area, mostly in autumn (Figure 4.12). Concentrations of sightings in The Netherlands, Belgium and in the inner German Bight are effort-related.

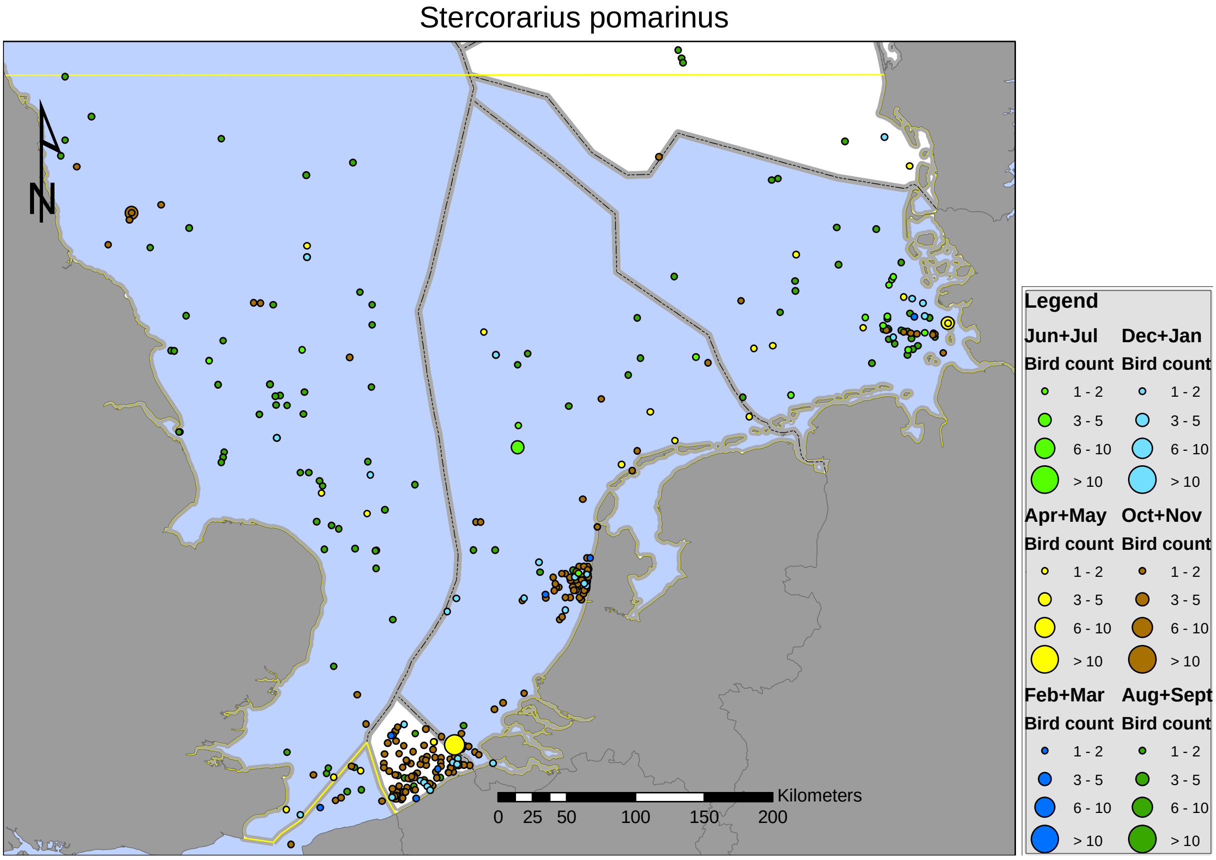


Figure 4.12. Observations of Pomerine Skuas in the southern North Sea (ESAS & MWTL databases).

### Long-tailed Jaeger *Stercorarius longicaudus*

Long-tailed Jaegers breed in Arctic and subarctic uplands around the North Pole, i.e. have a more southerly distribution than Pomarine Skuas. In contrast to this species, Long-tailed Jaegers also breed in Greenland and Scandinavia, and winter further south, also circumpolar, in the subantarctic (Del Hoyo *et al.* 1996). This is the most pelagic of the skuas and jaegers, and also the rarest species in the North Sea. However, birds that enter the North Sea (mostly on autumn migration, Figure 4.13) often migrate along the continental coastline and the species is seen relatively often by Dutch seawatchers, over the full length of the Dutch coastline. Records of these passage migrants are made from May through November, peaking in September (Van den Berg & Bosman 1999). Offshore records of this species are relatively rare.

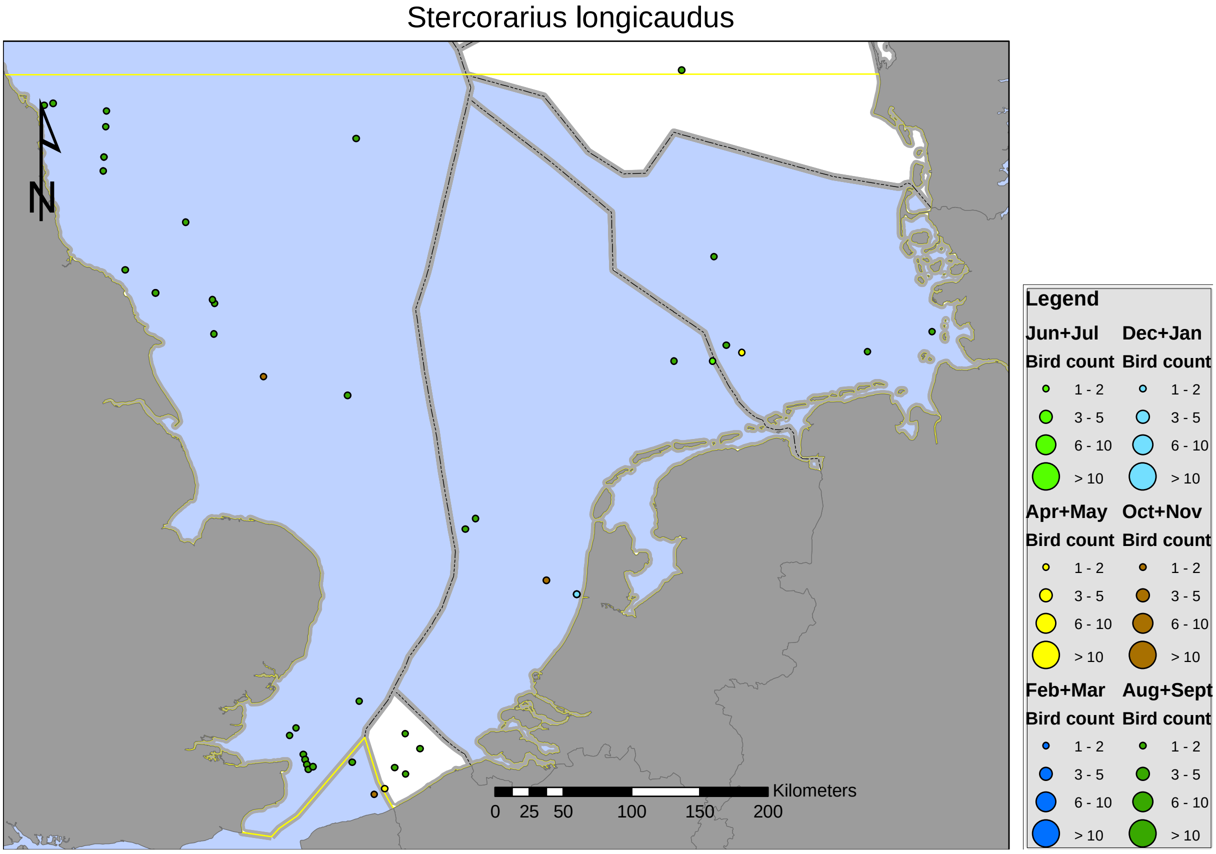


Figure 4.13. Observations of Long-tailed Jaegers in the southern North Sea (ESAS & MWTL databases).

## Little Tern *Sternula albifrons*

Little terns form a “superspecies”, a cluster of closely related species/races that breed dispersed over much of the world’s coastlines (Voous 1960; Del Hoyo *et al.* 1996). Around the southern North Sea, hundreds of pairs breed along the English east coast, in the Low Countries and in Germany each (Mitchell *et al.* 2004). At sea, Little Terns have a very coastal distribution and offshore records are rather exceptional (and referring to migrants). Most migration takes place along the coastlines of the North Sea, peaking in early May (Platteeuw *et al.* 1994). Autumn migration is less significant over the North Sea, suggesting more over-land migration. Most logged records at sea are very close to the mainland-Europe shores, mostly well landward of any projected offshore wind farms. UK nearshore waters have been mainly surveyed by plane (Bradbury *et al.* 2014) and these data are not available through ESAS. However, Perrow *et al.* (2006) published results from studies done off the English east coast, in relation to Scroby Sands Round 1 wind farm, showing a picture very similar to that along the eastern North Sea seaboard, with Little Terns not venturing far out to sea, concentrating in the Wash, Humber and Thames estuaries and staying away from much of the more open parts of the North Sea (Figure 4.14). However, Perrow *et al.* (2011) caution that pile driving for nearshore wind farms may impact local herring abundance, which in turn can negatively impact Little Tern foraging and breeding success.

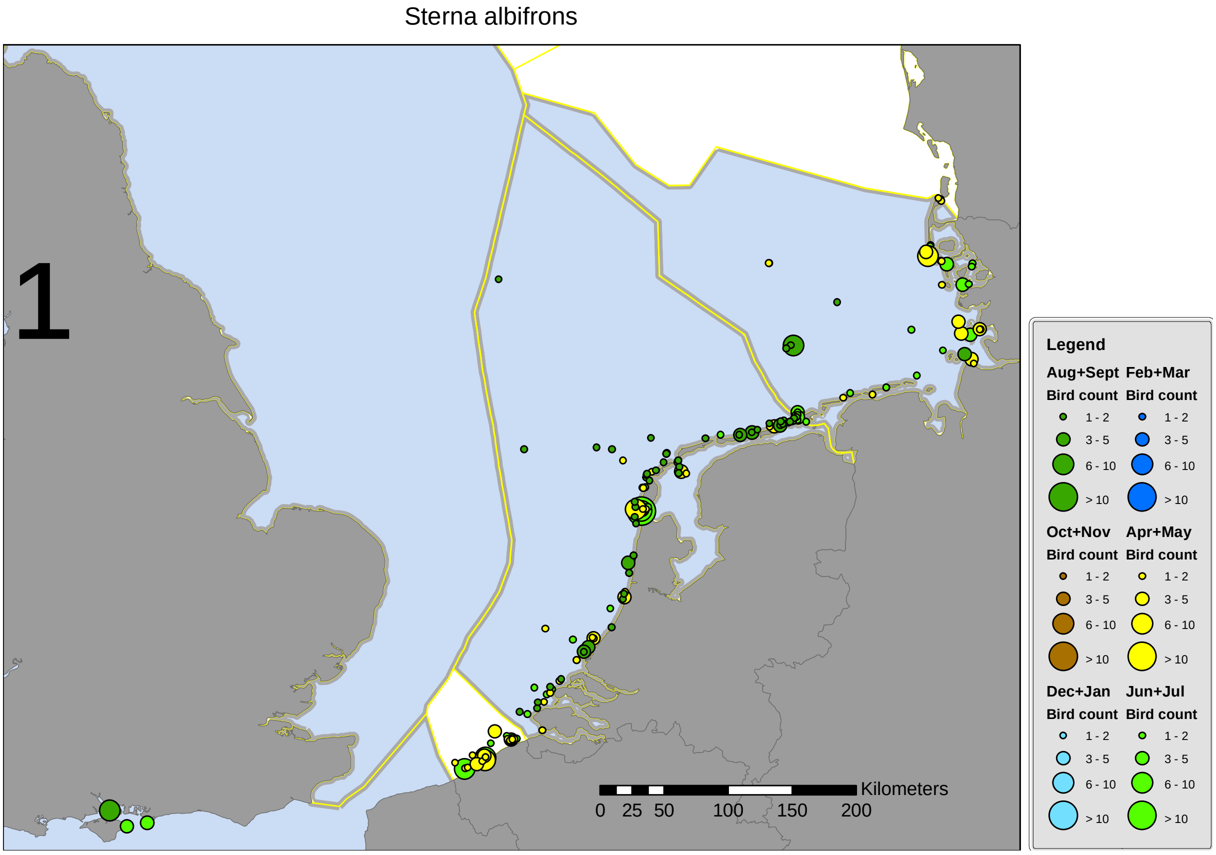


*Figure 4.14. Observations of Little Terns in the southern North Sea (ESAS & MWTL databases).*

### Black Tern *Chlidonias niger*

Black Terns breed dispersed in moorlands over much of temperate Europe, between 40°N and 60°N, with their westernmost colonies in The Netherlands and Belgium, near the North Sea (Voous 1960). In the breeding season, Black Terns are ‘marsh terns’ that are restricted to fresh water habitats, but in winter they move to the Atlantic, off western and southern Africa. During the transition from fresh water to oceanic habitats (and *vice versa*), they migrate along the North Sea mainland coastline (Van der Winden 2002), and may also forage in North Sea coastal waters (Figure 4.15). Some move further out to sea and even cross over, to e.g., the Thames estuary, and some early arrivals have been noted in German offshore waters.

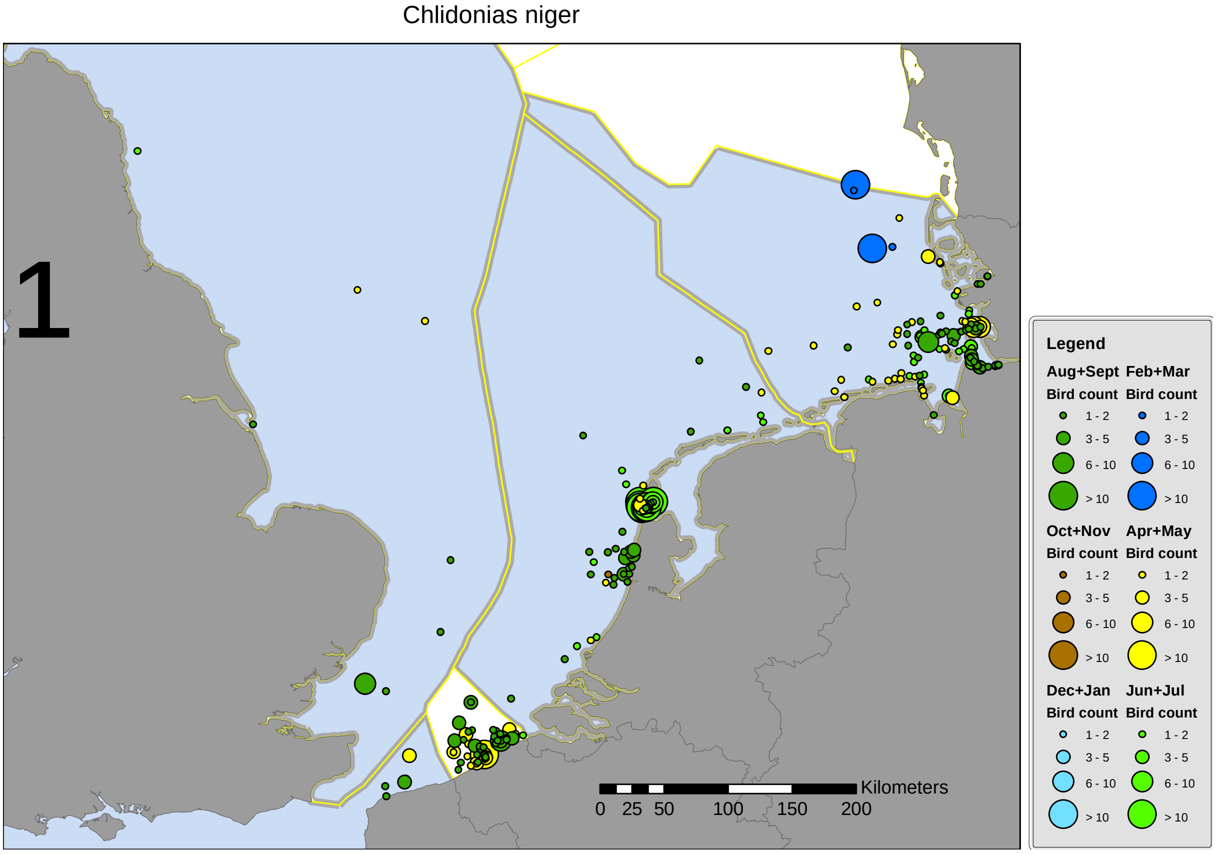


*Figure 4.15. Observations of Black Terns in the southern North Sea (ESAS & MWTL databases).*

### Black Guillemot *Cepphus grylle*

In NW Europe, Black Guillemots breed on rocky shores in western and northern Britain and Ireland, Faroe, Iceland, Spitsbergen and Scandinavia (Voous 1960). Off all the European seabirds, they are probably tied most strictly to land, and rarely venture more than a few km from their colonies. Records in the Low Countries, with their unsuitable sandy coastlines, are therefore rare (e.g., Van den Berg & Bosman 1999). In the open North Sea, Black Guillemots are very rare (Figure 4.16). Most birds quickly move to 'surrogate rocky shores' such as dikes, harbours and piers (i.e. outside the realm of offshore ship-based or aerial surveys), or manage to find the only true rocks, offshore in the study area, Heligoland.

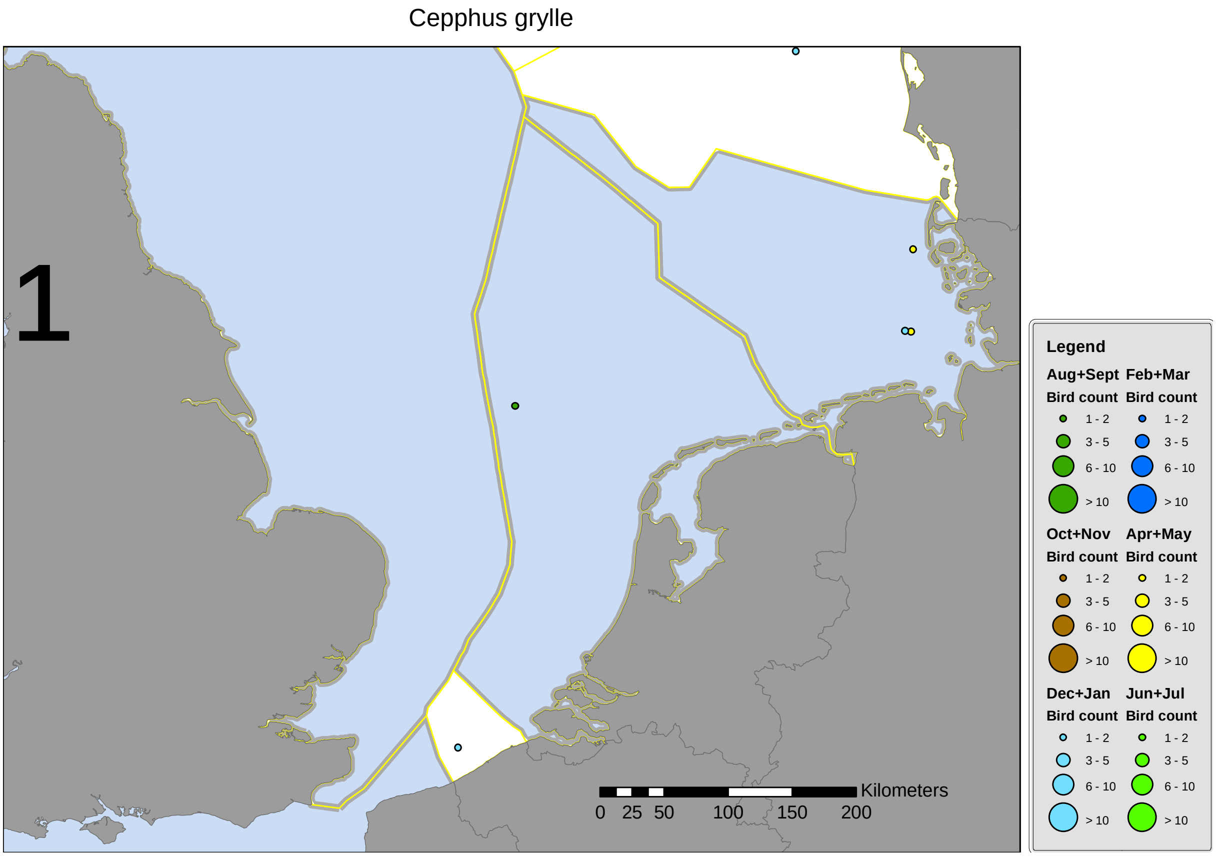
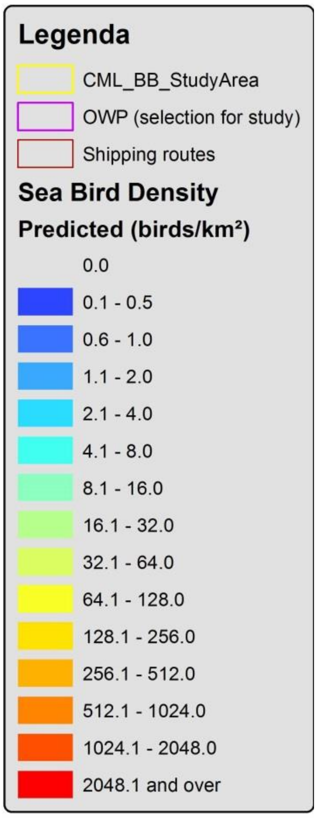


*Figure 4.16. Observations of Black Guillemots in the southern North Sea (ESAS & MWTL databases).*

### 4.2.5.2 Common seabirds

Bird densities have been (colour) coded to the same key (Box 1.)



*Box 1. Colour codes used for densities of common seabirds depicted in Figures 4.17-4.42*

### Red-throated Loon *Gavia stellata* & Black-throated Loon *Gavia arctica* ("small loons")

For Red-throated Loons, the North Sea is a major wintering area and in spring, also an important moulting area (particularly German and Danish waters: Skov *et al.* 1995). Black-throated Loons are probably mostly passage migrants, but small numbers also winter in the North Sea. In winter, the species are very similar in appearance and cannot always be separated when seen at distance, therefore the two species are taken together. We note, however, that the vast majority of loons wintering in the southern North Sea are Red-throated Loons (Camphuysen & Leopold 1994; Dierschke *et al.* 2012). In the northeast of the study area, relative numbers of Black-throated Loons are probably highest, at 22% of all small loons (Christensen *et al.* 2006), although more recent surveys here set this percentage back to 9% (Petersen *et al.* 2014), which is more in line with percentages found elsewhere. Further south, Black-throated Loons are seen mostly in spring, when there is a short but marked migration peak along the Dutch coastline (Camphuysen & Van Dijk 1983; Platteeuw *et al.* 1994).

An estimated 38 000 small loons winter along the eastern seaboard of the southern North Sea, concentrating in the German Bight (Skov *et al.* 1995), and roughly 10 000 birds winter of the English east coast, concentrating in the outer Thames (O'Brien *et al.* 2008, 2012). These small loons are highly sensitive to disturbance, either from shipping traffic (Schwemmer *et al.* 2011) or from offshore wind farms (Rexstad & Buckland 2012; Leopold *et al.* 2013a; Furness *et al.* 2013).

Small loons arrive in the study area in August/September. Their numbers build up further in October/November, to remain high over winter, with the highest densities in waters off NW Germany. In this same area, numbers peak in spring, when birds gather here in large numbers to moult. On the UK side, a large concentration starts building up off the Norfolk coast in autumn (October/November) and densities remain high here, and over a wide area stretching quite far offshore, until April/May (Figure 4.17).

Small loons (Red-throated Loons) were found to avoid offshore wind farms to a large extent in most post-construction studies, but none of these were done in areas with high pre-construction densities of loons. Walls *et al.* (2013) found a significant reduction in presence in the Robin Rigg offshore wind farm, west-Scotland (70% reduction in numbers of swimming birds, at low general densities); Leopold *et al.* (2013) aslo found statistically significant displacement from offshore wind farm OWEZ (also at low general densities, at the fringe of the area where loons occurred more abundantly); Rexstad & Buckland (2012) found similar suggestions of displacement in their study too, but the wind farm (Kentish Flats) here was located at the fringe of the bird concentrations, hampering statistical analysis. Loons completely avoided the Horns Rev offshore wind farms after construction, despite having been present in average densities prior to construction (Petersen *et al.* 2006a) and, similarly, no loons were recorded within the wind farm sites Nystad (Denmark, Baltic; Petersen *et al.* 2008), Horns Rev 2 (Petersen *et al.* 2014), or Alpha Ventus (Mendel *et al.* 2014), showing total avoidance again at low background densities.

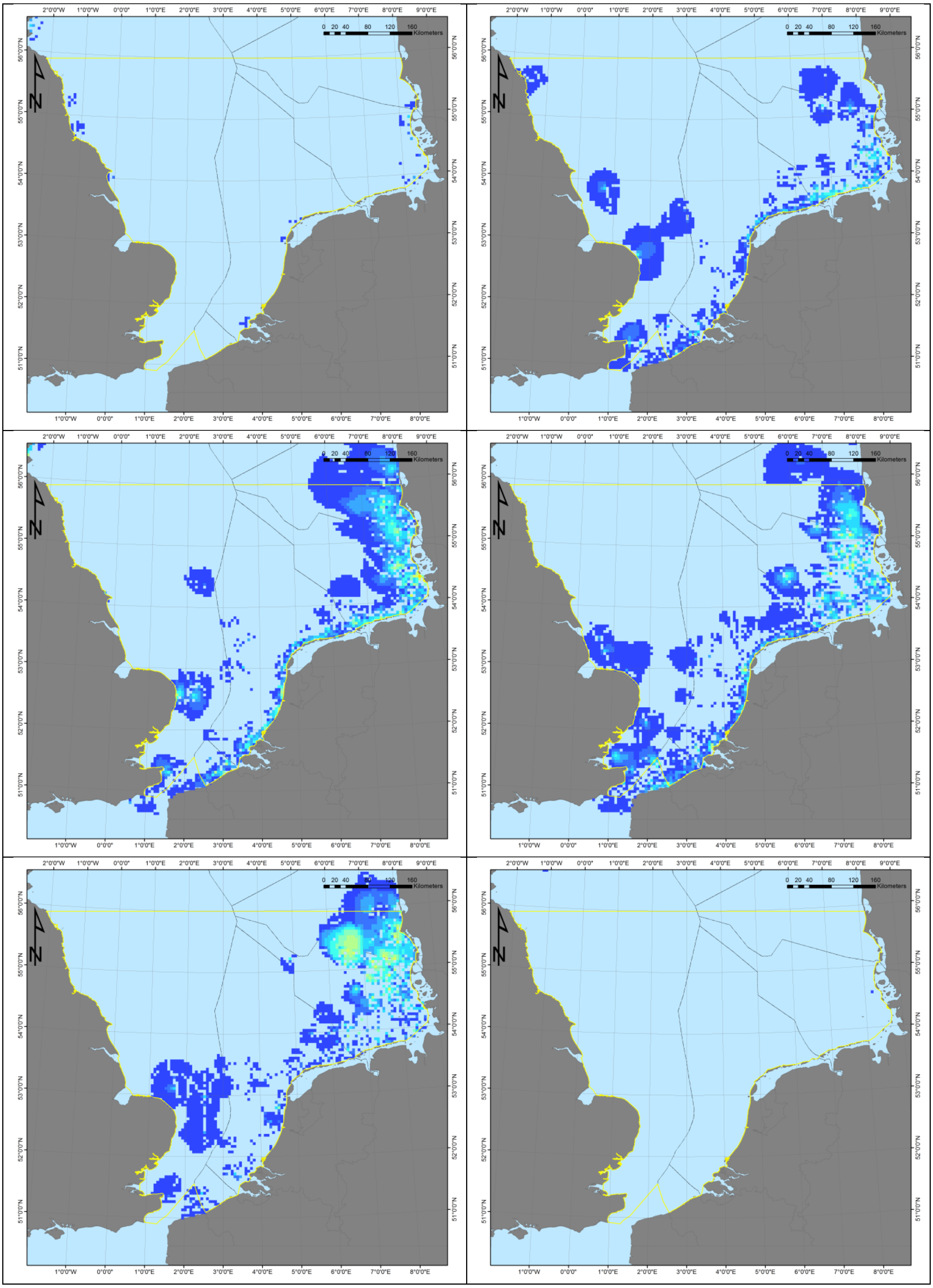


*Figure 4.17. Small loons: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

Great Crested Grebe *Podiceps cristatus*

Great Crested Grebes are mostly freshwater birds, but since the turn of the century, increasing numbers spend the winter in the North Sea, particularly off the Dutch mainland coast. Another 2500-2800 grebes are estimated to winter off the Belgian coast (Stienen & Kuijken 2003; Vanermen *et al.* 2013). Numbers at sea off the English coast are negligible, even in the Thames estuary which would seem to be the most suitable for the species (Dean *et al.* 2003). The most important wintering site in the southern North Sea is a narrow coastal strip of sea between Hook of Holland and Den Helder. Total numbers in these parts have been estimated at 20-40 000 birds. Most of these birds winter very closely inshore, e.g., landward of the existing offshore wind farms PAWP and OWEZ in this region (Van Bemmelen & Leopold 2013).

The distribution maps show a higly coastal, and mostly continental distribution pattern (Figure 4.18). Numbers at sea start building up in September/October, apparently starting from Belgian waters and spreading north, south and west from there in subsequent months. Peak numbers are present from December-March, and in spring numbers go down again quickly, with the distribution retracting back to Belgian waters. In the summer months, Great Crested Grebes are largely absent from the North Sea.

Very few wind farm impact studies have been conducted in marine waters with significant numbers of grebes. The most nearshore wind farm in The Netherlands, OWEZ, is situated on the outer fringe of the distribution of Great Crested Grebes wintering off the mainland coast. Despite low general densities at wind farm longitudes, a significant displacement effect of this wind farm was found (Leopold *et al.* 2013a). In the Nystad offshore wind farm the related Red-necked Grebe P. griseigena were seen just outside, but never within this wind farm (Petersen *et al.* 2008).

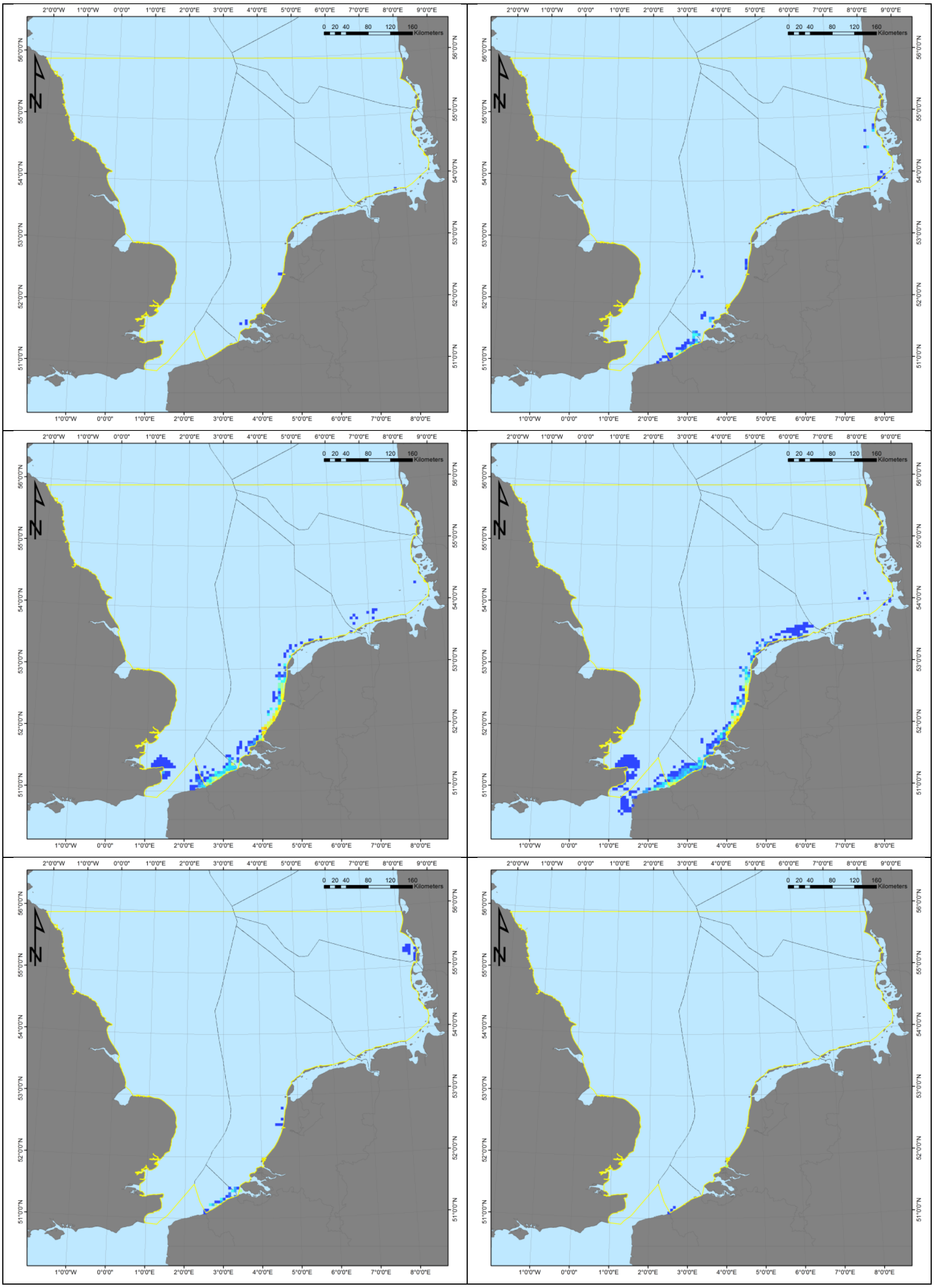


*Figure 4.18. Great Crested Grebe: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Northern Fulmar *Fulmarus glacialis*

Northern Fulmars breed on cliff coasts and islets around the British Isles, Faroe, Svalbard, Jan Mayen, Iceland and from Norway to northern Russia (Mitchell *et al.* 2004), and are generally more abundant in the northern and central North Sea than in the southern North Sea (Stone *et al.* 1995). In the Southern North Sea these birds tend to avoid waters influenced by rivers, i.e., they mostly stay away from a broad band of coastal waters off Europe's mainland and around Norfolk, UK (Figure 4.19).

Northern Fulmars are among the most numerous seabirds in the North Sea at large. The Scottish breeding population alone has 281 680 pairs (Mitchell *et al.* 2004) and birds from many more colonies further north visit the North Sea in the non-breeding season. The relevant European breeding population has an estimated size of 6 951 925 individuals (BirdLife International 2004), of which 1 872 000 birds are estimated to be present in the North Sea in winter (Skov *et al.* 2007). Given the high mobility of this species, many more individuals than this last figure are likely to move through the North Sea in any year.

The bimonthly distribution maps clearly show that Fulmars tend to avoid coastal waters and that densities are high in some of the northern parts of the southern North Sea. A central area in these northern parts mostly holds low densities, but very high densities may be found in other parts, often exceeding 250 birds/km$^2$ within such concentrations. This is probably a result of a few very high point counts, from which the high density values are smoothed outward, by the algorithms used. High point counts are often related to fisheries, as birds flock around fishing vessels discarding unmarketable fish and offal. The resulting concentrations of birds are thus a direct result of human activities but can only occur in areas where the birds are generally present in high numbers (Skov & Durinck 2001), suggesting that the peaks in bird densities visible on the various distribution maps are genuine, but only in a general sense. High densities may occur in the areas indicated, but cannot be predicted for any one moment, if fisheries activities cannot be predicted. Should fisheries be expelled from a site, for instance because of a closure in a future wind farm site, such fisheries-induced peaks in densities will no longer occur at the site.

Given that most Northern Fulmars live offshore, and that most wind farm impact studies to date have been conducted nearshore, it will be difficult to predict wind farm impact on this species, in offshore areas. At the fringe of Northern Fulmar distribution, Leopold *et al.* (2013) found a micro-distribution pattern that suggested total avoidance of wind farms (in a low density situation). Likewise, Vanermen *et al.* (2013) found indications that Northern Fulmar are displaced from offshore wind farms in Belgium, but due to low general densities, this could not be substantiated further. They even write: "the uncertainty of the obtained result is 100%".

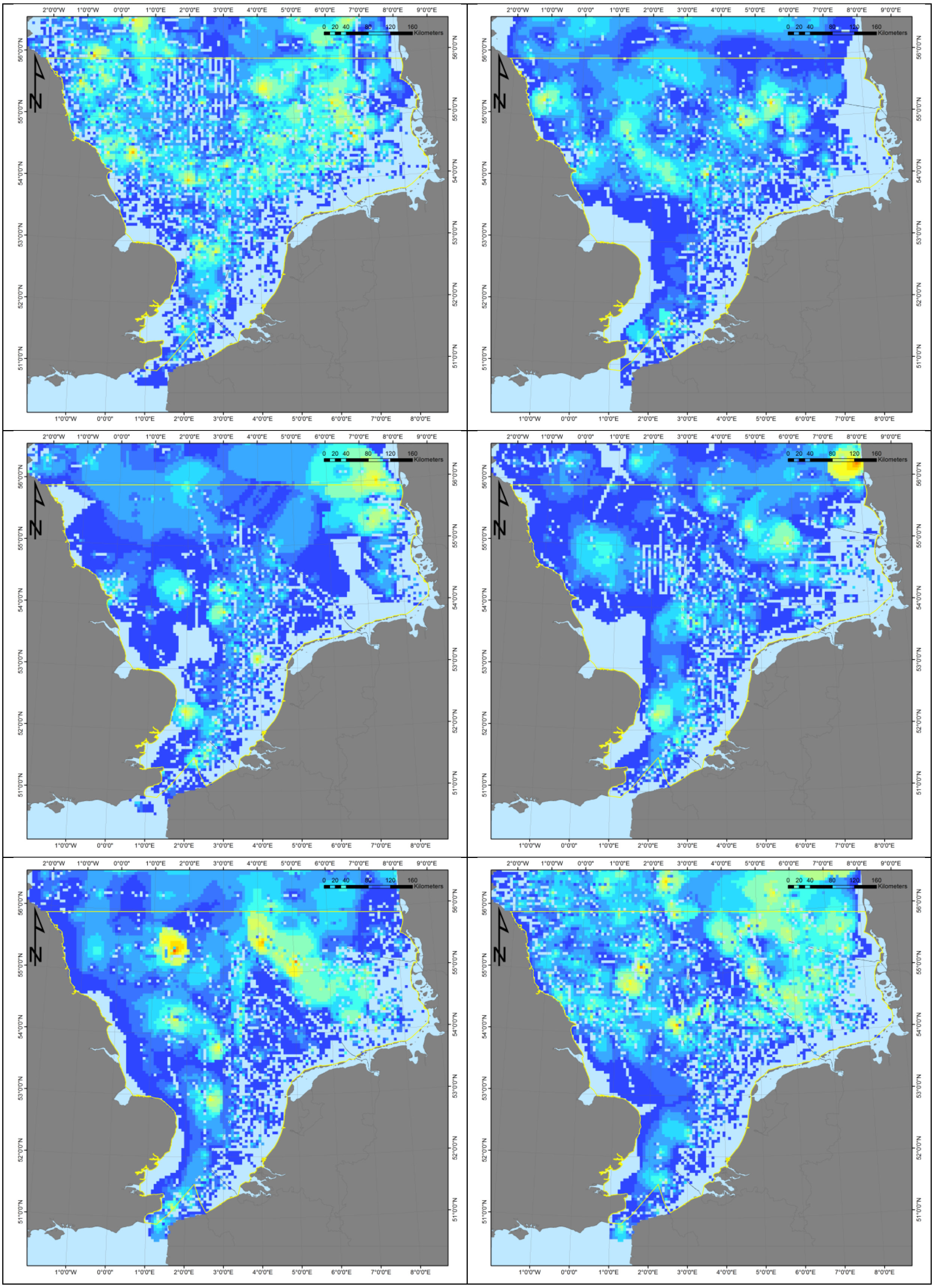


*Figure 4.19. Northern Fulmar: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Sooty Shearwater *Puffinus griseus*

Sooty Shearwaters breed in two regions in the Southern Hemisphere: around New Zealand and Australia and in South America, at the Falkland Islands (Malvinas), and on islands off Chile (Del Hoyo *et al.* 1992; http://www.birdlife.org/datazone/speciesfactsheet.php?id=3933). The South American population is relevant to this study. After breeding, these birds migrate northward, and spend their austral winter in the North Atlantic, between Baffin Island and the Mid-Atlantic Ridge, on the Grand Banks and the Newfoundland and Labrador shelves. The birds arrive here in April and leave by September. Some birds cross the Atlantic and move to the Ireland/ Rockall Trench (Hedd *et al.* 2012) and disperse into an area extending at least from the Faroe Islands to the Bay of Biscay (Pollock *et al.* 1997; 2000; Taylor & Reid 2001; Mackey *et al.* 2004). Both adult birds in active moult, and juveniles in fresh plumage were recorded in the NE Atlantic, near Rockall, about 350 km west of the Scottish mainland, in July (Keijl 2011). Numbers in the southern North Sea peak in Europe in August/September. From around Scotland, some birds move into the northwestern North Sea in these months (Stone *et al.* 1995) and some of these move on, through the southern North Sea to exit through The Channel. Dutch seawatchers see Sooty Shearwater passage in September/October. Most of these birds are probably juveniles or immatures, that need not arrive early in the Southern Hemisphere breeding colonies, as they are not yet old enough to start breeding (Camphuysen & Van Dijk 1983; Platteeuw *et al.* 1994). The total population size of Sooty Shearwaters is unknown, but runs in the millions (Del Hoyo *et al.* 1992).

The distribution maps show that Sooty Shearwaters are most abundant in the southern North Sea from August to November, with stragglers seen in all other periods, except February/March. Most Sooty Shearwaters are seen in the northwest of the study area (Figure 4.20).

No wind farm impact studies have been conducted in areas with significant numbers of Sooty Shearwaters. Walls *et al.* (2013) found much lower numbers of the related Manx Shearwaters post-construction in the general area of the Robin Rigg offshore wind farm, West Scotland, and none within the wind farm parameter, post construction. However, numbers observed were insufficient to conduct a full pre/post analysis.

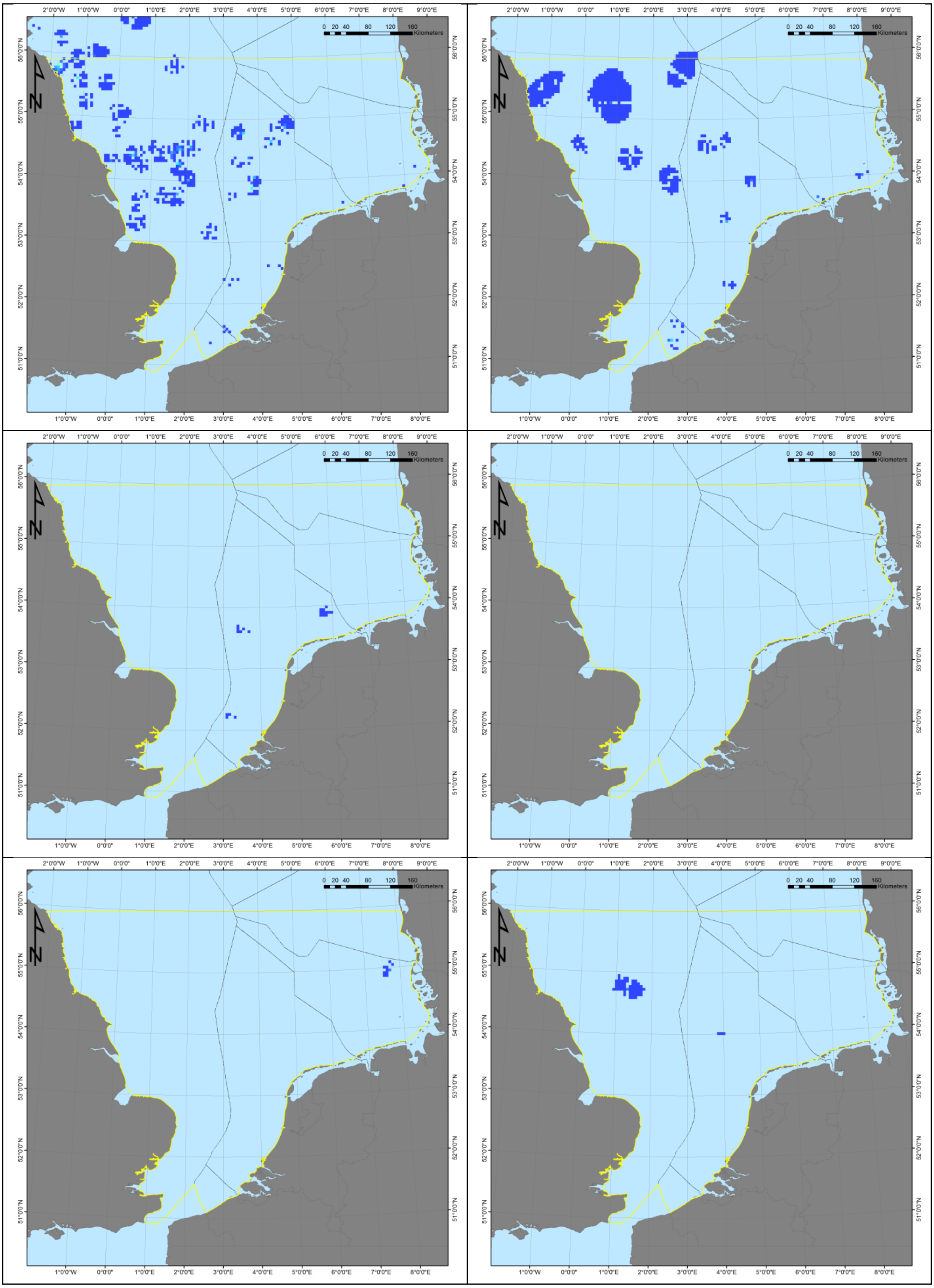


*Figure 4.20. Sooty Shearwater: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Manx Shearwater *Puffinus puffinus*

Manx Shearwaters breed almost exclusively scattered around Ireland and along the UK west coast. Important feeding grounds are located in the Irish Sea (Dean *et al.* 2012), the north and west of Scotland and west and south of the British Isles, down into the Bay of Biscay (Brooke 1990; Stone *et al.* 1995; Guilford *et al.* 2008, 2009; Mackey *et al.* 2004). These birds only spend a short breeding season in the British Isles and have a long migration route towards their wintering grounds off Brasil and Patagonia, Argentina, which leads them away from breeding latitudes from September to April (Guilford *et al.* 2009). Interestingly, given their western orientation as seen from the North Sea, some birds fly around northern Scotland in summer (May-October) and visit a rather narrow band of nearshore waters off the Scottish and English east coast (Stone *et al.* 1995). South of the River Humber and further offshore in the southern North Sea the species is rare (Camphuysen 1995a). What, exactly, they seek in these parts remains unknown, none of the birds that have been equipped with GPS loggers visited these parts. Even though some birds move out of the North Sea via its southern exit, and do so by passing close by the Dutch coastline (Camphuysen & Van Dijk 1983; Platteeuw *et al.* 1994), there is no clear offshore connection between the birds off the UK east coast and Europe’s mainland coastline.

The size of the population in the British Isles is circa 300 000 breeding pairs (Mitchell *et al.* 2004). Given the very low densities and restricted range of the species in the North Sea (Figure 4.21), total numbers here must be insignificant when compared to the total population.

No wind farm impact studies have been conducted in areas with significant numbers of Manx Shearwaters. The study of Walls *et al.* (2013) off west Scotland is the only one in which results on this species are reported. Much lower numbers of Manx Shearwaters visited the general study area after the Robin Rigg offshore wind farm became operational. No birds were found within the wind farm parameter, post construction. However, numbers observed were insufficient to conduct a full pre/post analysis.

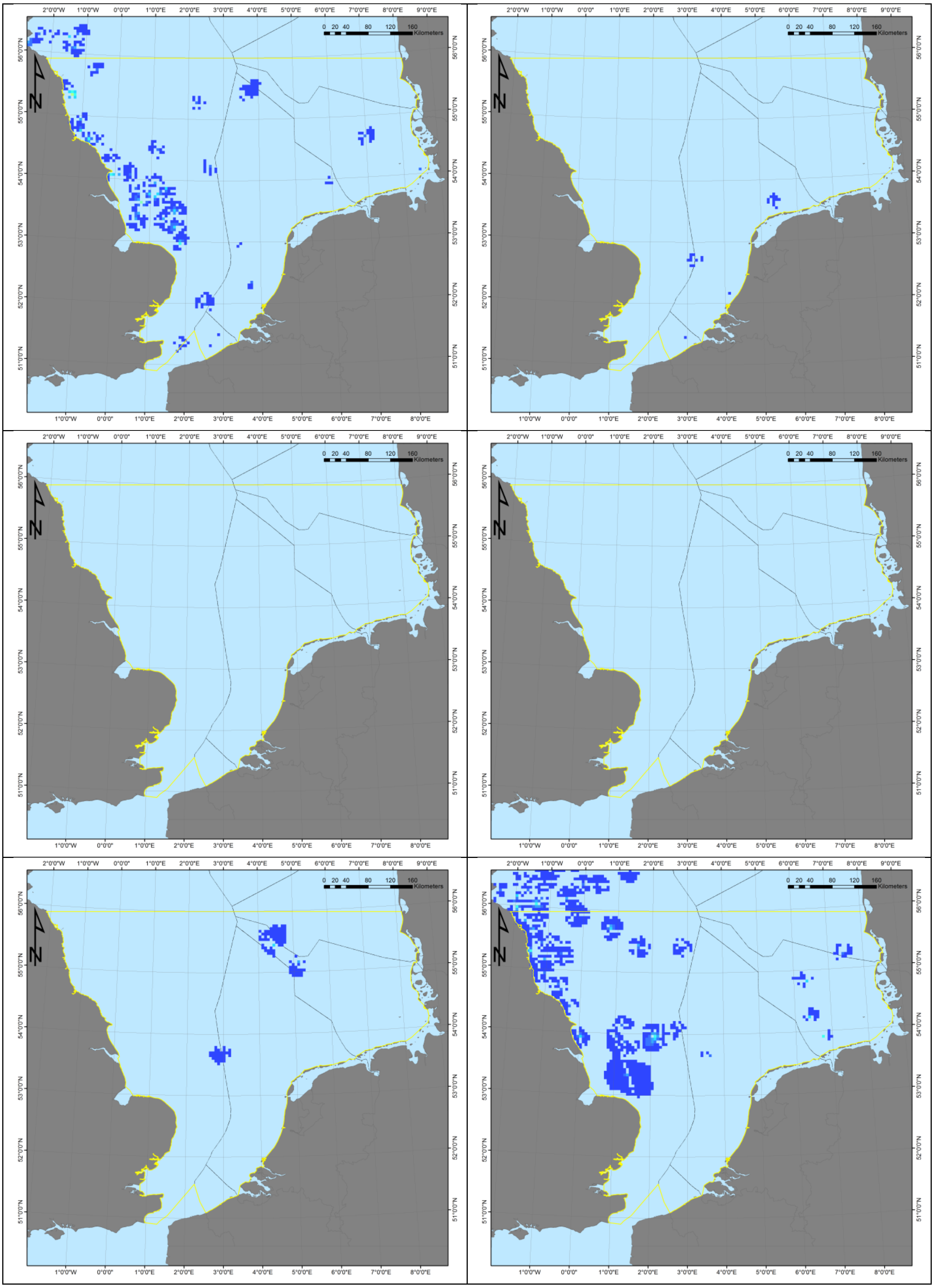


*Figure 4.21. Manx Shearwater: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Northern Gannet *Morus bassanus*

Only two gannet breeding colonies are found within the southern North Sea, Bepton Cliffs in England (ca 2500 pairs; Mitchell *et al.* 2004) and Heligoland (ca 200 pairs). The largest colony in the North Sea (and since 2014: the world) is just north of the study area, Bass Rock (from which the scientific name of the species is derived), currently with some 75 000 pairs (Murray *et al.* 2014). Gannets are strong flyers that range far from their breeding colonies. Therefore, also birds breeding north of the study area, particularly in NE Scotland, may visit the southern North Sea, while migrants from colonies further north may also migrate through the North Sea in autumn and spring, or winter here.

Birds from Bempton Cliffs easily fly to forage >100 km offshore (Langston & Boggio 2011), as do birds from Bass Rock (Hamer *et al.* 2000, 2007) and birds from either colony may even reach the Dutch sector of the North Sea on feeding trips, as may also do birds from Normandy, France (McClellan *et al.* 2014). Most adult gannets that use the southern North Sea in the breeding season are probably tied to the two regional colonies (Wakefield *et al.* 2013), but at other times of the year tens of thousands of gannets from colonies outside the southern North Sea use this area (Camphuysen *et al.* 1995a). Immature birds are probably only loosely connected to breeding colonies, and may range freely over the southern North Sea in all seasons. Northern Gannets catch fish by plunge diving, both unaided and using fish brought to the surface by fish-driving dolphins (Camphuysen *et al.* 1995a) or aided by fishing vessels (Camphuysen & Garthe 2000; Bodey *et al.* 2014).

The European breeding population is estimated at 418 250 birds (BirdLife International 2004), most of which may pass through the southern North Sea at some time in their lives. Northern Gannets occur widely spread throughout the southern North Sea (Figure 4.22). Concentrations are found near the breeding colonies, and intermittedly, at various places further offshore, when birds respond to tempory available rich feeding opportunities. A rather large area in the German Bight is avoided, to some extent, i.e., has relatively low densities throughout the year.

Northern Gannets appear to avoid flying through offshore wind farms to a great extent, and probably will not forage between wind turbines (Krijgsveld *et al.* 2011; Leopold *et al.* 2013a; Vanermen *et al.* 2013; Krijgsveld 2014; Mendel *et al.* 2014). This would imply that the total foot print of offshore wind farms can be seen as loss of feeding habitat for this species. Even worse, large wind farms 'near' major colonies may block the path to distant feeding grounds. For instance, Murray *et al.* (2014) speculate: "*The recent consent for the construction of four windfarms off the Fife and Angus coasts could ... pose a threat to Bass Rock's Northern Gannets since these could effectively block two-thirds of their access to the open sea*". Consequently, however, this would also mean that the risk of collisions in this species would be minimal.

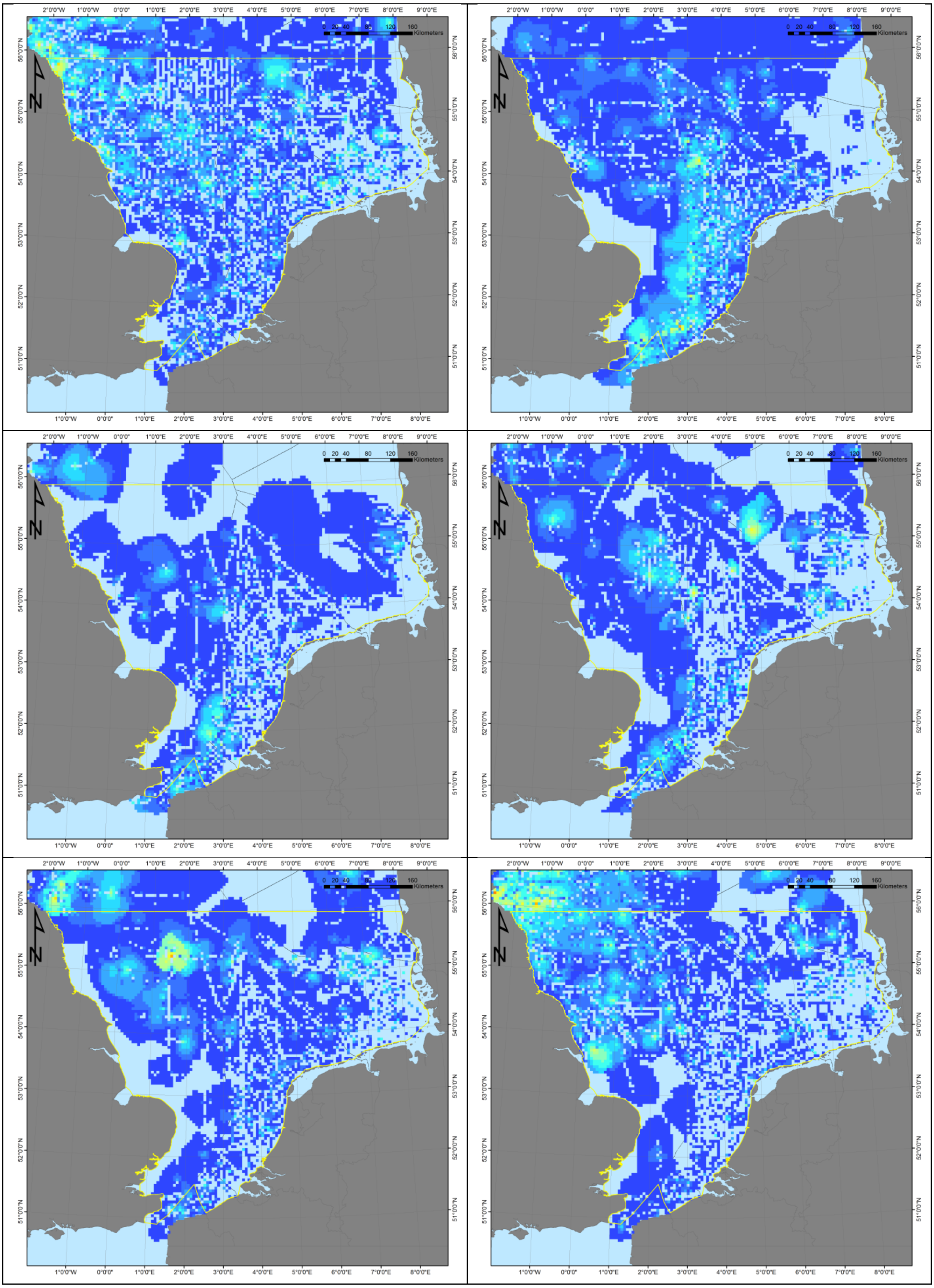


*Figure 4.22. Northern Gannet: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Great Cormorant *Phalacrocorax carbo*

Two subspecies of Great Cormorants live around the southern North Sea: the subspecies *sinensis* breeds in marshlands in the low countries and the subspecies carbo breeds on cliff coasts in the UK and France (Van Eerden *et al.* 1995). This classic distinction between the subspecies has gradually lifted, as numbers increased due to better protection and eutrophication, enhancing feeding possibilities. In the UK, more and more inland colonies have been established, fuelled partly by immigration of *sinensis* cormorants (Mitchell *et al.* 2004), while in The Netherlands, *carbo*-subspecies cormorants are also increasingly found (http://www.dutchbirding.nl/news.php?id=12). Both subspecies are efficient generalist foragers, taking a wide variety of fish species and sizes, and dietary overlap between the two subspecies is large (Fonteneau *et al.* 2009), suggesting that they can exploit similar habitats. During at-sea surveys, the two subspecies are often not distinguished and are further treated together. Great Cormorants are quick to exploit new feeding opportunities, such as using discards from beam trawlers (Camphuysen 1999) or using offshore wind farms as a platform for feeding and at-sea wing-drying and socialising (Leopold *et al.* 2011, 2013).

The joint breeding populations of the UK, The Netherlands, Germany, Denmark, Norway & Sweden number some 207 128 pairs, or 517 821 individuals (including non-breeders; BirdLife International 2004). Their at-sea distribution is mostly coastal, year round, but birds *en route* crossing the North Sea have been noted on many occasions (Figure 4.23), and the species has become numerous in the existing offshore wind farms OWEZ and PAWP off the Dutch mainland coast. From this last development, it may be expected that future wind farms could also be colonised, and that these birds will use each new offshore wind farm as a stepping stone to reach the next one. Given that Great Cormorants (of the *sinensis* subspecies) are tree-nesters that are used to navigate between branches, and that they show no fear of offshore wind farms, or rather, are attracted to them, displacement from wind farm site is no issue for this species. On the contrary, new feeding grounds have been opening up for cormorants, that can be exploited from offshore wind farms.

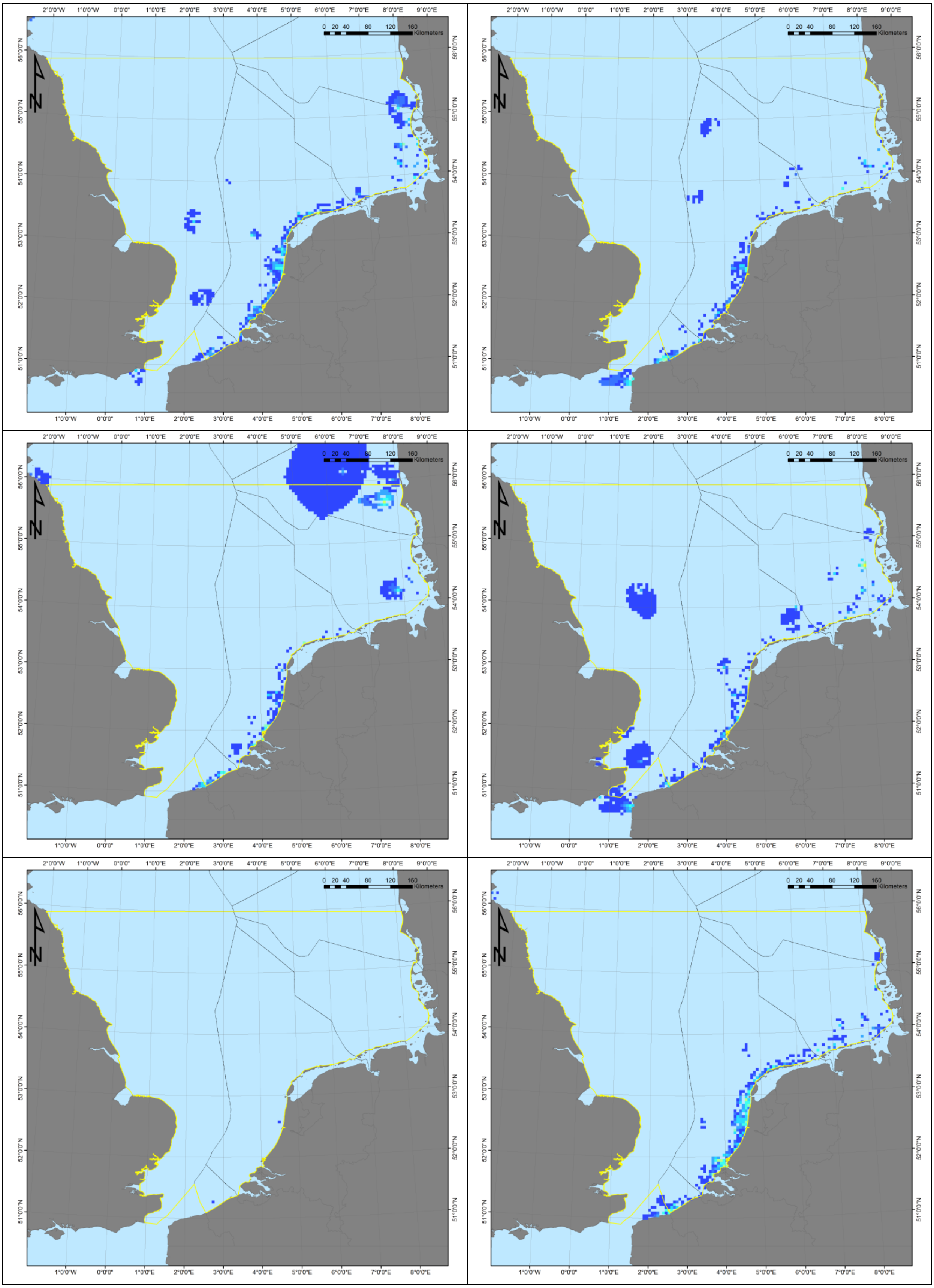


*Figure 4.23. Great Cormorant: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Common Eider *Somateria mollissima*

Common Eiders are coastal seabirds, or rather birds of coastal bays and estuaries like the Wash, and of shallow marginal seas like the Baltic and the Wadden Sea. Eiders are migratory to some extent, and breeding populations, relevant to the southern North Sea are found in the British Isles, Denmark, Germany, The Netherlands, southern Norway and Russia. Here, some 532 853 breeding pairs (1 332 133 birds) are found (Wetlands International 2014). Most of these (1 048 000 birds, including birds from Fenno-Scandinavia) winter in the Baltic; an estimated 463 000 birds winter in the North Sea (Skov *et al.* 2007). Most "North Sea" birds remain in the Wadden Sea and in the Humber and Wash estuaries, outside the range covered in this report. Food shortage in the Wadden Sea, or very rich pickings in the coastal North Sea may make Eiders move in large numbers to the latter (Leopold 1993; Camphuysen *et al.* 2002)

There is exchange between Common Eiders breeding in the UK, Norway and the Wadden Sea (Baillie & Milne 1989; Wernham *et al.* 2002). Even though the distribution of Common Eiders in the North Sea is largely coastal, some migrants have been noted crossing the North Sea (Figure 4.24). In general, however, Common Eiders are unlikely to encounter many offshore wind farms in the North Sea. Migrating Common Eiders are known to avoid flying through offshore wind farms (in the Baltic) and to be reluctant to land near turbines (Tulp *et al.* 1999; Petersen *et al.* 2006a; Larsen & Guillemette 2007; Masden *et al.* 2009). On the other hand, Common Eiders may, in future, learn to exploit benthos, such as Blue Mussels *Mytilus edulis*, growing on the base of wind turbines.

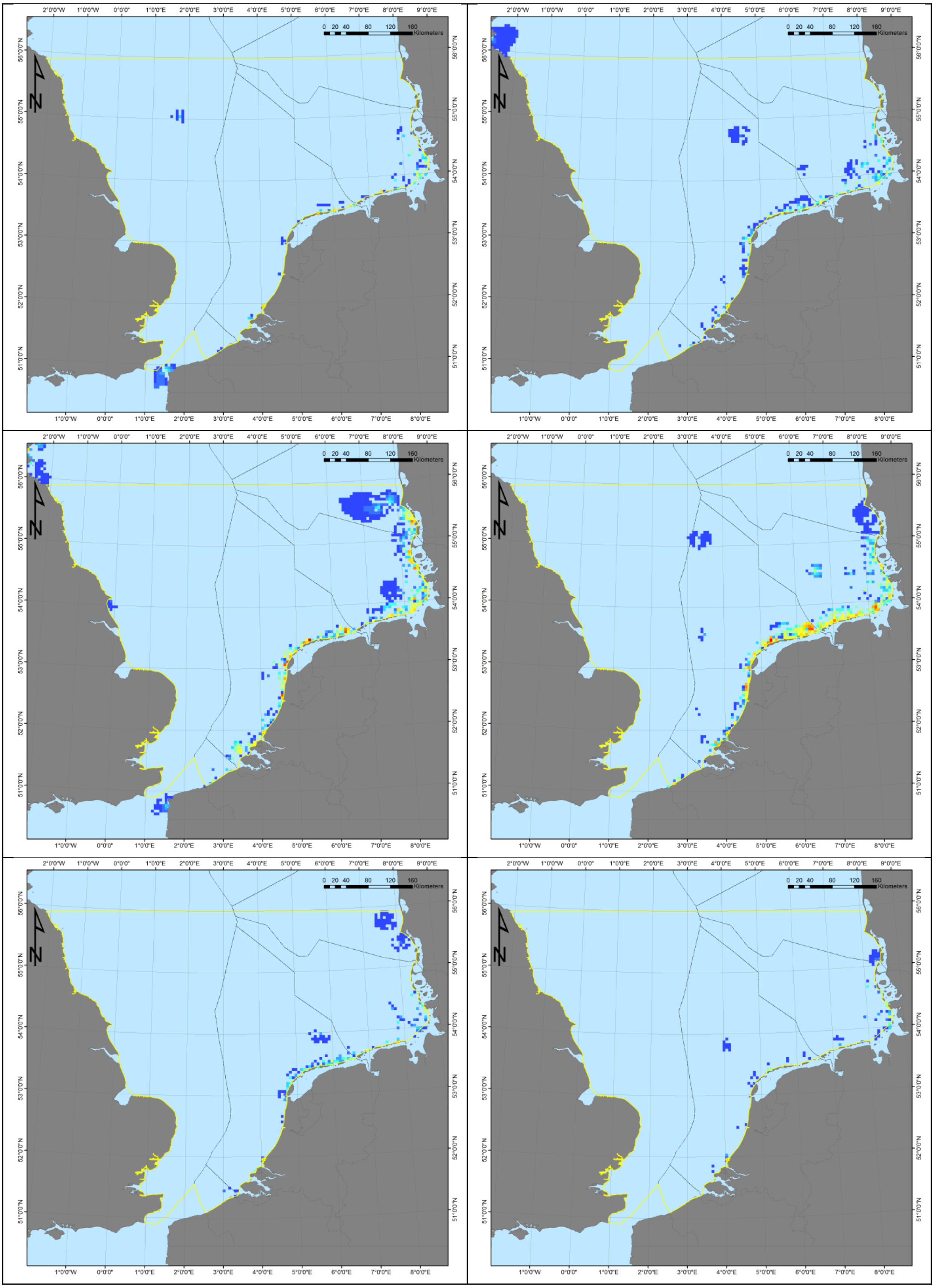


*Figure 4.24. Common Eider: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Common Scoter *Melanitta nigra*

Common Scoters form large wintering flocks in coastal waters, that are rich in suitable food (shellfish) and relatively undisturbed. Flocks of 100 000 and more have been counted in Dutch waters (Leopold *et al.* 1995); similar numbers may be found in Germany (Garthe *et al.* 2007) and Denmark (Petersen *et al.* 2006b). Numbers along the English east coast are much smaller, adding up to only a few thousand birds (Dean *et al.* 2003; WWT Consulting 2009).

Common Scoters have their feeding habitat in shallow nearshore waters (Figure 4.25), but there is regular migration between mainland Europe and the UK, resulting in flocks seen flying offshore (Offringa 1993). Also when migrating along the Dutch mainland coast, these ducks do not shy away from slightly more offshore waters if this takes them to where they want to go quicker (Platteeuw 1990). Offshore migrants have also been spotted in the German Bight, both in spring (April/May) and in autumn (October/November): see distribution maps for this species. Even though wintering Common Scoters, on their nearshore feeding grounds, are unlikely to encounter offshore wind farms, migrating flocks might.

Birds seen flying across the North Sea to the UK in autumn, have been noted to avoid flying through the Dutch offshore wind farms OWEZ and PAWP (Krijgsveld *et al.* 2011; Leopold *et al.* 2011, 2013). Similar avoidance was noted regarding the wind farms Nysted and Robin Rigg, outside the North Sea (Petersen *et al.* 2006a; Walls *et al.* 2013). At Horns Rev 1, strong avoidance was observed in the first years of operation, but in one subsequent year the scoters did enter this wind farm in large numbers, when food availbility was particularly high within wind farm limits (Petersen *et al.* 2006a, Petersen & Fox 2007).

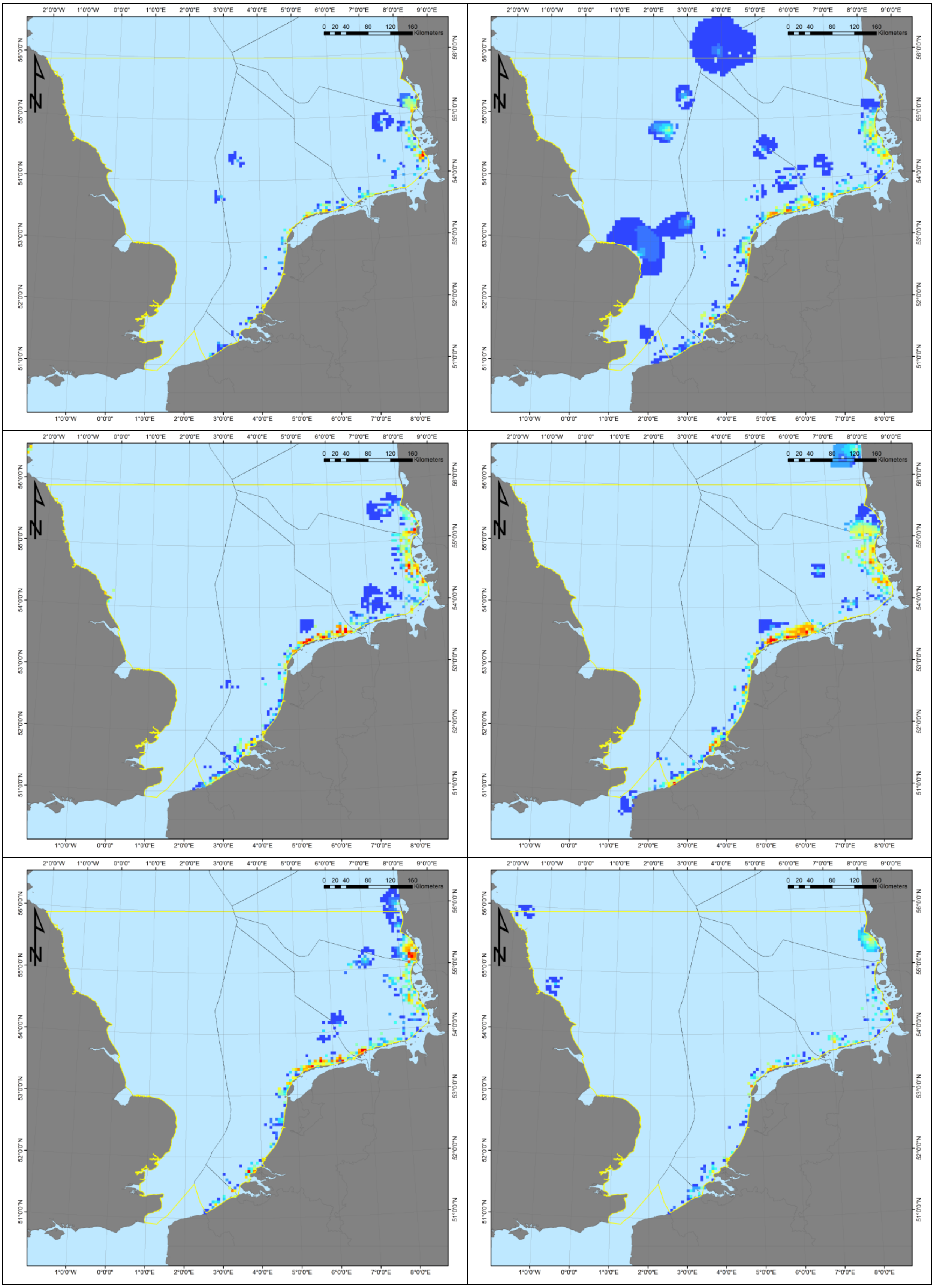


*Figure 4.25. Common Scoter: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Velvet Scoter *Melanitta fusca*

Velvet Scoters are much less common in the North Sea than Common Scoters, with an estimated wintering population of 121 000, versus 570 000 Common Scoters (Skov *et al.* 2007). Most Velvet Scoters winter in the Baltic, and are less inclined to cross Jutland to reach the North Sea, as are Common Scoters. Many that do cross over, remain in Danish and German waters, but in some years considerable numbers move on further southwest (Leopold 1993). Velvet Scoters tend to join (the much larger) wintering flocks of Common Scoters in the eastern North Sea (Figures 4.25 & 4.26), making them often difficult to count accurately. Given the very similar behaviour in winter, much that applies to the better-known Common Scoters, probably also applies to the Velvets. For this species, however, the North Sea is a relatively less important wintering ground and any adverse effects of offshore wind farms are less likely to impact this species on the population level.

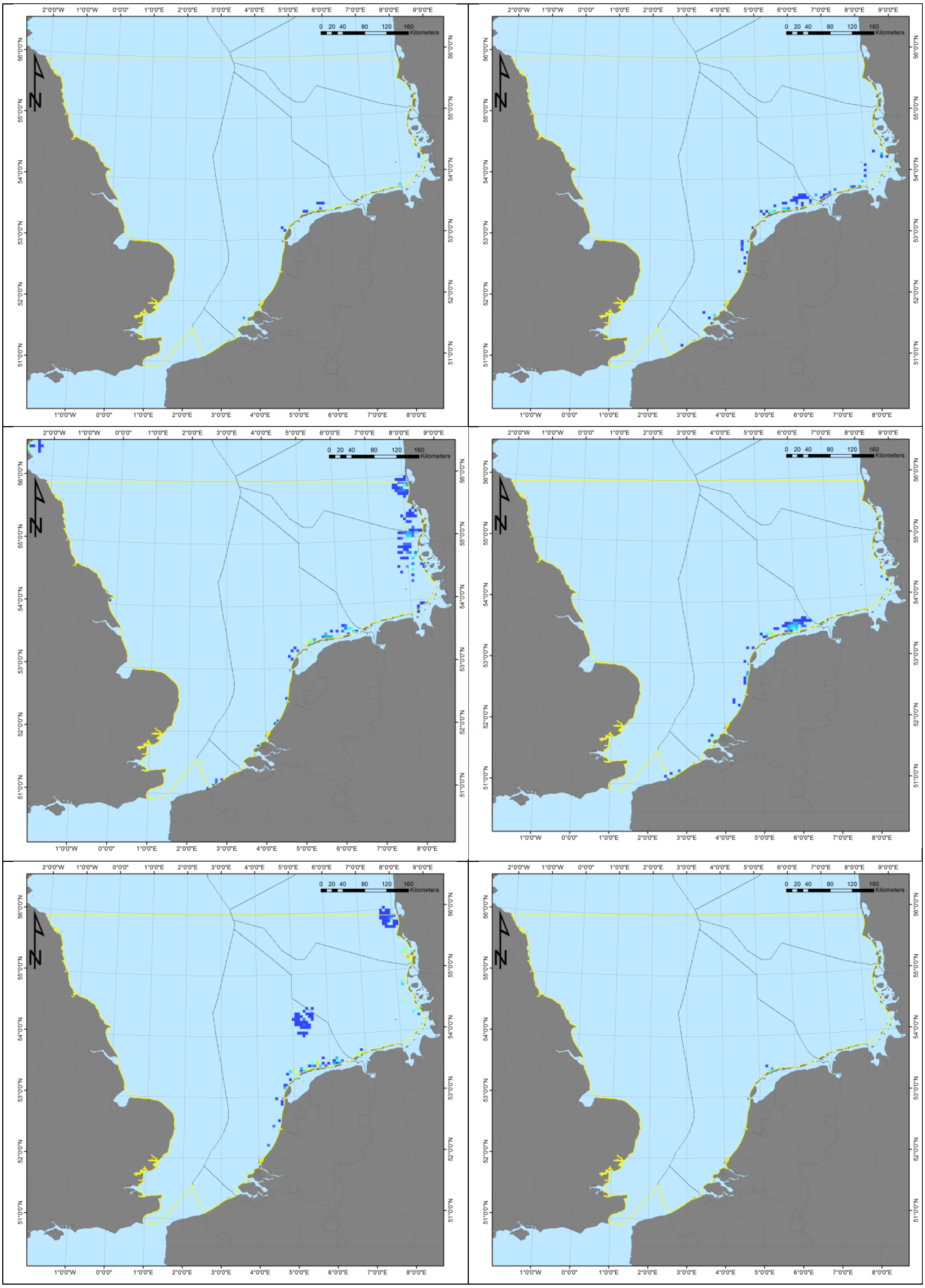


*Figure 4.26. Velvet Scoter: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Parasitic Jaeger *Stercorarius parasiticus*

Arctic Jaegers have a circumpolar, holarctic breeding range, with their southernmost colonies around the Great Lakes in North America, and in northern Scotland (particularly Orkney and Shetland) in Europe (Voous 1960; Mitchell *et al.* 2004). Numbers of breeding pairs in the latter area have fluctuated between 1039 pairs in 1969-70 and 3388 pairs in 1985-88 and are currently around 2000 pairs (Mitchell *et al.* 2004). Given the expanse of the breeding area, the spread of the species over this, and the remoteness of much of the breeding range, it is not surprising that breeding numbers in northern Europe are not accurately known. The European population is estimated at some 36 000 pairs (BirdLife International 2004) but could be half, or double this figure (Mitchell *et al.* 2004). Thousands of Parasitic Jaegers are seen yearly to migrate in autumn along the Dutch shoreline (Platteeuw *et al.* 1994), indicating that a large proportion of the entire European population migrates through the North Sea.

Their passage is mostly swift and their presence in the southern North Sea lasts for a rather short period of time. At sea, their distribution is very dispersed (Figure 4.27); concentrations of birds, or rather, higher densities are only found along the coastlines that the birds move along. No wind farm studies have provided estimates of impacts on this species, which, given their quick, dispersed passage over the open North Sea, and their very high aerial maneuvrability, is probably low.

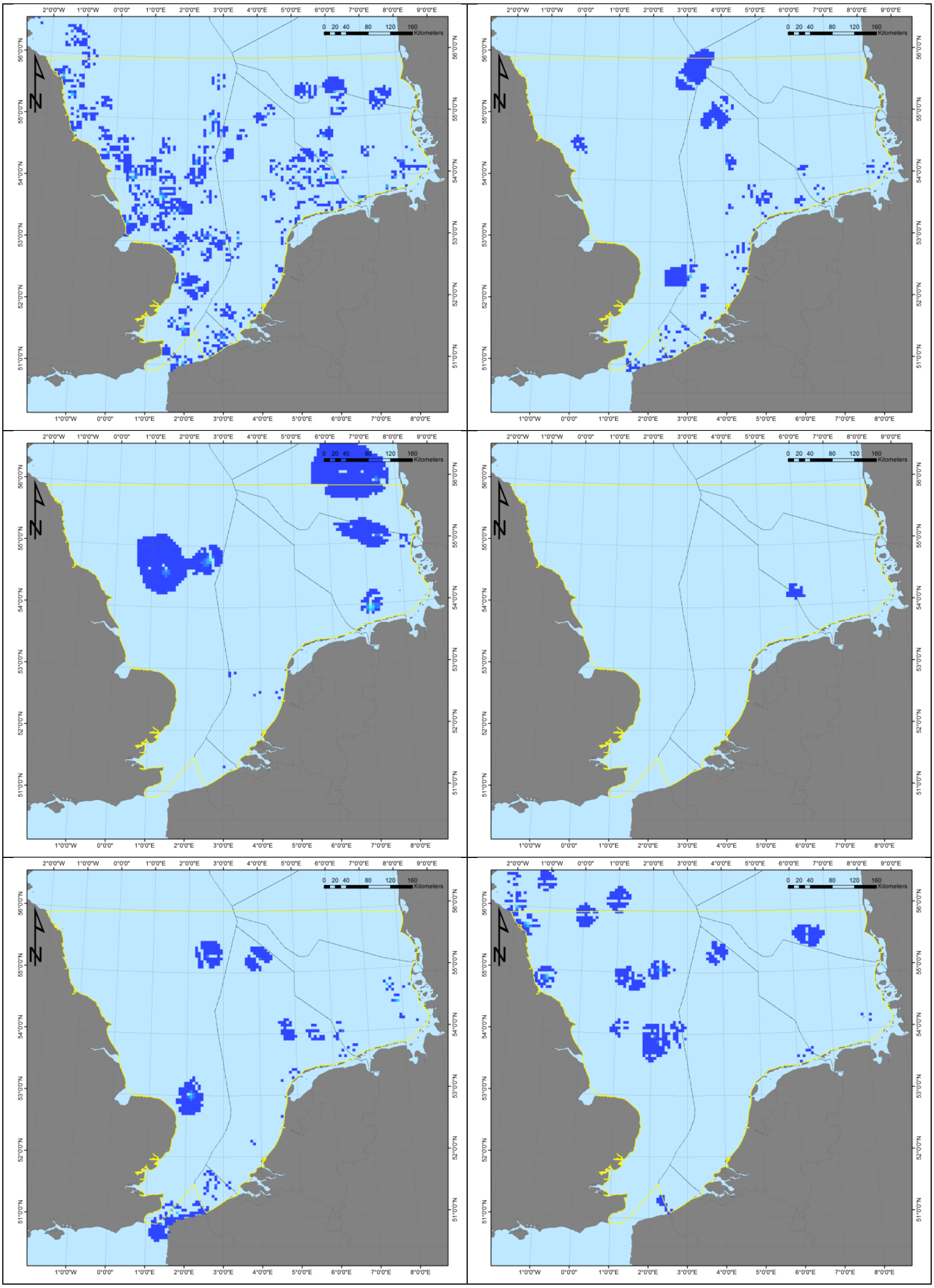


*Figure 4.27. Parasitic Jaeger: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Great Skua *Stercorarius skua*

Great Skuas have a very small (world) population, numbering no more than circa 16 000 pairs. Most of these breed on Orkney and Shetland, while on Iceland numbers have increased over the last century, resulting in range expansion towards the north (Barents Sea) and to the south, into Scotland (Furness 2002; Mitchell *et al.* 2004). The southern North Sea is an important migration route for Great Skuas (Furness 2002), as may be inferred from ring recoveries in The Netherlands (Leopold 2006; Figure 4.28).

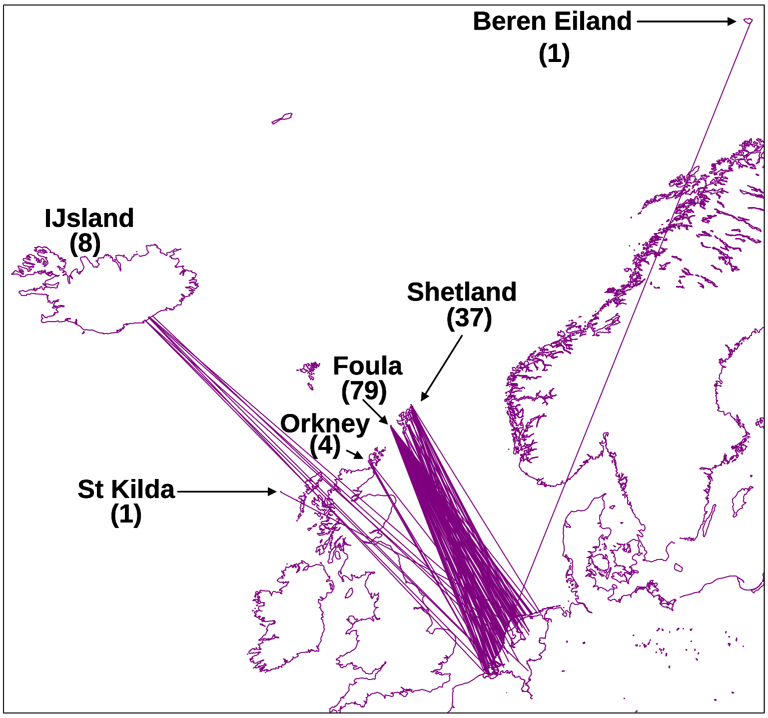


*Figure 4.28. Ringing recoveries of Great Skuas in The Netherlands, connected (by straight lines) to the locations where these birds were ringed (from: Leopold 2006).*

Migration, particularly in autumn, is leisurely. Great Skuas do not migrate over very large distances as most winter in the Biscay area (Furness 2002; Magnusdottir *et al.* 2012), and birds on migration may linger for some time in areas that provide good feeding. Remarkably, these birds combine migration with primary moult and do so for instance in the southern North Sea, where this has been documented in the Brown Ridge area (NL) by Van Bemmelen *et al.* (2012). Moult is slow, and starts right after the birds leave their breeding areas, late July or early August, and is completed in the wintering quarters, in January/February (Van Bemmelen *et al.* 2012). Loose autumn concentrations of Great Skuas, of total numbers of international importantance, or close to that, have been found in the Brown Ridge area (Van Bemmelen *et al.* 2012) and in the Frisian Front area (Geelhoed *et al.* 2013). In autumn, Great Skuas occur widely spread over the entire southern North Sea, excepting only the inner German Bight. Numbers in the study area are considerably lower in other seasons.

Given their small total population size and intensive usage of the southern North Sea during migration (Figure 4.29), the species is vulnerable to extra mortality. Direct mortality, from turbine strikes, could potentially be highly detrimental to this species. However, these birds are extremely maneuverable in the air, and should be well able to avoid strikes. Vanermen *et al*. (2013) offers the best data on wind farm vulnerability of this species, but must conclude that, to date, even the best data are still extremely unreliable, due to a very limited number of observations in offshore wind farms, so this issue needs to be resolved.

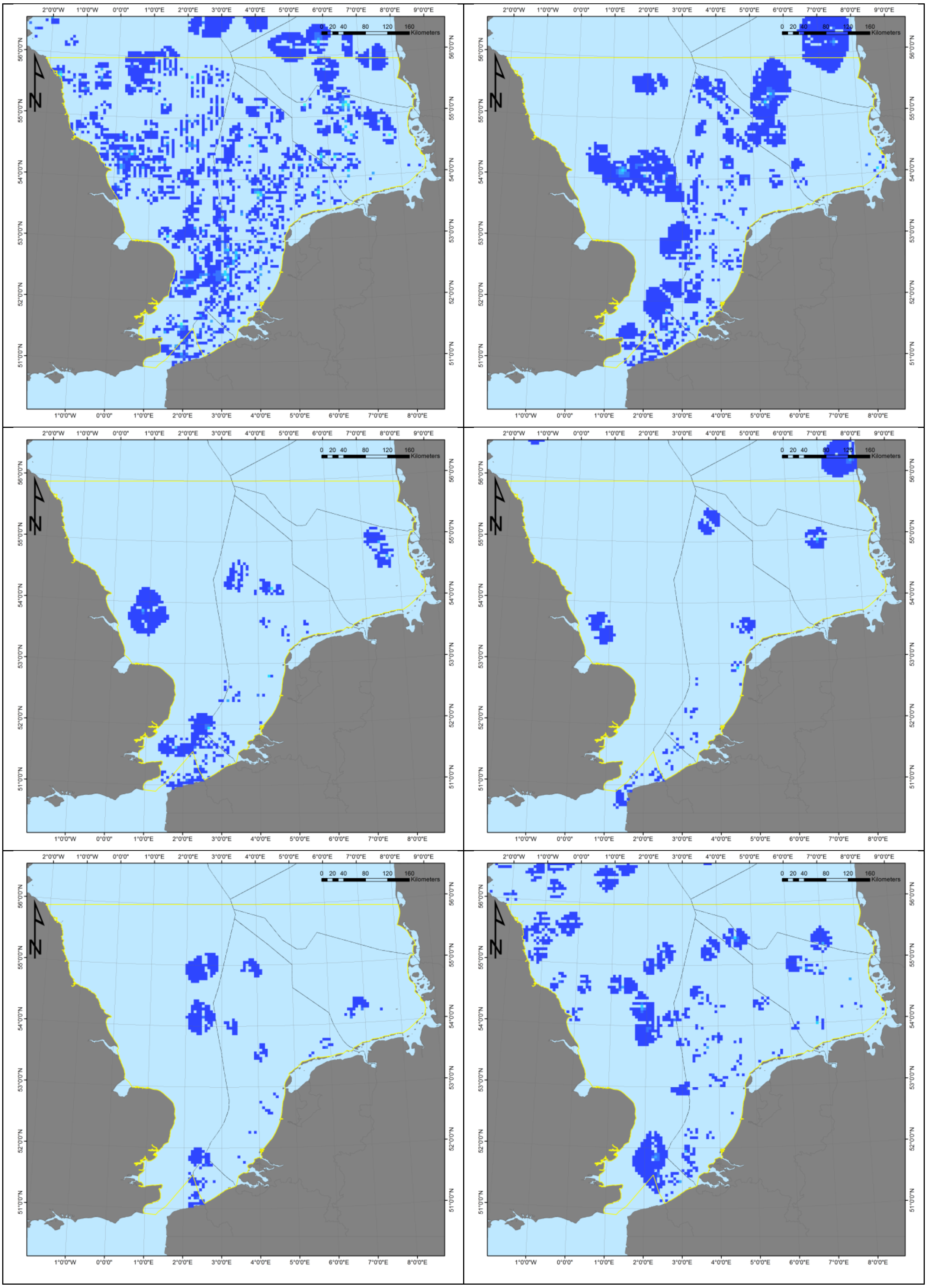


*Figure 4.29. Great Skua: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

Little Gull *Hydrocoloeus minutus*

Little Gulls are passage migrants and wintering birds in the Southern North Sea (Camphuysen & Leopold 1994; Garthe 1993). The species breeds in a large area around the Baltic Sea and Russia and most birds winter in the Mediterranean, off western Iberia and France and in nearshore waters in the North Sea and around the British Isles (Del Hoyo *et al.* 1996). The total European population size is estimated to comprise 27 729 breeding pairs (69 323 birds; Wetlands International 2014). Some 5400 birds winter in the North Sea (Skov *et al.* 2007), but much larger numbers of Little Gulls are passing through the area in autumn and spring, with a spectacular spring migration peak in April along the eastern North Sea board (Camphuysen & Van Dijk 1983; Platteeuw *et al.* 1994; Camphuysen 2009). Nearly the entire European population of Little Gulls may pass along the Low Countries in spring and thousands may stage in the eastern North Sea coastal waters for several weeks in April, if conditions are favourable (den Ouden & Stougie 1987, 1990; Keijl & Leopold 1997; Schwemmer & Garthe 2006; Garthe *et al.* 2007). The eastern, rather nearshore parts of the southern North Sea mostly show the highest densities within the study area (Figure 4.30).

Further out at sea, Little Gulls mostly fly low above the water and would thus be not very vulnerable to turbine strikes. However, during peak-migration, large flocks have also been seen passing over Dutch nearshore waters in large flocks, at rotor-height altitudes (M. Platteeuw in litt. Based on personal observations during seawatches in Noord-Holland, by M. Platteeuw, N. van der Ham, W. van Splunder). Data on displacement from operational offshore wind farms have yielded different results. Little Gulls did not seem reluctant to enter OWEZ (where the turbines are widely spaced), but were never seen within PAWP, with its much higher turbine density (Leopold *et al.* 2013a). Vanermen *et al*. (2013) found only weak evidence of displacement form offshore wind farms in Belgian waters: their results are suggestive of displacement, but not statistically significant. In Germany, at Alpha Ventus, no birds were seen within wind farm perimeters, despite rather high general post-construction densities of Little Gulls in the area (Mendel *et al.* 2014). At Horns Rev, Little Gulls appeared to show both displacement and attraction (at different times) regarding the wind farm, but numbers of observations were too low to be certain of either (Petersen *et al.* 2006a).

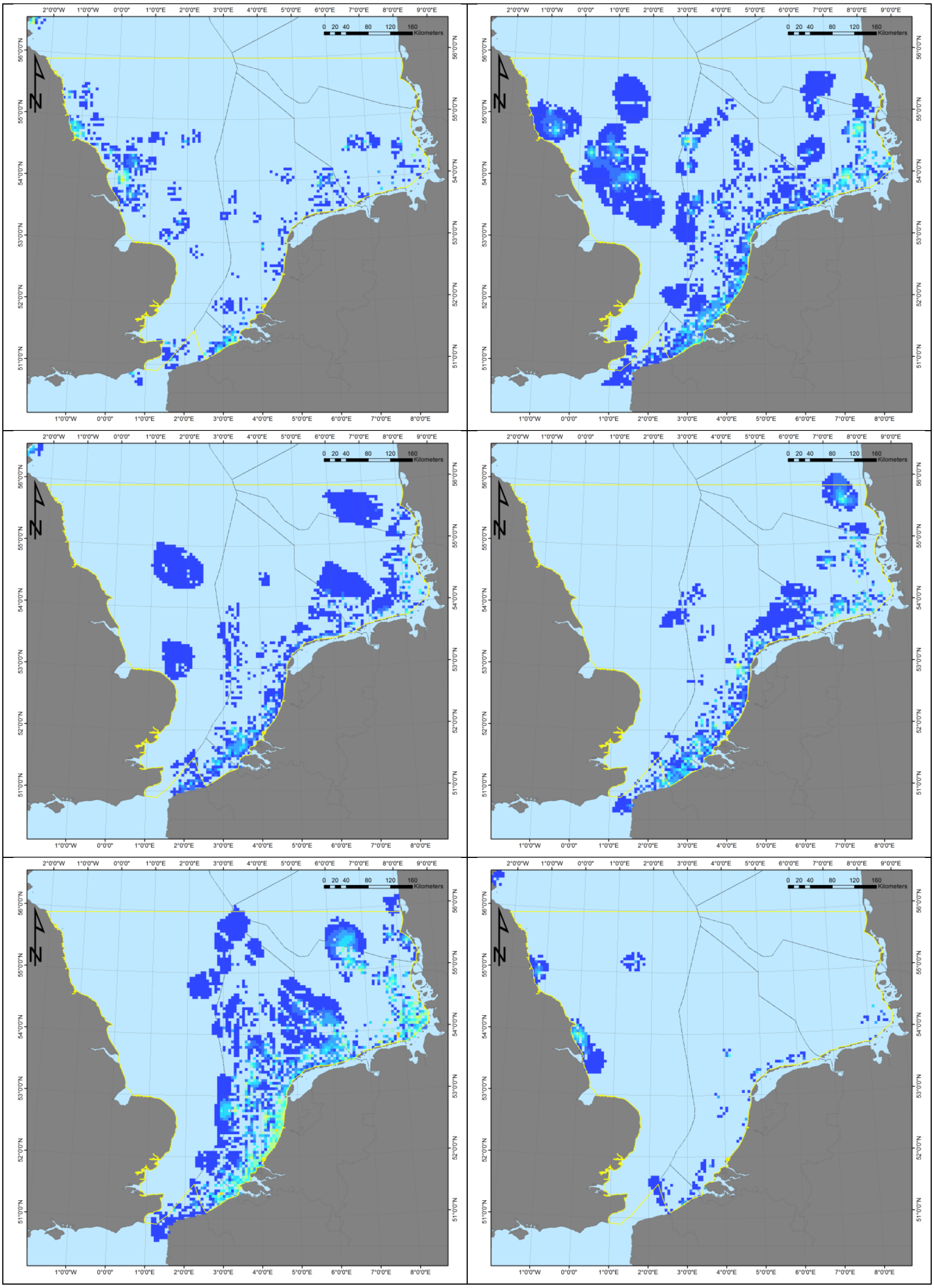


*Figure 4.30. Little Gull: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Black-headed Gull *Chroicocephalus ridibundus*

Black-headed Gulls breed in coastal and inland marshlands over a broad band across temperate Europe and Asia, with marginal numbers in Greenland and the eastern USA. Large numbers breed in countries around the North Sea alone (some 550 000 pairs; BirdLife International 2004; Mitchell *et al.* 2004) and ringing recoveries show that large numbers of birds from central and eastern Europe also visit the North Sea (Wernham *et al.* 2002). In winter, when much of central Europe becomes inhospitable to waterbirds, many Black-headed Gulls move west to the North Sea. Some cross over to the UK, but most turn southwest and follow the mainland coastline to more southerly wintering grounds (Figure 4.31). In the North Sea, most birds stick to nearshore waters. However, this species also shows complex moult migrations that involve crossings to the British Isles (Camphuysen & Leopold 1994) and dispersed flocks of migrants may be found offshore at any time of year.

Given that the bulk of Black-headed Gulls remain in nearshore waters, well landward of most projected wind farm sites, its large relevant population and its high reproductive capacity (three eggs per brood), this species does not seem to be very vulnerable to offshore wind developments in the North Sea. The first impact studies in nearshore wind farms off Europe’s mainland coasts generally yielded insufficient numbers of observations to further substantiate this, but the species tended to fly around, rather than through OWEZ (Krijgsveld *et al.* 2011, Krijgsveld 2014).

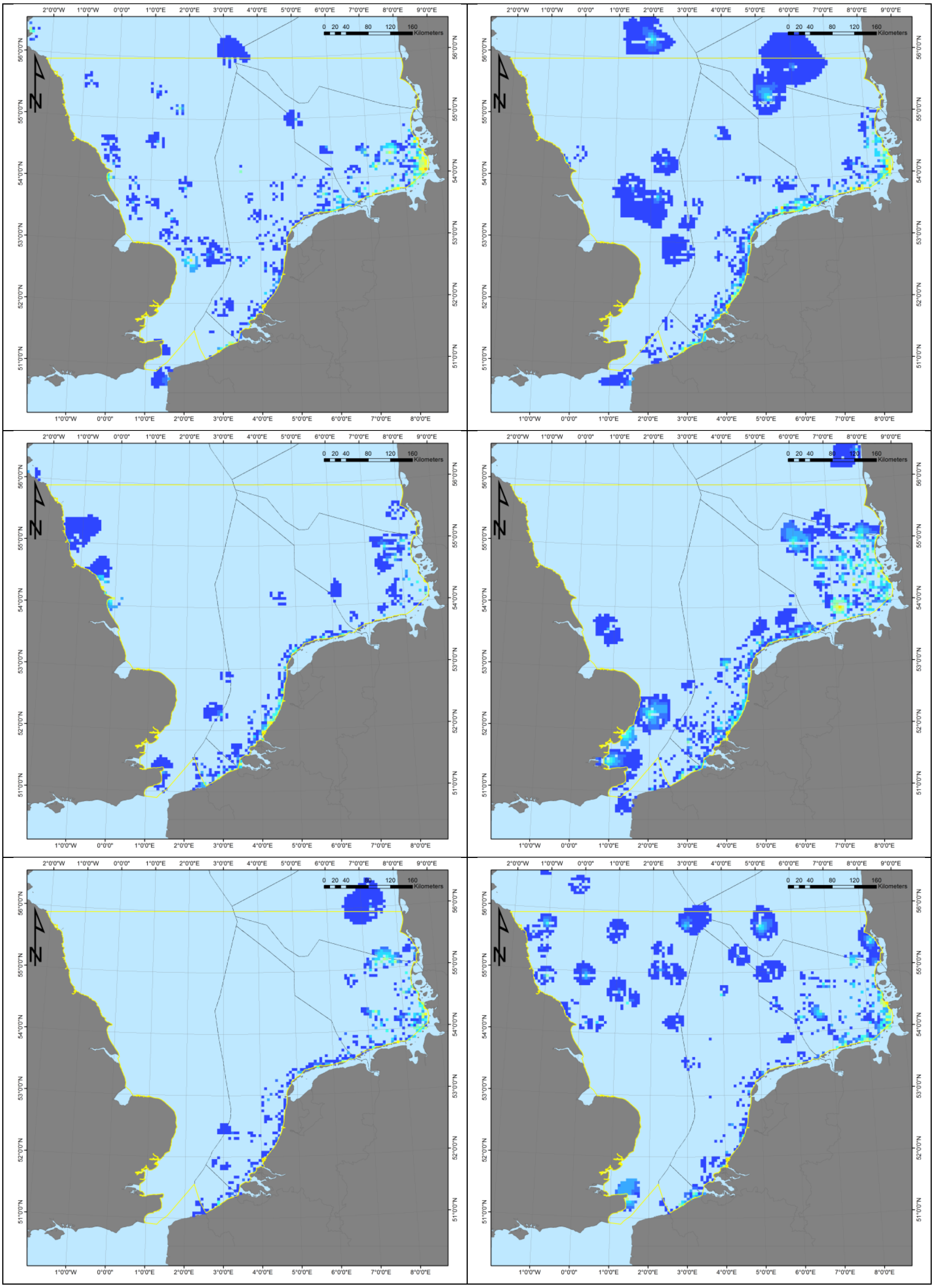


*Figure 4.31. Black-headed Gull: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

Mew Gull *Larus canus*

Mew Gulls breed breed across the Palearctic over a broad band across temperate Europe, Asia and NW America, both in coastal and in inland habitats. The birds winter in inland marsh and grasslands, along rivers and in estuaries and in coastal seas. Well over 200 000 pairs breed across countries around the North Sea (BirdLife International 2004; Mitchell *et al.* 2004), and in winter many birds from further east migrate to the study area (Wernham *et al.* 2002). Around 176 000 are estimated to winter in the North Sea (Skov *et al.* 2007). They may be found anywhere in the study area throughout the year, but the highest densities occur usually in a broad band along the coastlines, at either side of the southern North Sea (Figure 4.32).

Given the large relevant population and high reproductive capacity (three eggs per brood), this species seems little vulnerable to offshore wind developments in the North Sea. Results from impact studies in offshore wind farms vary, showing either avoidance of, indifference or attraction to wind farms (Krijgsveld 2014).

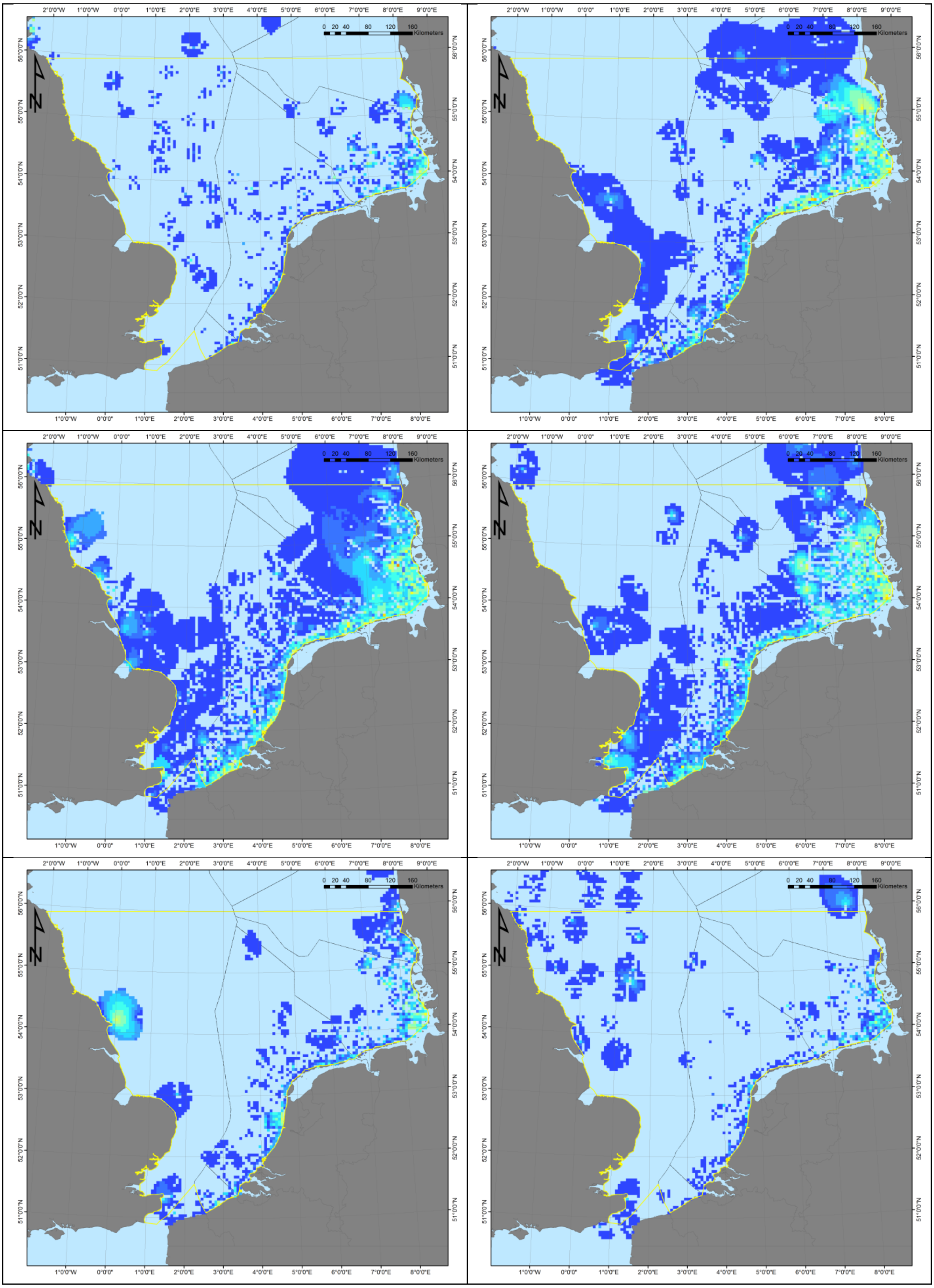


*Figure 4.32. Mew Gull: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Lesser Black-backed Gull *Larus fuscus*

Lesser Black-backed Gulls are sea-going birds, at least in the breeding season (Camphuysen 2013), that breed all along the North Sea coastlines, and further north, from Iceland to deep into northern Russia (where other (sub)species may be involved) and south of the North Sea, in France and Iberia (Del Hoyo *et al.* 1996). There has been much debate over subspecific status of various groups within the species, with UK birds ("subspecies" *graellsii*) having lighter mantle colouration than mainland European birds ("*intermedius*"), while birds in northern Norway are even darker (*fuscus*) and relatively longer-winged. At the time of writing, the consensus is that all three forms belong to the same species: *Larus fuscus* (http://www.bou.org.uk/british-list/bird-names/ for the UK, and http://www.dutchbirding.nl/content/page/files/webprog20141128-96.pdf for The Netherlands). Even so, the nominate form *Larus f. fuscus* is only a rare visitor to the North Sea (Van den Berg & Bosman 1999) and most birds living in the southern North Sea, or passing through, stem from the UK (67 500 pairs in England and Scotland), mainland NW Europe (Belgium, Netherlands, Germany, Denmark, Norway & Sweden: 91 000 pairs) and Faeroes (9 000) and Iceland (25 000; Mitchell *et al.* 2004).

Lesser Black-backed Gulls may be found in the southern North Sea throughout the year, but the highest densities occur from spring through autumn, and in a wide band along Europe's mainland coasts (Figure 4.33). Lesser Black-backed Gulls do not appear to fly around offshore wind farms and seem mostly indifferent to these (Leopold *et al.* 2013a; Krijgsveld 2014), though maybe they gain some altitude while flying through the wind farms (Camphuysen 2011). There is some difference between studies, with at the extremes, a tendency to displacement (Mendel *et al.* 2014) and to attraction (Vanermen *et al.* 2013). In any case, large numbers of dead Lesser Black-backed Gulls have not washed ashore, due east of the Dutch offshore wind farms OWEZ and PAWP, since these became operational (Dutch Seabird Group, unpublished results). Like in all gulls and other seabirds that flock around fishing vessels to feed on discards, the data show a great deal of noise. Very large concentrations of these gulls have been encountered on many occasions (c.f. Camphuysen 1995b). As fishing will generally not be allowed inside offshore wind farms, large, fisheries-related concentrations of these birds will become rare on-site. Concentrations encountered in the past, still greatly influence the dataset used in the current analyses and may lead, in places, to unrealistically high densities at projected offshore wind farm sites.

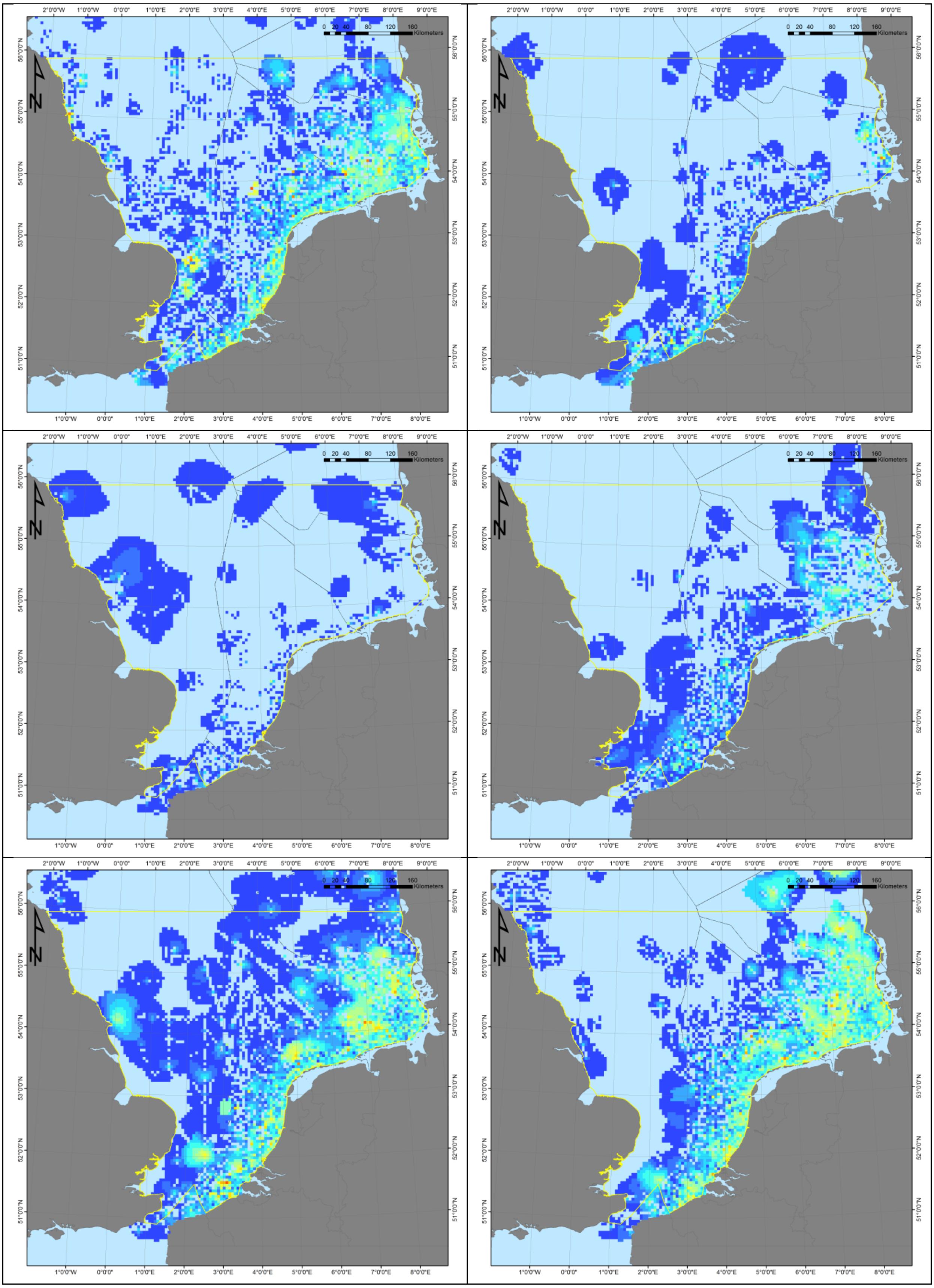


*Figure 4.33. Lesser Black-backed Gull: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### European Herring Gull *Larus argentatus*

European Herring Gulls are often considered "seagulls" but in the breeding time, North Sea Herring Gulls are much less sea-going than Lesser Black-backed Gulls (Camphuysen 1995b; Camphuysen *et al.* 2008, 2011). In fact, in the breeding season, Herring Gulls hardly take to the open North Sea and remain in coastal waters and estuaries. Also in August, during primary moult, Herring Gulls remain mostly nearshore. This picture changes dramatically in autumn, when local birds and birds from more northern regions that migrate into the North Sea disperse all over the southern North Sea (Camphuysen & Leopold 1994; Camphuysen 1995b; Stone *et al.* 1995), where they compete with other scavenging seabirds for fishery discards and offal (Camphuysen *et al.* 1995b). In offshore waters, Herring Gulls are thus mainly present in significant numbers in the non-breeding season (Figure 4.34): Skov *et al.* (2007) estimate that 918 000 European Herring Gulls winter in the North Sea. Many of these are northern birds, of the nominate subspecies *argentatus*, that breed in Denmark and Fenno-Scandia (Del Hoyo *et al.* 1996). Local, North Sea breeders (subspecies *argenteus*) and their offspring migrate (or rather disperse) over relatively short distances and many remain within the North Sea region as well (Camphuysen 2013).

An impact of offshore wind farms has been difficult to assess during impact studies (e.g. Leopold *et al.* 2013a; Vanermen *et al.* 2013). Like in other gulls, the data show a great deal of noise caused by fishing vessels attracting large numbers of birds from large distances. As fishing will generally not be allowed inside offshore wind farms, large, fisheries-related concentrations of these birds will become rare on-site. Statistical analysis may indicate dispersal in such situations, but Herring Gulls might still be attracted to offshore wind farms for other reasons, like resting on the various structures, or feeding at the base of these, when exposed by falling tides. Herring Gulls, like other gulls are know collision victims of turbines on land (Baptist 2005; Krijgsveld *et al*. 2009a,b; Verbeek *et al*. 2012), but collisions at sea have not yet been witnessed.

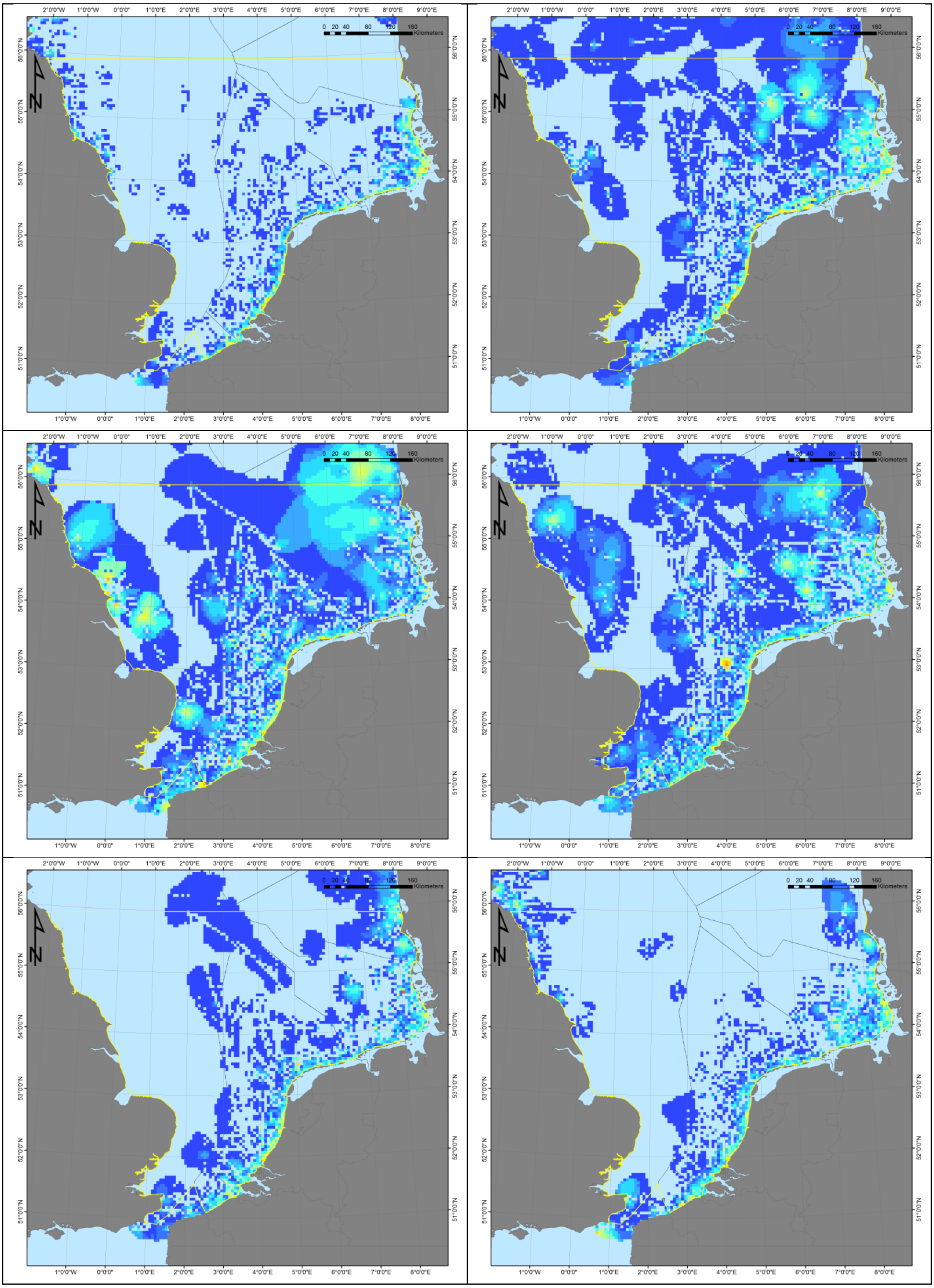


*Figure 4.34. European Herring Gull: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Great Black-backed Gull *Larus marinus*

Great Black-backed Gulls may be found offshore in the southern North Sea at any time of year, but during the breeding season only immatures remain, that have no direct connection to any breeding colony. The species mainly breeds on rocky shores, often in the vicinity of other seabirds, that it might take as food. Recently, this species has colonised The Netherlands (with only a few mock-rocky coastal structures) and breeds here now in low numbers (Van den Berg & Bosman 1999). Across the North Sea, the species is also rare along the English east coast (Mitchell *et al.* 2004) and most birds in the southern North Sea must stem from the UK western and northern coasts (the UK and Ireland have nearly 20 000 breeding pairs), or from Iceland (2300 pairs), Faroes (1200), Denmark (1500), Fenno-Scandia (55 000 pairs; Mitchell *et al.* 2004; Wernham *et al.* 2002). Skov *et al.* (2007) estimates that some 300 000 Great Black-backed Gulls winter in the North Sea.

Great Black-backed Gulls disperse over the entire southern North Sea (Camphuysen & Leopold 1994; Stone *et al.* 1995) and are strong competitors behind fishing vessels but their numbers were often lower than those of other species in the associated flocks (Camphuysen *et al.* 1995b). Great Black-backed Gulls tend to be slightly more numerous in nearshore waters, but concentrations also occurred in different parts of the study area at times (Figure 4.35). Great Black-backed Gulls have been noted to rest in offshore wind farms (attaction), but this local increase in numbers is probably offset by fishing boats being expelled from offshore wind farms, taking their tails of gulls with them (displacement). Impact studies have found no significant effects of offshore wind farms on the numbers of Great Black-backed Gulls on site (Leopold *et al.* 2013a; Vanermen *et al.* 2013).

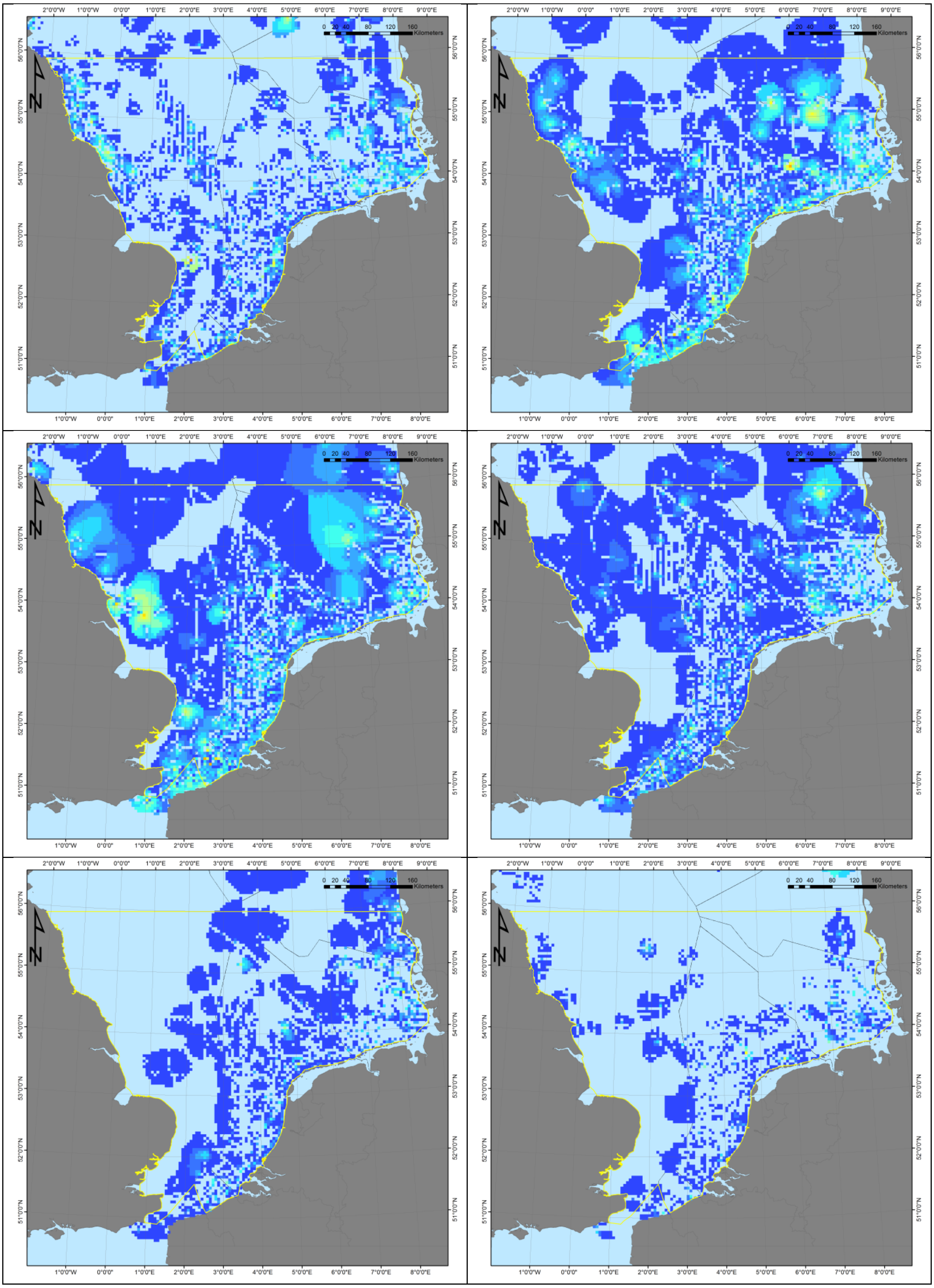


*Figure 4.35. Great Black-backed Gull: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Black-legged Kittiwake *Rissa tridactyla*

Black-legged Kittiwakes have a year-round presence in the southern North Sea, but in the breeding season, densities are rather low in the southern parts of the study area (Figure 4.36). Breeding colonies are found along the English east coast, at Heligoland, Germany, and on several offshore gas platforms on the Dutch and English Continental shelves. The largest numbers are found in England, at Bempton Cliffs and Flamborough Head, were 42 659 pairs were counted during the 1998-2002 'Seabird 2000' sensus. However, numbers had decreased sharply here, from 85 000 pairs in 1985-88 (Mitchell *et al.* 2004). Numbers at Helgoland are now stable, at some 7 000-9 000 pairs after a steady increase from the 1960s (Markones *et al.* 2009); numbers breeding on offshore platforms in the southern North Sea are poorly known, but probably around 100 pairs (Camphuysen & De Vreeze 2005, Camphuysen & Leopold 2007; Geelhoed *et al.* 2011). The species has a large total population that may be estimated at 6-8 million pairs, breeding on cliffs (and surrogate cliffs) from temperate to arctic habitats around the North Pole. Large numbers (1 034 000) winter in the North Sea, where Kittiwakes are among the most numerous birds (Skov *et al.* 2007).

Kittiwakes show little fear of offshore wind farms: impact studies have shown no declines in densities within wind farm parameters (Leopold *et al.* 2013a; Vanermen *et al.* 2013; Walls *et al.* 2013).

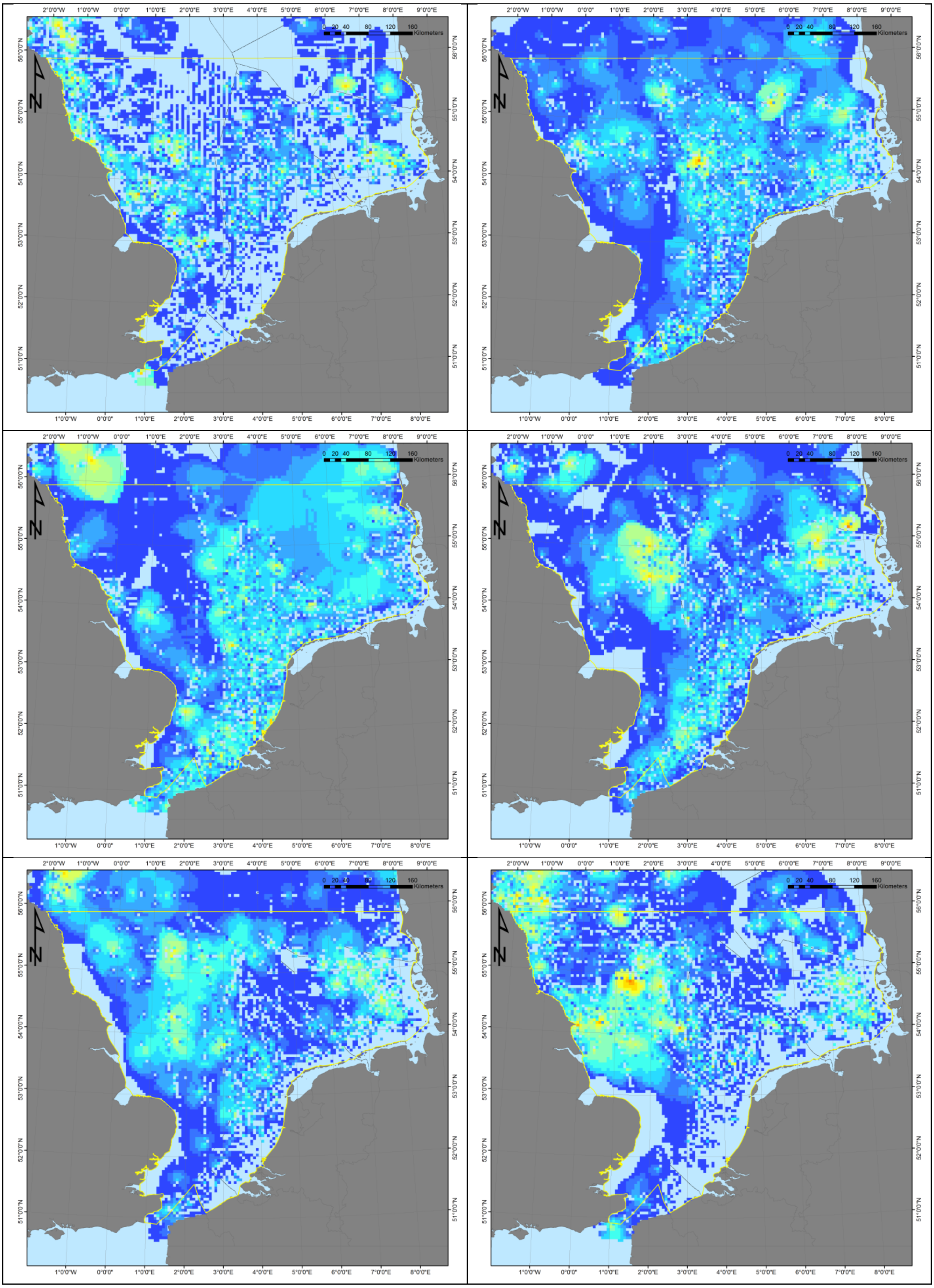


*Figure 4.36. Black-legged Kittiwake: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Sandwich Tern *Thalasseus sandvicensis*

Sandwich Terns form a superspecies with several closely related species. *Thalasseus s. sandvicensis* breeds around the North Sea and the British Isles, around the Baltic, along the NW Mediterranean, the Black and Caspian Seas, and winters in coastal waters off western Africa, in the Mediterranean and from the Red Sea to NW India and Sri Lanka (Del Hoyo *et al.* 1996). Birds around the North Sea form a metapopulation, with frequent exchanges between colonies across the region (Stienen 2006; Fijn *et al.* 2011, 2014). UK, Scandinavian, Danish, Dutch, Belgian and French colonies together number some 45 000 pairs (Mitchell *et al.* 2004).

Sandwich Terns have long been considered as rather coastal birds in the North Sea (e.g. Camphuysen & Leopold 1994), but have recently been found to venture also quite far offshore (Poot *et al.* 2011b; Figure 4.37) and are thus likely to interact with (future) offshore wind farms. No significant effect on Sandwich Tern densities was found for the two operational Dutch offshore wind farm (Leopold *et al.* 2013a). This is in contrast to work in the offshore wind farm Horns Rev (Denmark) and in Belgium where terns supposedly flocked around the outer turbines, to feed in the tidal wakes behind the monopiles (Elsam Engineering & Energi 2005; Elsam Engineering 2005; Petersen & Fox 2007; Vanermen *et al.* 2013).

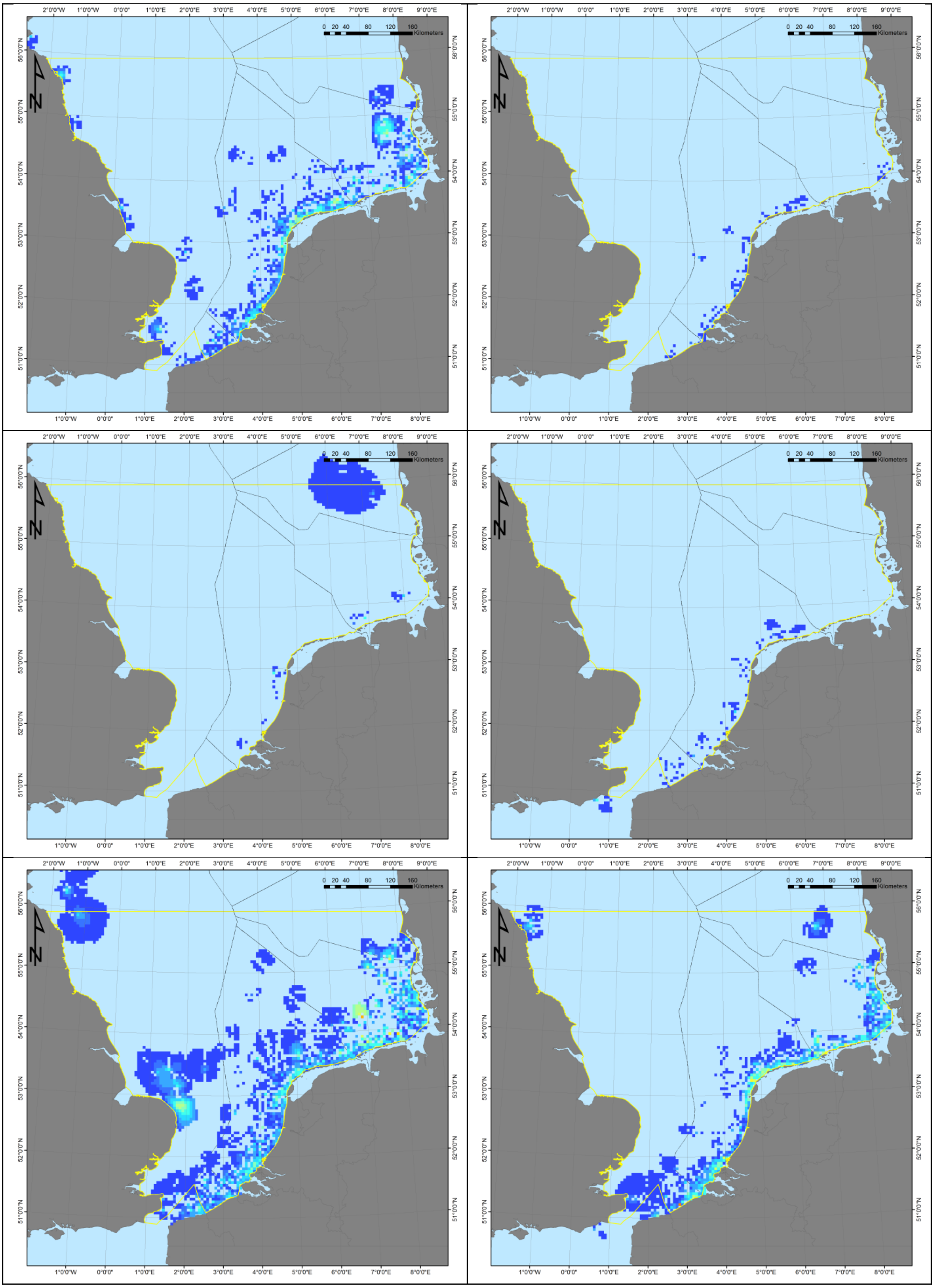


*Figure 4.37. Sandwich Tern: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Common & Arctic Tern (“commic terns”) *Sterna hirundo/paradisaea*

Common and Arctic Terns have a very similar appearance and behaviour at sea and cannot always be separated during surveys. Therefore, these two species are treated together as ‘commic’ terns. Like the Sandwich Terns discussed above, Common and Arctic Terns are summer visitors to the southern North Sea. ‘Commic’ terns were seen in significant numbers in the breeding season and during spring and autumn migration, with the highest densities in nearshore waters on either side of the southern North Sea (Figure 4.38).

Common Terns are far more numerous breeders in the study area than Arctic Terns that are more northerly breeders. Most birds seen at see were therefore likely to be Common Terns. These birds tend to remain within 10 km of the coastline (Poot *et al.* 2011b) on at-sea feeding trips. Birds seen further offshore are likely to be either migrants or loafing birds that had no bonds with breeding colonies at the time (Camphuysen 1991; Camphuysen & Winter 1996). Population sizes of both species around the North Sea are large and migrants from many colonies further north and northeast may pass through the study area. At the population level, these species do not seem at risk from offshore wind development, also because most birds are found in rather nearshore waters and because these terns mostly fly below rotor heights. However, a wind farm placed (too) closely inshore and close to a breeding colony might have an impact as commuting birds may get hit by the rotor blades (Everaert & Stienen 2007; Stienen *et al.* 2008; Leopold *et al.* 2013b).

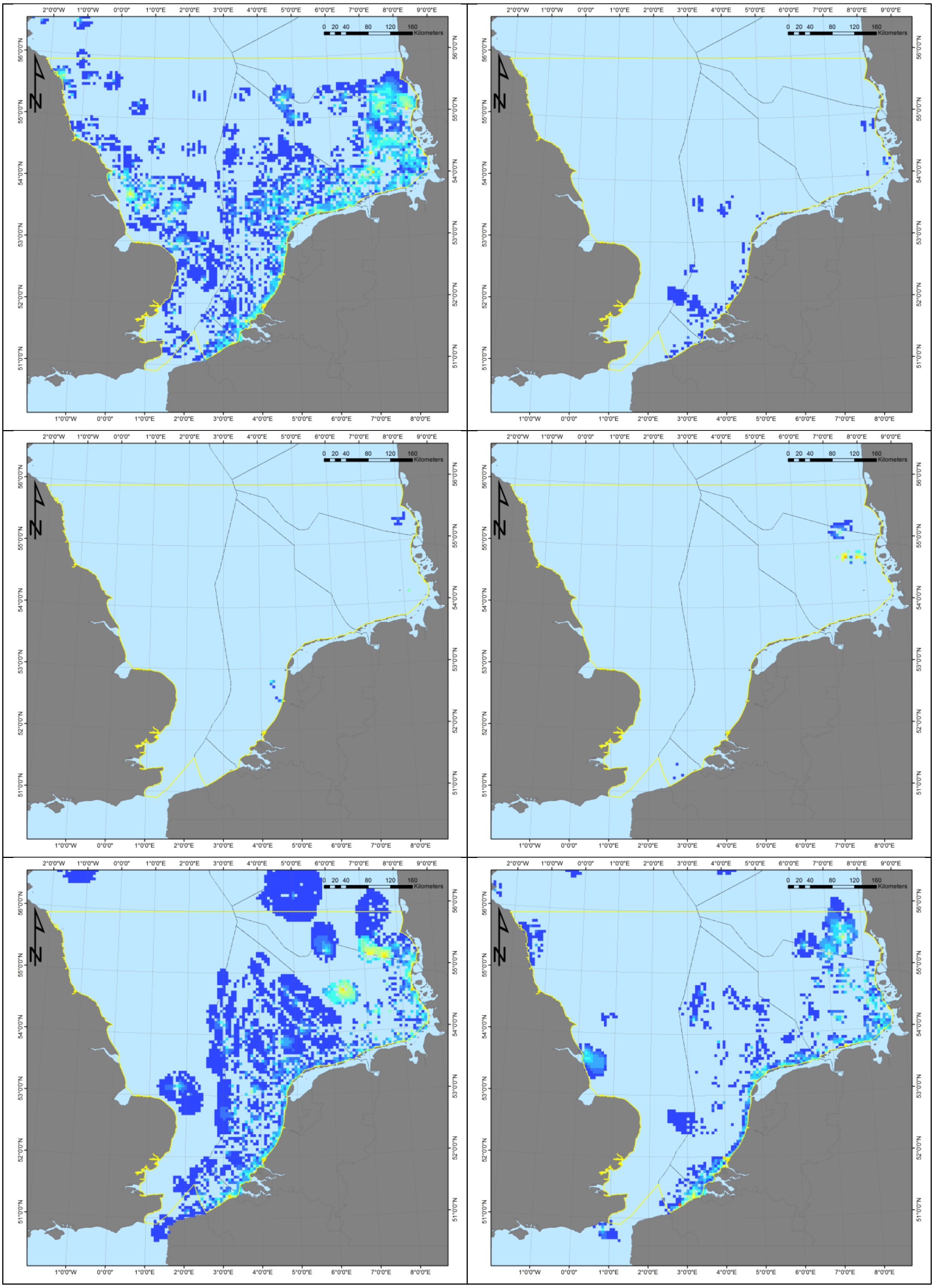


*Figure 4.38. Common Tern & Arctic Tern: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

Common Murre *Uria aalge*

Common Murres breed on cliff-coasts around the North Sea, the Baltic Sea, the northern Atlantic and Pacific Oceans (Nettleship & Birkhead 1985; Mitchell *et al.* 2004). The species breeds abundantly on the cliff coasts of the NW North Sea, but in the southern North Sea, breeding only occurs in one part of the English coast (Bempton Cliffs / Flamborough Head: 46 685 pairs) and on Heligoland in the German Bight 2500 pairs; Mitchell *et al.* 2004). Common Murres are one of the most abundant wintering seabirds in the North Sea, with an estimated winter population of 1 562 000 birds (Skov *et al.* 2007). The highest densities are found, year round, in UK waters, often with an eastward offshoot, south of the Dogger Bank along a frontal area known in The Netherlands as the Frisian Front (Figure 4.39). Here, numbers reach international importance in summer, when moulting birds and parent-birds with their chicks flock into the area (Van Bemmelen *et al.* 2013; Van Bemmelen & Leopold 2014). In the Southern Bight, numbers also reach international importance threshold along the UK/Dutch border, near the Brown Ridge (Van Bemmelen *et al.* 2012).

Common Murres are probably the most suitable birds to study effects of wind farms on seabirds, as they occur in relatively large numbers in many water types across the southern North Sea and are not attracted to fishing vessels. These features make them ideal for spatial modeling. Effect studies in and around wind farms have shown that Common Murres are susceptable to displacement by offhore wind farms, but that this displacement is not absolute: within wind farm parameters lower bird densities were found than expected on the basis of densities found in the vicinity (Elsam Engineering & Energi 2005; Elsam Engineering 2005; Petersen & Fox 2007; Leopold *et al.* 2013a; Vanermen *et al.* 2013; Walls *et al.* 2013). Results from these studies also indicated that this effect was not limited to the wind farm areas themselves, but that an area of several kilometres around offshore wind farms may also be avoided to some extent (see Annex D3). Murres are not likely to become victims of collisions in large numbers, as these birds rarely fly at rotor heights.

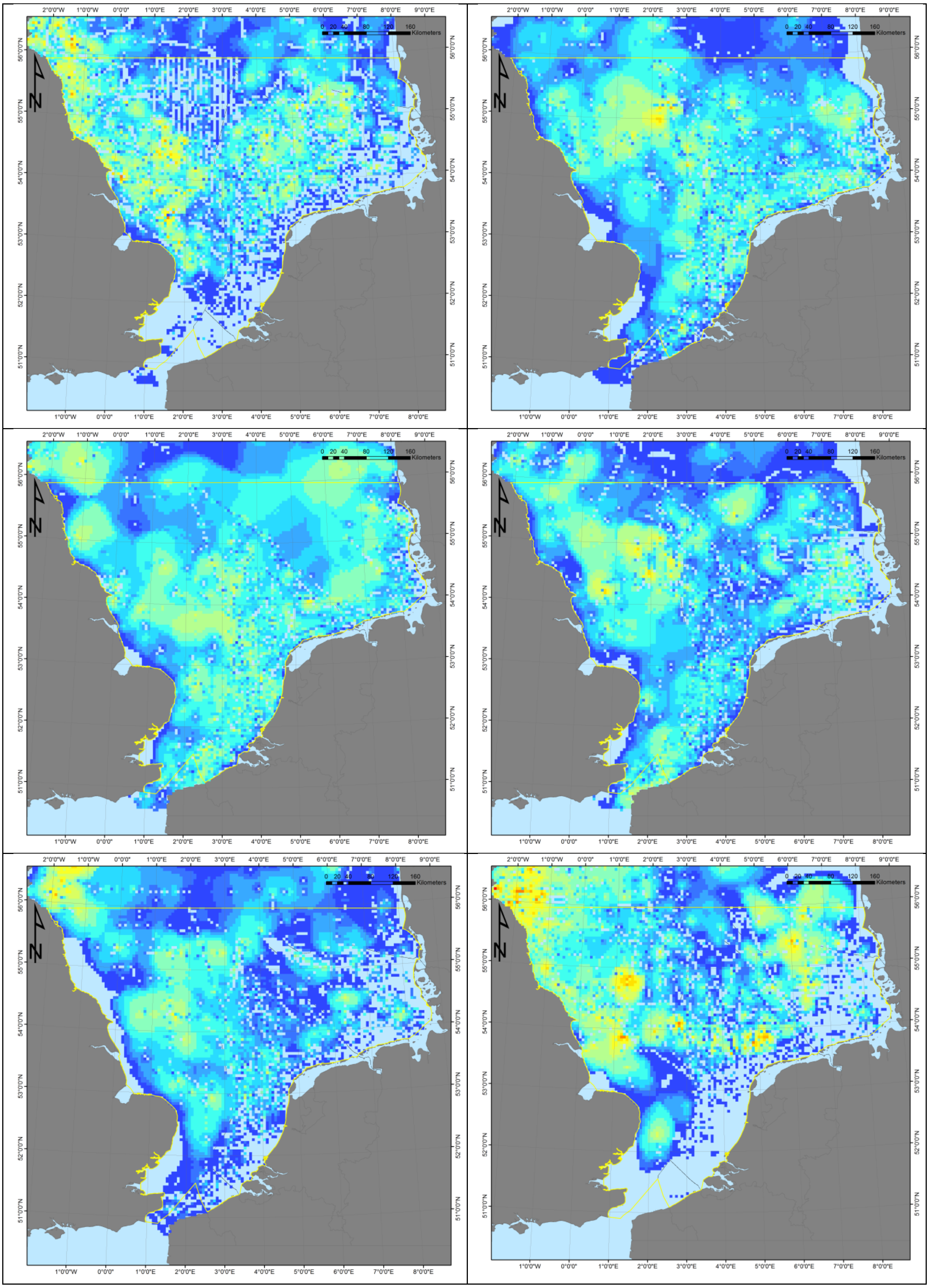


*Figure 4.39. Common Murre: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

Razorbill *Alca torda*

Razorbills are endemic to the North Atlantic, with two subspecies. The "northern" *Alca t. torda* breeds in Eastern North America, Greenland, Bear Island, Norway, White Sea-Murmansk and the Baltic, while *A. t. islandica* breeds in Iceland, the Faeroes, the British Isles, Heligoland and NW France (Del Hoyo *et al.* 1996). The latter, more southerly group is the main wintering subspecies in the southern North Sea (Camphuysen 1998).

In summer, most Razorbills are confined to waters near the breeding colonies in the NW of the study area, but they disperse widely over the entire southern North Sea at other times of year (Figure 4.40). Internationally important numbers may be found in autumn at the Dutch Frisian Front (Van Bemmelen *et al.* 2013) and in winter along the UK and Dutch border, around the Brown Ridge (Van Bemmelen *et al.* 2012). In the entire North Sea, some 324 000 Razorbills are found in winter (Skov *et al.* 2007); this involves birds from a rather large area, mostly from the northern and western UK, but also some more northerly birds, including some of the torda subspecies (Camphuysen 1998).

Razorbills are food specialists, both in the breeding season and in winter (Ouwehand *et al.* 2004). This could make them relatively vulnerable to offshore wind development, as the species tends to avoid swimming into wind farms in The Netherlands and Belgium (Leopold *et al.* 2013a; Vanermen *et al.* 2013) and probably also at Horns Rev, Denmark (Petersen *et al.* 2014), while displacement has also been found to be insignificant elsewhere, e.g. at the Robin Rigg offshore wind farm (Walls *et al.* 2013). The vulnerability with regard to collisions is probably low, because Razorbills rarely fly at rotor height.

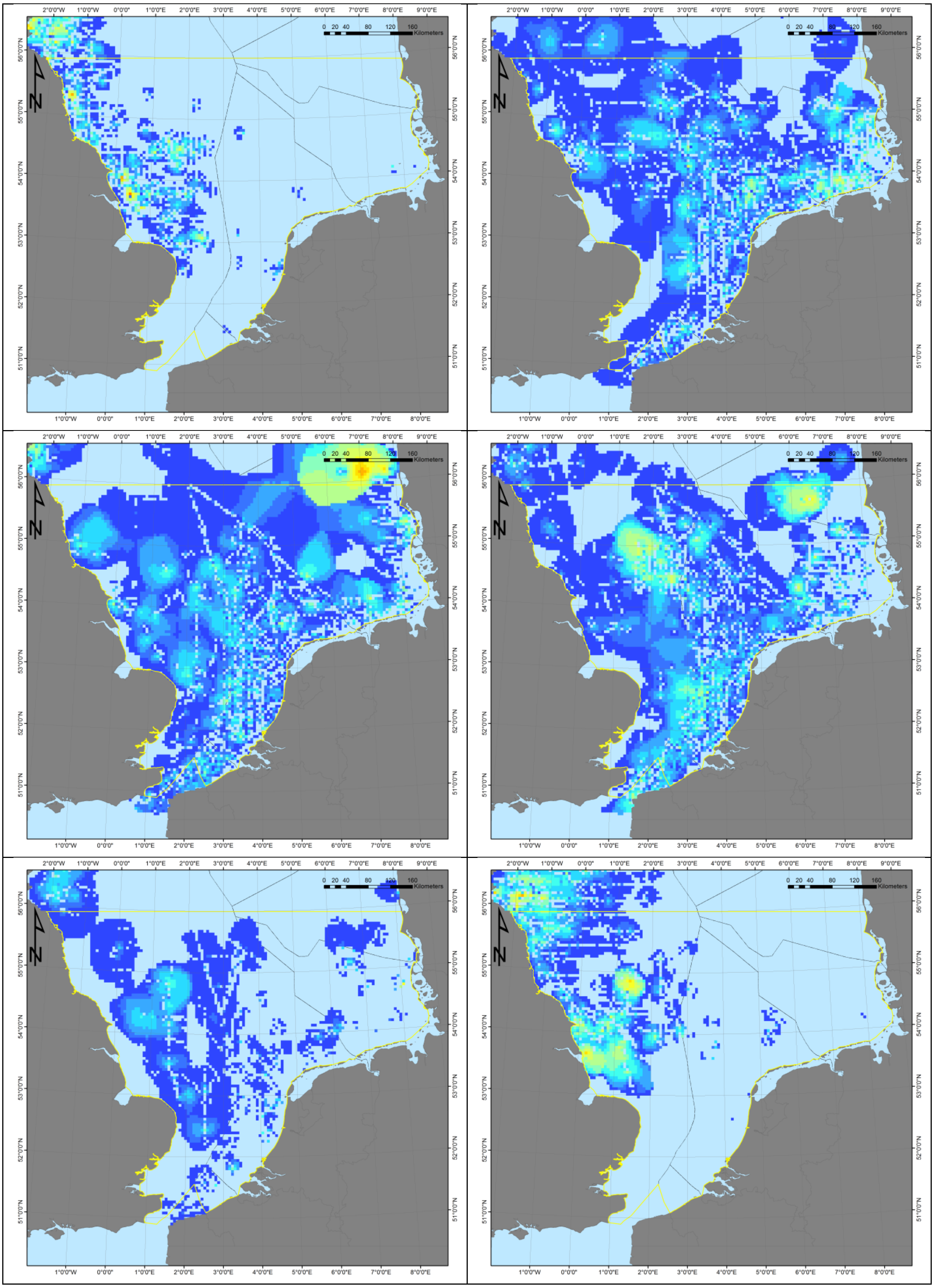


*Figure 4.40. Razorbill: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

Little Auk *Alle alle*

The nominate *Alle a. alle* is endemic to the northern Atlantic. The main breeding colonies are found in Greenland, Jan Mayen, Spitzbergen (Svalbard), Franz Josefland, Nova Zembla and Severnaya Zemlya. The species winters from the pack ice zone down into the nothern North Sea, and sometimes penetrates in an invasive-like manner futher south (Camphuysen & Leopold 1996; Del Hoyo *et al.* 1996). The total population size is over 12 milion pairs (Nettleship & Evans 1985) and an estimated 853 000 winter in the North Sea (Skov *et al.* 2007). In the North Sea, the main wintering ground is the Norwegian Deep and another part that often holds sizable numbers is found around the Dogger Bank (Skov *et al.* 1995): birds of this latter area can also be found within the study area (Figure 4.41). Little Auks found further south are mostly stragglers, probably with little chance of survival (Camphuysen & Leopold 1996).

Given the vast population size, the general behaviour of the species (mostly found on the water, and in flight, mostly at low altitudes) and its northern distribution, wind farms in the southern North Sea are unlikely to pose a significant threat to this species on the population level.

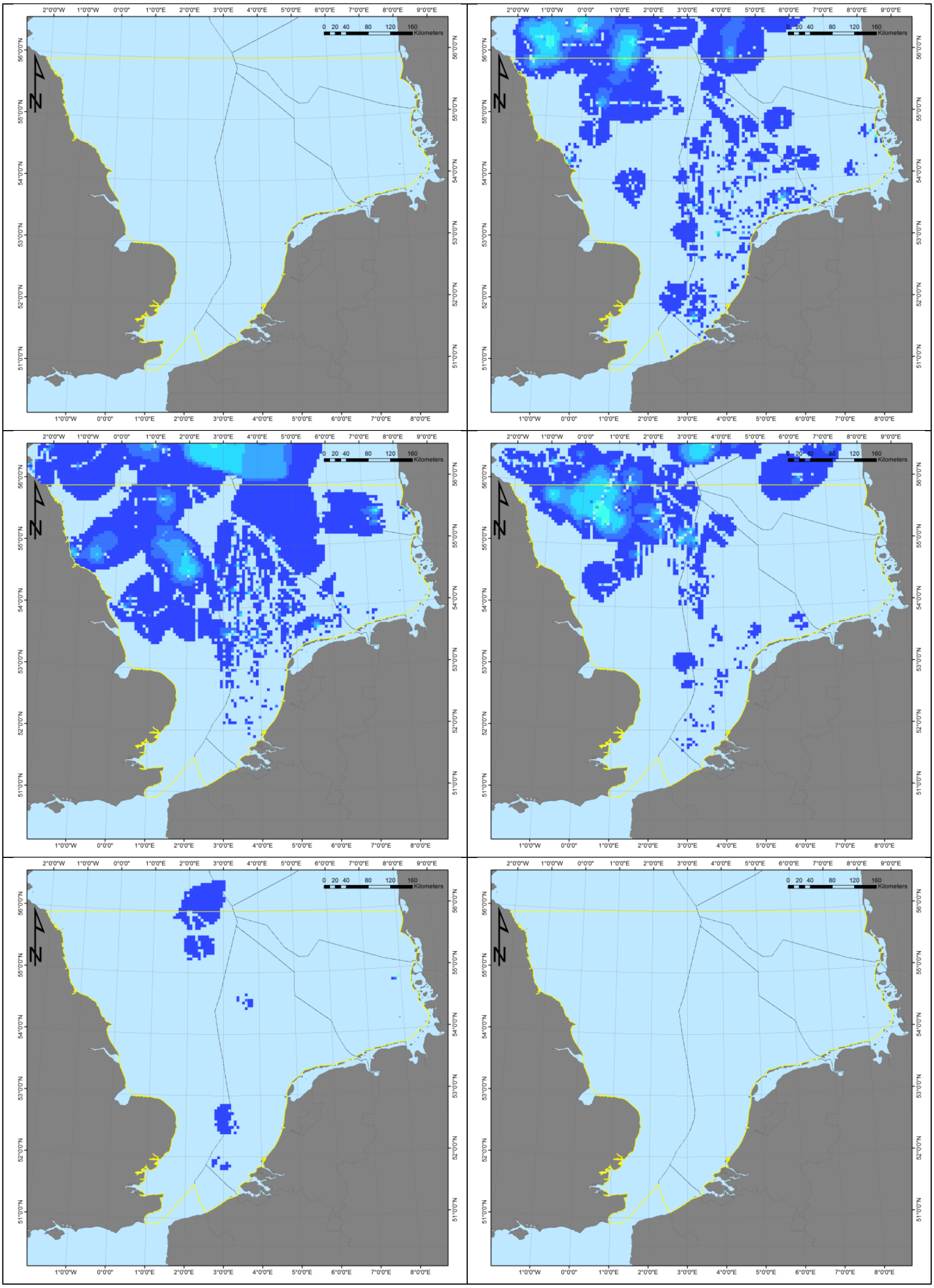


*Figure 4.41. Little Auk: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### Atlantic Puffin *Fratercula arctica*

Atlantic Puffins are endemic to the North Atlantic, breeding on both sides of this ocean. Colonies may be very large, and three subspecies are distinguished: the subspecies *Fratercula arctica grabae* is found breeding in the North Sea (UK, France, Sweden, Norway) and is the form that is most commonly found in the southern North Sea (Camphuysen 2003). Atlantic Puffins are among the commonest seabirds in the North Atlantic with a total population of 6-7 million pairs. However, numbers have been decreasing at many important colonies in the Atlantic over the last 20 years (Harris & Wanless 2011). In winter, the birds disperse widely over the North Atlantic, down to the western Mediterranean (Harris & Wanless 2011), where they live largely unnoticed, due to their small size, unobtrusive behaviour and highly dispersed occurrence at sea. Skov *et al.* (2007) estimate that circa 75 000 must be wintering in the North Sea.

The species is mainly found in the NW of the study area, in UK waters (Figure 4.42). In some winters birds show ‘invasions’ further south (Camphuysen 2003) but such birds may have poor chances of survival. Results from impact studies in offshore wind farms on the species are not yet available. The species is probably rather insensitive to collisions, as puffins live most of their lives at sea below rotor height. How they will respond to habitat loss due to offshore wind development remains to be resolved, but given that other auks tend to avoid going into offshore wind farms, habitat loss may be expected to become an issue for this species.

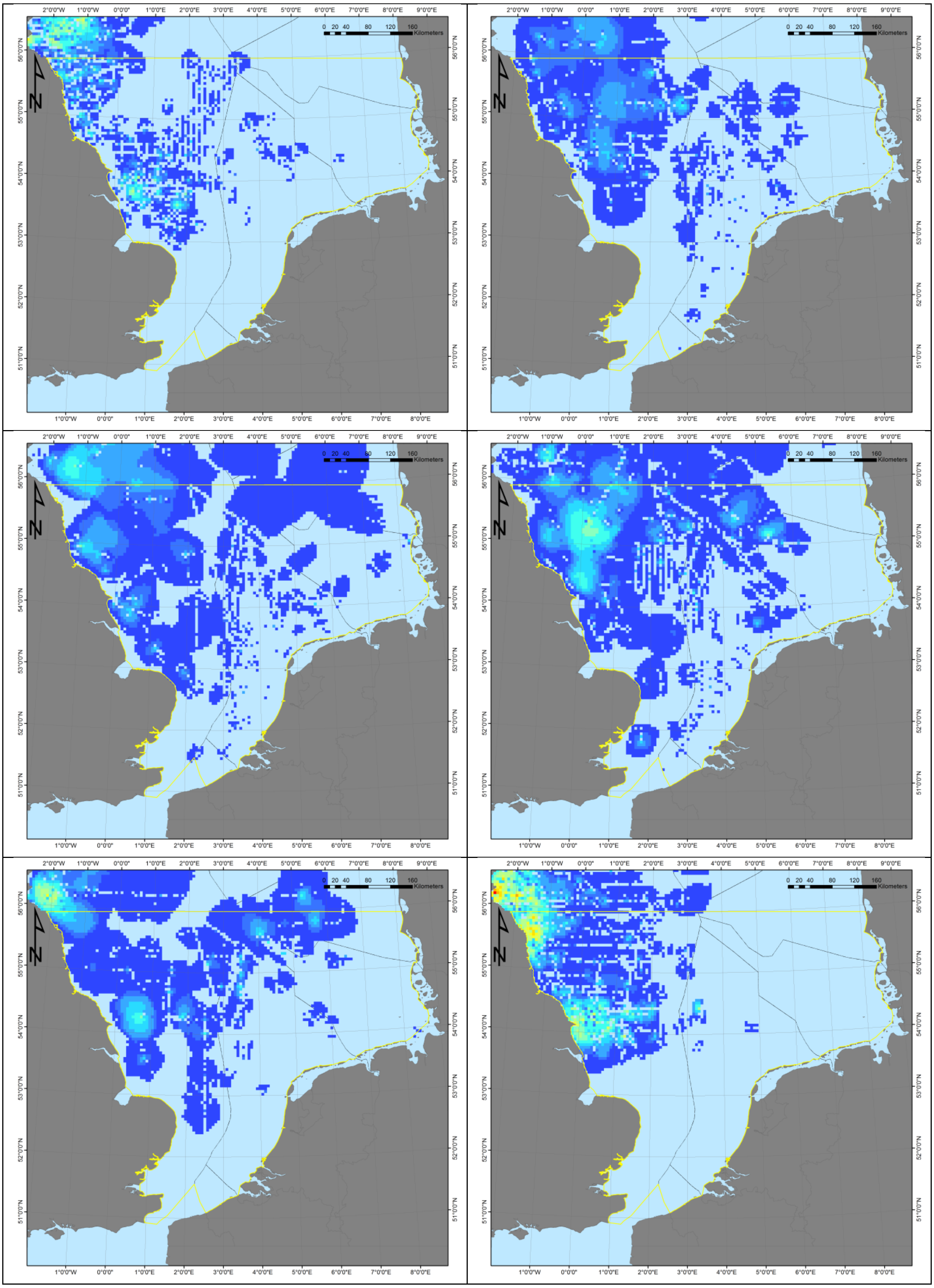


*Figure 4.42. Atlantic Puffin: distributions patterns in August/September, October/November, December/January, February/March, April/May and June/July, from top left to bottom right. For key to colours representing different densities: see Figure Key.*

### *4.2.6 Migratory birds species*

#### 4.2.6.1 Migratory bird species selection

The starting point is the list of species recorded in The Netherlands (514 species as of 22 August 2014, see Annex D1-D4). This list contains many vagrant species that are (very) rare in The Netherlands and therefore have a very low risk of colliding with a wind turbine in the Southern North Sea. By filtering out these vagrant species the list is reduced to 275 species that can be considered as regularly occurring in numbers that may result in a reasonable chance of collision with a wind turbine in The Netherlands or the southern part of the North Sea.

This list still contains a considerable number of bird species that do not usually fly over the North Sea, and can therefore be filtered from the selection. The remaining list includes those species that frequently fly (migrate) over sea and therefore have a reasonable chance to collide with wind turbines in the southern part of the North Sea. With the removal of rare and strictly terrestrial and non-migrant species, a total of 154 species remain.

*Table 4.8. Species selected for Collision Rate Modelling*

Tundra Swan[3]
Pink-footed Goose
Barnacle Goose
Brent Goose
Common Shelduck
Tufted Duck
Greater Scaup
Common Eider
Common Scoter
Velvet Scoter
Eurasian Wigeon
Red-throated Loon
Black-throated Loon
Northern Fulmar
Northern Gannet
Great Cormorant
Great Crested Grebe
Western Osprey
Peregrine Falcon
Grey Plover
Northern Lapwing
Red Knot
Sanderling
Dunlin
Common Snipe
Eurasian Woodcock
Bar-tailed Godwit
Eurasian Curlew
Common Redshank
Ruddy Turnstone
Parasitic Jaeger
Great Skua
Black-legged Kittiwake
Black-headed Gull
Little Gull
Mew Gull
Lesser Black-backed Gull
European Herring Gull
Great Black-backed Gull
Little Tern
Black Tern
Sandwich Tern
Common Tern
Arctic Tern
Common Murre
Razorbill
Short-eared Owl
Common Swift
Goldcrest
Eurasian Skylark
Barn Swallow
Willow Warbler
Common Starling
Common Blackbird
Song Thrush
Redwing
European Robin
Northern Wheatear
Western Yellow Wagtail
Meadow Pipit
Common Chaffinch

In order to achieve a list of representative and most relevant species potentially vulnerable to collisions with offshore wind turbines while migrating over the southern North Sea (51°-56°N), a final selection step was performed of on the remaining 154 species. The goal was to secure at least the selection of all species that: 1) mainly live at sea or at the shore, 2) regularly migrate over the southern part of the North Sea and have a relatively small population size which makes that additional mortality caused by offshore wind farms can easily result in a critical situation (effects on the population size and/or conservation status), 3) regularly migrate over the southern part of the North Sea in large numbers and are representative for a larger groups of species. The resulting list comprises 61 species for which Collision Rate Modelling is performed to predict the (cumulative) additional mortality caused by realisation of future wind farms in the southern part of the North Sea (Table 4.8).

#### 4.2.6.2 Estimation of the number of fatalities

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[3] Unless stated differently the Tundra Swan is *Columbinaus columbianus bewickii*, formerly known as Bewick's Swan. See also Appendix D.

The aim of this paragraph is to come up with an expert opinion on the number of migrating non-seabirds crossing the North Sea twice a year in order to calculate the total number of collisions offshore on a yearly basis. Firstly, the total catchment population is determined for all species, secondly it is determined which part crosses the North Sea, either to travel to and from wintering grounds in the British Isles or to make short cuts to and from more southern wintering areas. Thirdly an assumption will be made with respect to which part of the birds crossing the North Sea will fly at rotor height and also which part flies exclusively at night, this specifically in relation to collision risks. Based on the different radar studies done in the North Sea, gradients are determined in migration intensity over the studied part of the North Sea to estimate location-specific fluxes. These fluxes will be calibrated with the fluxes determined by the available radar studies, before they will be used as input into the Collision Rate Models in order to estimate location-specific numbers of fatalities per species group. Below for each 'category' of non-seabirds migrating over sea, a description is given of what we know about flight routes and flight behaviour in order to estimate the intensity of migration over the Southern North Sea.

Migration over the North Sea

The North Sea separates the European mainland (including The Netherlands) from Scandinavia and the United Kingdom. Every year tens of millions of birds migrate over the North Sea. Besides the typical seabirds, even larger numbers of shorebirds, waterbirds and true landbirds like small passerines migrate over the North Sea, on their way between their breeding grounds and their wintering areas, taking the shortest route as possible in case this is allowed by the weather conditions (Alerstam 1990, Lensink & Van der Winden 1997; LWVT/SOVON 2002). There are large differences between species groups with respect to flight routes, flight behaviour (diurnal versus nocturnal), flight height, and timing and intensity of migration over sea. Although the migratory destinations on land are quite well understood for many species, details of the routes that they follow when flying over sea are rarely known (Wright *et al*. 2012a,b). A general pattern established in several radar studies is that the intensity of migration is highest along the coast and gradually declines with distance from the coast (for the Dutch situation this was studied by Van Gasteren *et al*. 2002, Krijgsveld *et al*. 2005, 2011, Fijn *et al*. 2012). This pattern is explained by the leading line effect of the coast for both seabirds (strongest effects during onshore winds) as well as for shorebird and waterbird species following the coast flying over sea (Alerstam 1990). During the day strong leading line effects also occur in landbirds on the onshore side of the coast (LWVT/SOVON 2002). This phenomenon takes place most strongly during the day and is weather/wind dependent. Especially in case of head winds diurnal migrating birds avoid the crossing of open sea, minimizing the risk of getting exhausted, but also not to get predated (mainly by gulls). Crossing the North Sea by both nocturnal and diurnal migrants is strongly restricted to favourable conditions, mainly determined by the occurrence of tail winds. Still, during the relatively long flights birds, can get overtaken by bad weather, in which case birds can decide to change their migration direction dramatically in response to the changing weather circumstances.

Seabirds are defined as species that are fully dependent on food gathered at sea, either near-shore or (far) offshore. Landbirds are those species breeding (and feeding) in terrestrial habitats (including freshwater bodies). Outside the breeding season, many species still depend on terrestrial habitats. Others, like waterfowl and waders, occur in (semi) coastal marine environments.

The North Sea forms a barrier for landbirds migrating from the continent to Great Britain or *vice versa*. Migrants wintering in the UK mainly originate from Scandinavia and Northeast-Europe. Birds originating from (western) Scandinavia and wintering in Southwest-Europe or Africa may cross the North Sea as well. The last group of North Sea migrants consists of birds breeding in the UK, Iceland or Greenland and wintering in southern and southwestern Europe or Africa (Lack 1963, Lensink & Van der Winden 1997). A total of nine routes could be distinguished (Figure 4.43).

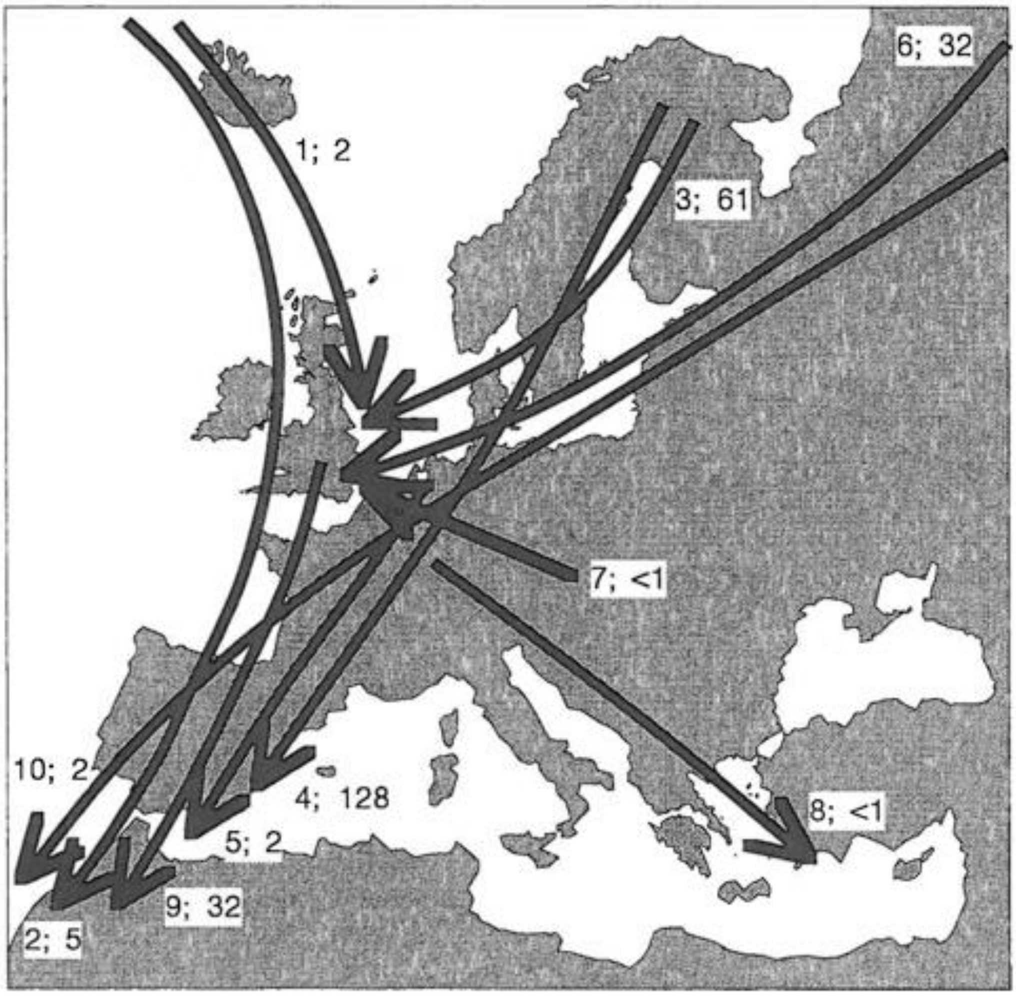

*Figure 4.43. The most important post-breeding migration routes of birds crossing the North Sea (after Lensink & van der Winden 1997). Figures stand for route number and number of migrants (millions).*

In summer the main flight direction of birds above the North Sea is SW-W (Lack 1962). In late summer, the time summer visitors leave for South-Europe and Africa, the main direction is S-SE. In autumn the main direction over the North Sea of migrants is W-SSW, with a substantial amount of birds flying S-SE. In autumn the latter probably fly according to the two-directional hypothesis formulated by Buurma (1987): birds leaving Scandinavia either start flying (N)W but change direction after several hours towards S- SE, or start flying S(E), several hours later followed by a track towards (S)W.

Landbirds

Quantitatively, landbirds form the majority of birds migrating over the southern part of the North Sea. The majority of landbirds migrating over the North Sea consists of passerines. Krijgsveld *et al*. (2011) estimated that approximately 70% of the birds recorded by radar at the Offshore Wind Farm Egmond aan Zee (OWEZ) were passerines.

Two general flight routes of migrating landbirds over the southern North Sea can be distinguished (Lensink & Van der Winden 1997):

1. Birds migrating from northeastern breeding grounds towards southern / southwestern wintering grounds follow to some extent the shoreline of the European mainland. For some species and under specific circumstances, leading line migration along the coast leads to an increase in the intensity of migration in a narrow band over the coastline and the adjacent few kilometres of sea.
2. Birds migrating from the European mainland and Scandinavia towards the United Kingdom cross the North Sea. Towards the south, the intensity of migration from the European mainland to the UK

decreases, which is probably caused by thrushes crossing the North Sea directly from Scandinavia towards the UK or The Netherlands. Intensity of migration of waders and waterbirds from the Wadden Sea to the UK also decreases from north to south. On the contrary the intensity of migration of diurnally migrating songbirds increases from north to south (along the coast), as most birds prefer to cross the North Sea at the narrowest point (Calais). These birds follow the shoreline south before crossing the North Sea.

Landbirds preferably migrate over large sea-surfaces under favourable weather conditions like tailwind, no precipitation and no dense cloud cover. Under these favourable conditions, most migrating landbirds like for instance passerines and waders fly at a height of several hundreds of meters up to >2 kilometre. In less favourable conditions (headwind) most birds fly in lower air layers (including rotor height).

In spring and autumn the intensity of migration of passerines is very high at the coast (Lensink & Van der Winden 1997; LWVT/SOVON 2002). This is partially caused by channeling of migration, which is particularly pronounced during daylight hours, but to a lesser extent also occurs at night (Buurma & Van Gateren 1989).

The largest number of landbirds migrating over the southern North Sea concerns passerines migrating between the European mainland and the United Kingdom. This involves species like Common Starling *Sturnus vulgaris*, Common Blackbird *Turdus merula*, Redwing *T. iliacus*, Song Thrush *T. philomelos*, Goldcrest *Regulus regulus*, Eurasian Skylark *Alauda arvensis*, European Robin *Erithacus rubecula* and Common Chaffinch *Fringilla coelebs* (Lensink & Van der Winden 1997; LWVT/SOVON 2002). Every year a few hundreds of thousands (Goldcrest, European Robin) to several million (thrushes, Eurasian Skylark and Common Chaffinch) or even >10 million birds (Common Starling) per species cross the southern North Sea in an east-west direction. In favourable weather conditions (tailwind), a lot of birds cross the North Sea at night.

There are also passerine species that mainly migrate in a north-south direction over the southern North Sea. These movements mainly concern long distance migrants, travelling from northern Europe towards southern Europe or even Africa (like Common Swift *Apus apus*, Barn Swallow *Hirundo rustica*, Willow Warbler *Phylloscopus trochilus*, Northern Wheatear *Oenanthe oenanthe*, Western Yellow Wagtail *Motacilla flava* and Meadow Pipit *Anthus pratensis*)

Apart from songbirds and waterbirds, also some birds of prey migrate over sea. Quantitatively this species group is not very important. However, as the biogeographical populations are smaller than those of most songbirds and waterbirds, birds of prey should also be considered in terms of potential cumulative effects of multiple wind farms in the southern North Sea. Western Ospreys *Pandion haliaetus* that migrate from their breeding grounds in northern and northeastern Europe towards southern Europe and Africa partly migrate along the coast of the North Sea or even over the North Sea. The same holds true for the Peregrine Falcon *Falco peregrinus*, although the birds from northern Europe do not migrate further than southern Europe. For both species, the amount of birds flying over the Southern North Sea will be limited to a maximum of a few hundred individuals. The Western Ospreys that breed in the UK mostly do not cross the Southern North Sea, as they migrate to the continent via France (Lensink & Van der Winden 1997; Wright *et al*. 2012a,b). The Peregrine Falcons that breed in the UK are resident birds (LWVT/SOVON 2002). The Short-eared Owl *Asio flammeus* migrates from N Europe towards W and SW Europe. The intensity of migration strongly depends on the availability of food (rodents / voles or mice). Lensink & Van der Winden (1997) estimated the intensity of migration over the North Sea (not only the southern part) to fluctuate somewhere between 1 and 10 000 birds.

Waterbirds and shorebirds

Besides the passerines, also various waterbird species migrate over the southern North Sea. This mainly concerns geese (like for instance Barnacle Goose *Branta leucopsis* and Dark-bellied Brent Goose *Branta* *bernicla*), swans (like the Tundra Swan *Cygnus bewickii*), ducks (like Common Shelduck *Tadorna tadorna*, Tufted Duck *Aythya fuligula*, Greater Scaup *A. marila* and Eurasian Wigeon *Anas penelope*) and several wader species from the saltwater environment (like for instance Grey Plover *Pluvialis squatarola*, Red Knot *Calidris canutus*, Sanderling *C. alba*, Dunlin *C. alpina*, Bar-tailed Godwit *Limosa lapponica* and Ruddy Turnstone *Arenaria interpres*) as well as the freshwater/land environment (like Northern Lapwing *Vanellus vanellus*, Common Snipe *Gallinago gallinago* and Eurasian Woodcock *Scolopax rusticola*) or both (like Common Redshank *Tringa totanus* and Eurasian Curlew *Numenius arquata*).

For species like the Tundra Swan, Dark-bellied Brent Goose, Eurasian Wigeon and Northern Lapwing, migration in an east-west direction over the southern North Sea plays an important role (Wright *et al*. 2012a,b). These species migrate in important numbers from the European mainland to the United Kingdom and vice versa. Of these species the Tundra Swan is of specific interest as the biogeographical population is relatively small (21 500 birds, Wetlands International 2014), which might easily cause even a small additional mortality to lead to effects on population size and/or conservation status.

For most other waterbirds, like for instance most wader species and especially those that breed high up north like Grey Plover and Red Knot, migration along the coastline of the European mainland in a northeast-southwest direction is dominant. Many of the birds following this route will more or less follow the coastline and migrate in close proximity of the coast, however, an unknown part of the birds also migrates further at sea. A substantial part of the migrational movements of birds along the coast of the North Sea concerns waders and waterbirds from the Wadden Sea.

A third important factor for some waterbird species is migration in summer. A good example is the Common Shelduck, of which the migratory movements are discussed in more detail below.

#### 4.2.6.3 Species group accounts of representative non-seabirds migrating over the North Sea

In 1997 a review on the number of migrants over the North Sea was compiled (Lensink & Van der Winden 1997). Based on published population estimates in different parts of the breeding range, in combination with information on the number of birds in the wintering area, the number of migrants migrating over the North Sea was estimated. These estimates include migrants crossing the North Sea from the continent and *vice versa* and the birds migrating on sea, but parallel to the coastline. For some species groups, new population estimates have been published after 1997. By connecting the main breeding ranges with the main wintering areas, nine migration routes were distinguished. Below an overview is given of the biogeographical population and the numbers of birds involved crossing the southern North Sea every year, for different groups of representative non-seabird species.

##### Tundra Swan

The Tundra Swan migrates from the breeding grounds in the NE Europe (Russia) to the wintering grounds in western Europe (and vice versa). Depending on the weather conditions, a part of the population migrates towards the United Kingdom. This means that twice a year approximately 7500 birds may cross the southern part of the North Sea (Lensink & Van der Winden 1997; Wetlands International 2014, Wright *et al*. 2012a,b).

##### Pink-footed Goose, Barnacle Goose and Dark-bellied Brent Goose

Birds from the breeding population of Pink-footed Goose *Anser brachyrhynchus* of Svalbard migrate through Denmark to The Netherlands and Belgium (Lensink & van der Winden 1997). A large fraction of these birds flies over the northeastern part of the southern North Sea. The breeding population at Svalbard consists of 63 000 birds (Wetlands International 2014), which means that annually 10 000 – 100 000 Pink-footed Geese migrate over a small section of the southern part of the North Sea. Barnacle and Brent Geese that breed in northern Russia migrate to their wintering grounds in western Europe.

Many of these birds migrate to or through The Netherlands and a large fraction migrates along the coast or over sea (Lensink & Van der Winden 1997). From both species (Barnacle and Brent Goose) several tens to a few hundreds of thousands of birds can migrate over the southern North Sea (Wetlands International 2014).

### Common Shelduck

Common Shelducks that breed in NW Europe winter along the coasts of the southern North Sea and the Atlantic coastline, until Africa. Almost the entire flyway population (300 000 birds, Wetlands International 2014) can winter west or south of The Netherlands, which means that a large part of the flyway population may migrate over the southern North Sea (Lensink & Van der Winden 1997). The Common Shelduck does not only migrate in spring and autumn, but also migrates in summer towards the German and Dutch Wadden Sea, where the birds go through the moulting process (Platteeuw 1980; LWVT/SOVON 2002). This means that most of the birds that breed in the UK (approximately 15 000 pairs, http://blx1.bto.org/birdfacts/results/bob1730.htm) cross the southern North Sea in summer, possibly resulting in a significant proportion of the population crossing four times a year.

### Tufted Duck, Greater Scaup, Eurasian Wigeon

Tufted Ducks, Greater Scaups and Eurasian Wigeons that breed in the northern Europe migrate in autumn towards their wintering grounds in (south)-western Europe. Part of these birds follow the coastline on their way south and migrate over sea. The wintering population of Tufted Ducks in the UK mainly originates from the Continent, while ringing records show that many of the birds that breed in the UK migrate towards the Continent (including The Netherlands) to winter (Lack 1986; Wernham *et al*. 2002). A large part of the flyway population of Eurasian Wigeons (>1 000 000 birds, Wetlands International 2014) winters in the UK and/or The Netherlands. Altogether this means that, in a worst case scenario, more than 100 000 Tufted Ducks and Greater Scaups and several hundreds of thousands of Eurasion Wigeons migrate over the southern North Sea (Lensink & Van der Winden 1997).

### Western Osprey, Peregrine Falcon

Western Ospreys that migrate from their breeding grounds in N and NE Europe towards Southern Europe (and, in the case of Ospreys, down into Africa) partly migrate along the coast of the North Sea or even over the North Sea. For both species the amount of birds flying over the southern North Sea will be no more than a few hundred. The Western Ospreys that breed in the UK mostly do not cross the southern part of the North Sea as they migrate to the continent via France (Lensink & Van der Winden 1997). The Peregrine Falcons that breed in the UK are resident birds (LWVT/SOVON 2002).

### Grey Plover, Red Knot, Sanderling, Dunlin, Bar-tailed Godwit

These wader species (partly) breed high up north, up to Siberia and Greenland and NE Canada and winter in southern Europe and Africa (LWVT/SOVON 2002). On their way from Siberia, Greenland or NE Canada to Africa, they follow the northwestern coast of Europe. The Wadden Sea is an important moulting place for Dunlins. After moult they disperse further south. Lensink & Van der Winden (1997) estimated that approximately 100 000 (maximum) Grey Plovers and several hundreds of thousands of Red Knots, Sanderlings, Dunlins and Bar-tailed Godwits migrate over the (southern) North Sea.

### Northern Lapwing, Common Snipe, Eurasian Woodcock, Eurasian Curlew, Common Redshank, Ruddy Turnstone

Northern Lapwings, Common Snipes, Eurasian Curlews and Common Redshanks migrate from N and NE Europe towards W and SW Europe. Movements of these species over sea, follow the western European seaboard or are conducted from the Netherlands, Germany and Denmark towards the UK, and *vice versa*). Of these species several hundreds of thousands of birds migrate over the southern North Sea (Lensink & Van der Winden 1997). An unknown, but probably very small number of Common Snipes migrate from Iceland to SW Europe. Most of these birds winter in the UK and do not, therefore, fly over the North Sea. Common Redshanks also migrate from Iceland towards NW Europe (e.g. to the Wadden Sea). This concerns approximately several tens to several hundreds of thousands of birds (Lensink & Van der Winden 1997). Eurasian Woodcocks migrate from N and NE Europe towards W and SW Europe. Migration takes place at night. The number of birds migrating over sea is unknown, but probably amounts to several hundreds of thousands of birds (Lensink & Van der Winden 1997). Ruddy Turnstones migrate from northern Europe, Greenland and NE Canada through western Europe towards NW Africa. Migration over the southern North Sea is estimated at several tens of thousands of birds (Lensink & Van der Winden 1997).

### Short-eared Owl

The Short-eared Owl migrates from northern and eastern Europe towards W and SW Europe. The intensity of migration strongly depends on the availability of food (rodents / mice). Lensink & Van der Winden (1997) estimated the intensity of migration over the North Sea (not only the southern part) at maximally 10 000 birds.

### Common Swift, Barn Swallow, Willow Warbler, Western Yellow Wagtail

These bird species migrate a long way from NW Europe towards S Africa. Barn Swallow and Willow Warbler are known to follow a more westerly route in autumn as compared to spring. Lensink & Van der Winden (1997) estimate the number of birds migrating over the North Sea (not only the southern part) at several tens of thousands for the Barn Swallow and the Western Yellow Wagtail, several hundreds of thousands for the Common Swift and even several millions for the Willow Warbler.

### Goldcrest, Eurasian Skylark, European Robin, Northern Wheatear, Meadow Pipit, Common Chaffinch

Goldcrest, Eurasian Skylark, European Robin and Common Chaffinch migrate in large numbers from the breeding grounds in N and NE Europe towards W and SW Europe (only a few birds winter in Africa). Many birds (at least several hundreds of thousands Goldcrests and European Robins and approximately a few million Eurasian Skylarks and Common Chaffinches) cross the southern North Sea on their way towards and from the UK (Lensink & Van der Winden 1997; LWVT/SOVON 2002). The Northern Wheatear and Meadow Pipit also migrate from Iceland and Greenland towards SW Europe and Africa. Most of these birds fly over the UK, but do not cross the southern North Sea. Most birds flying over the southern North Sea originate from N Europe. Lensink & Van der Winden (1997) estimated that no more than several tens of thousands of Northern Wheatears, but several hundreds of thousands of Meadow Pipits fly over the southern North Sea.

### Common Blackbird, Song Thrush, Redwing, Common Starling

Common Blackbirds and Common Starlings that breed in N and NE Europe migrate in autumn towards W Europe. The Song Thrushes and Redwings that breed in approximately the same region migrate a bit further towards SW Europe. Redwings from Iceland, however, migrate southward over the UK and do not fly over the southern North Sea. All four species migrate in large numbers, approximately 1-10 million birds per species or even >10 million birds (Common Starling), over the southern North Sea from the continent towards the UK and *vice versa* (Lensink & Van der Winden 1997).

#### 4.2.6.4 Estimation of collision fatalities using the Band model

In this section we already present the Band model used for collision rate modelling. It is applied to cross-check the results of the general approach chosen by us to estimate and assess bird sensitivity to offshore wind. The methods that are part of this approach, are outlined in chapter 3 and further explained in section 4.3. For details on the cross-check: see section 5.2.

Collision Rate Modelling using the Band model

Estimates on the cumulative number of bird victims at offshore wind farms in the Southern North Sea were calculated using the **SOSS Band model** (Band 2012). This model was developed in collaboration with The Crown Estate's (UK) Strategic Ornithological Support Services (SOSS), which was established to identify key ornithological issues relating to the UK offshore wind industry. The SOSS group consisted of experts from a number of countries including the Netherlands (Bureau Waardenburg, SOSS secretariat partner). In 2012, the Strategic Ornithological Support Services (SOSS) group published guidance on using a model for assessing bird collisions at offshore wind farms (Band 2012), which was accompanied by a spreadsheet for carrying out the calculations (hereafter called the Band model (2012)).

Rationale of the Band model

The Band model (2012) is developed around the core theoretical model for collision risk, first described in Tucker (1996) and later by Band (2000) and Band *et al*. (2007). This model calculates the **probability of collision** for a bird making a single transit through the rotor-swept area of an active wind turbine, **assuming no avoiding action by the bird**[4] (see also under: 'Justification of the chosen avoidance rate', below. This collision probability is then applied to the numbers of birds passing through the rotor-swept area giving an estimate for the number of collisions.

The Band model (2012) is specifically developed for use in offshore wind farms. It is species- and wind farm-specific, meaning that the number of collisions is calculated for a specific species with a specific wind turbine and applied to a specific number of turbines (wind farm). The model provides a standard method for calculating the numbers of birds passing through the rotor-swept area and consequently for the number of collisions of a specific wind farm.

The Band model (2012) provides two fundamental approaches for calculating the number of bird collisions (referred to as the Basic and Extended models) as well as the option of using data on the densities of flying birds or estimated fluxes (for local birds and migrant birds respectively).

Seabirds versus migrant birds

Calculations using the Band model (2012) can be based on either data on the **densities** of flying birds or estimated **fluxes**. The model has been primarily developed for using density data from ship-based or aerial survey data and aims to address seabird species that are well recorded by these types of surveys. Alternatively, a second variant of the model has been specifically developed for addressing migrant species and as such uses data on the numbers of bird passages (fluxes, e.g. measured by radar).

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[4] Tucker (1996) considers birds as being able to take evasive action through changes in flight speed in instances where relative rotor movement is below a certain level, such as close to the hub. This aspect is not described in the models in Band (2000), Band *et al*. (2007) or Band (2012).

Basic versus extended Band model

The Band model also provides two alternative methods for calculating collision probability. These are largely based on the type of flight height data available. The **basic model** is based on a mean collision probability across the rotor-swept area, whereas the **extended model** is based on the mean collision and encounter probabilities across the rotor-swept area weighted with respect to flight height. In essence, collision probabilities increase with decreasing distance to the hub and the probability of passing into the rotor-swept area (encounter probability) decreases with increasing distance from the vertical midpoint of the rotor (figure 4.44). This coupled with the **flight height distribution** results in a more representative collision probability with the extended model, for species with strong variation in distribution throughout the rotor height. The basic model is more suitable when flight height distributions throughout the rotor height are less well-known or other factors, such as the variation around estimates of numbers, are of more importance.

In the Band model (2012), species-specific **collision probabilities** are initially calculated assuming no avoidance, while avoidance rates are applied later during the modelling. The flight height distribution, as well as the collision probability are a function of **bird characteristics** (body length, wingspan, speed and flight type (flapping or gliding)), and the **wind turbine configuration** (number of blades, rotor radius, rotation speed, maximum blade width, pitch and hub height). Bird data, length (m) and wingspan (m), were taken from Snow & Perrins (1997a; 1997b) and for ranges the midpoint was taken. Alerstam *et al.* (2007) published flight speeds (m/s) for most of the species, which was supplemented in a few cases by published data elsewhere (Pennycuick 1997; Guilford *et al*. 2008; Welcker *et al.* 2009). For species where flight speed data were lacking, figures based on a closely related species were taken. Flight type was set at flapping flight for all species, including those that with good conditions mostly glide, such as Northern Gannet and gulls, as this results, in the modelling, in slightly higher collision probability than for gliding. In the Band model the flight type determines the shape of the bird as it passes the rotor. A gliding bird is modelled as a flat cross, whereas a flapping bird is taken to be more of a sphere, the latter implying a higher collision probability. Although a bird in active (flapping) flight can take avoiding action easier than a gliding bird, the Band model has removed this aspect from the Tucker model. Furthermore, it is likely that were the bird to respond to the rotors, it would likely to be flapping at the moment is crosses the rotors.

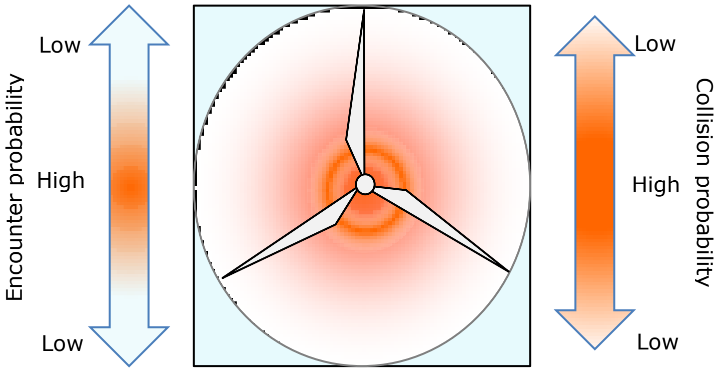


*Figure 4.44. Representation of variation in encounter probability with height and collision risk throughout the rotor-swept area. Combined, these two factors result in a low overall collision risk at the altitudinal extremities of the rotors.*

### Extended model for birds at sea

The current calculations for birds at sea were carried out with the extended model, where the collision probability for a transit through the rotor-swept area is weighted, based on the flight height distribution of each species, with respect to both the collision and encounter probabilities for the rotor-swept area. These flight height distributions were taken from the review study in Johnston *et al.* (2014).

### Basic model for migrant birds

The calculations for migrant birds were carried out with the basic model, this because detailed information on flight height distributions throughout the rotor height for birds during migration is largely lacking. Furthermore, variation in the estimates of the numbers of birds passing through a wind farm are considered more important to the number of collisions than estimates of flight height distribution and as such the basic model was used.

### All wind farms with 3 MW turbines – worst case

Wind turbine characteristics were set for all wind farms to a worst case scenario of all possible wind turbines, i.e. a relatively small, fast rotating turbine. This was found in the Vestas type V90-3.0 offshore turbine. This turbine has three blades, a rotation speed of 16,1 rotations/minute, 45 m rotor radius, 70 m hub height, 3,5 m maximum blade width and 15 degrees pitch. The collision risks of the relevant bird species by such a turbine are 12 – 36% higher than by another 3 MW but larger turbine (i.e. Vestas V112-3.0 MW). Note that for collision rate modelling 3 MW turbines have been used as these present the worst case scenario while for displacement (section 4.2.2) effects of 4 MW turbines are investigated, both as requested by the commissioner.

Based on the number of turbines, the total rotor-swept area ($m^2$) was calculated for each wind farm. For the wind farms Hornsea Project Three SPC 7 and 8 no turbine number was available, and the assumption was made that the number of turbines equals that of the wind farms Hornsea Project Three SPC 5 and 6 (i.e. 100 turbines).

### Densities of flying seabirds

The current calculations were based on ship-based counts and aerial surveys of seabirds, including two species groups (i.e. loon and commic terns) regularly occurring in the Southern North Sea. Subsequently, densities of these species were predicted in 3 km x 3 km mapping grid cells for the entire Southern North Sea for two-month periods. Bird densities were transformed to **flying bird densities** by applying the fraction of time spent in the air, given by Bradbury *et al.* (2014). These bird densities were averaged per wind farm area to come to one density (birds / $km^2$) for each wind farm of the Southern North Sea, with the assumption that the densities are equal in two subsequent months.

### Fluxes of migrant birds

The available information suggests that around 85 million birds cross the North Sea in autumn. In spring roughly 60% makes the return journey as well. The majority are thrushes, starlings and finches. Waterfowl, like geese, ducks and waders, are just a minority in the total number of birds crossing the North Sea. Details are summarized in Annex D5.

For each regular migrant species, an estimated number of migrants crossing the North Sea is given (Annex D6). These figures can be recalculated into a flux (mean traffic rate of birds, MTR as n/km). The distance between the southern tip of Norway and the border between Belgium and France, as the starting point of the Channel, is 750 km. If 85 million birds pass over a length of 750 km length, the MTR in autumn is around 114 000 ex/km. The maximum is calculated for European Starling (>30 000 ex/km).

*Justification of the chosen avoidance rate*

**Avoidance** is recognised as one of the most important factors in determining the actual collision rates of birds (Chamberlain *et al*. 2006, Drewitt & Langston 2006). Put simply, avoidance reduces the numbers of birds at risk of collision. Avoidance behaviour can be described at two levels: macro-avoidance: avoidance of the entire wind farm, and micro-avoidance: avoidance of individual turbines within the wind farm. The total avoidance rate can then be illustrated as:

Total avoidance = 1-((1-macro-avoidance) * (1-micro-avoidance))

Despite the attention on collisions and mortality rates with regard to studies on bird-wind turbines interactions, few figures for **avoidance rates** exist. The level to which numbers are reduced remains largely unknown, as avoidance has proven difficult to quantify and is likely to vary in response to a wide range of environmental and ecological factors, as well as the configuration of the wind farm (Krijgsveld 2014).

Based on the available evidence, it is widely accepted that total avoidance levels amongst birds are likely to be high, commonly higher than 98% and for many seabirds above 99% (Cook *et al*. 2012). Guidance from Scottish National Herritage (SNH) suggests using 98% as a default for many species including loons, gulls and terns (SNH 2010), although Cook *et al*. (2012) consider 99.75% may be a more realistic overall avoidance rate. Calculations of the current study followed the recommendations of Maclean *et al*. (2009), who recommended, based on a review of available studies, to use the avoidance rates of: 99% for loons, ducks, geese, grebes, Great Cormorant, terns and Atlantic Puffin; 99.5% for Northern Gannet, auks and gulls; and 99.9 % for Northern Fulmar and shearwaters, until additional information becomes available. Avoidance rates used were based on Cook *et al*. (2012) and Wright *et al.* (2012a,b).

Results – total number of collisions in seabirds

Using these three avoidance rates, actual species-specific collision rates were estimated for each wind farm of the Southern North Sea per month. Annex D2 shows the sum of the numbers of collisions in a year for all wind farms together. The total numbers of collisions per avoidance rate in the last row make the importance of selecting an appropriate avoidance rate clear. A decrease in avoidance rate from 99.9% to 99.5% results in a fivefold increase in collision numbers, whereas from 99.5% to 99.0% in a doubling of collision numbers. Using the species-specific avoidance rates recommended by Maclean *et al.* (2009), results in a total number of 45 287 seabird collisions per year for all seabird species in all of the wind farms of the southern North Sea together (on average approximately 6 collision seabird victims per turbine per year).

The largest number of collisions would occur among species with an avoidance rate of 99.5% (i.e. 42 975 collisions/year). Specifically, the largest number of collisions would occur among respectively Lesser Black-backed Gulls, Black-legged Kittiwakes and Greater Black-backed Gulls. Victims of these species would account for 77.4% of all collisions, with nearly half of all the collisions being Lesser Black-backed Gulls. All in all, gull species together would account for 89% of all collisions, with lower numbers for Mew, Black-headed and Little Gulls. Besides gulls, only Northern Gannets and Common Eiders would experience collision rates above 1 000 victims/year (i.e. 2631 and 1735, respectively). These two species, together with the seven gull species, would account for 98.6% of all seabird collisions in the southern North Sea.

The number of collisions varies greatly among wind farms from 4189 bird collisions/year at Thornton Bank II to 0.04 at Gunfleet Sands Demonstration Project (Annex D3). The six wind farms with the highest incidence of bird collisions account for approximately one third of all the collisions, with the top four wind farms providing nearly a quarter of all collisions (i.e. 24%). Besides Thornton Bank II, these wind farms are: Thornton Bank I, SER1 (or Borssele 1) and RENTEL, with all more than 2 000 collisions/year. Although these wind farms are not the largest, they still seem to pose a potential risk for high numbers of collisions. However, we think that the value of these figures should be taken with great care as potentially an overestimation of densities has taken place in species aggregating behind fishing vessels (particularly gulls). This problem is discussed further in section 6.1.

If collision rates are corrected for the number of wind turbines in a wind farm, the relative risk of a certain wind farm becomes more obvious. On average, six seabird collisions/turbine/year are expected. The largest number of collisions per wind turbine was calculated at Nordergründe. This relatively small wind farm (18 wind turbines) could cause 1841 collisions/year, but these calculated fatalities turned out to concern mainly Common Eiders, which very likely must be regarded caused by an earlier sighting (present in the ESAS database) of a flock of migrating birds, resulting in a local calculated abundance peak. The next wind farm (i.e. RENTEL) in the row would cause much less collisions, 27 turbine/year. In comparison, most of the wind farms (i.e. 99 of the 106) would cause less than 20 collisions/turbine/year.

### Results – total number of collisions in migrant birds

In order to arrive at a preliminary total number of collision victims in migrant birds, the fluxes measured at OWEZ by radar have also been used for comparison. These figures are for birds flying at altitudes of 25 m to 150 m. For birds flying within this height band it was assumed that birds were evenly distributed. Figures for both macro avoidance, birds that avoid the entire wind farm, and micro avoidance, birds within the wind farm that avoid individual rotors, were taken from figures calculated at OWEZ (table 14.1 in Krijgsveld *et al*. (2011)). Basic data for Band model calculations of victims as used in this report are presented in annexes 6, 7 and 8 in Poot *et al*. (2011a). Using the fluxes measured at OWEZ the number of collision victims in migrant birds is estimated to be around 95 000 birds, which translates into an approximate 11 victims per turbine per year, on top of the 6 collision seabird victims presented above.

Based on population estimates presented in Annex D5 for migrant birds crossing the North Sea, the number of collision victims were calculated on species level, applying the Basic Band model. The results on species level and per wind farm are presented in Annex D6. The total number of collisions in migrant species can be found in Table 4.24 in which the totals ares compared with the PBR and 1% of the natural annual mortality.

### Discussion

An important issue here is that the relatively high numbers of collision victims in gulls might be an artifact, resulting from a potential overestimation of reconstructed densities due to the incidental sightings of large numbers of birds, in many cases probably related to fisheries activities, and inverse distance weighing methodology used in this study for creating the seabird distribution maps. Unrealistically high densities may have been estimated for future wind farm sites, leading, in turn, to unrealistically high estimates of collision victims among these seabirds. Still, these gulls were seen in the (general) area and are at risk of collisions, even if no fishing will be allowed in future wind farms.

### *4.2.7 Bat species at sea*

#### 4.2.7.1 Introduction

For quite some time observers of bird migration along the Dutch coast regularly record bats flying in from sea (Nick van der Ham in litt., Michel de Lange in litt., Rinse van der Vliet in litt., Pim Wolf in litt., pers. obs. Sander Lagerveld & Maarten Platteeuw). Most sightings during coastal migration counts are made in de period from late April to May and during September /October. Bats have also been observed during ship-based surveys in the North Sea in September (pers. obs. Sander Lagerveld) and have been found on oil platforms, ships and remote islands (Boshamer & Bekker 2008; Petersen *et al.* 2014; Russ *et al.* 2001; Skiba 2007; Walter *et al.* 2007). In 2013 a Nathusius' Pipistrelle was found in the Netherlands, which was banded three years earlier in the United Kingdom (pers. comm. Teddy Dolstra), providing the first firm evidence that bats are able to cross the North Sea. The occurrence of bats at sea has also been reported from other areas, such as the Baltic Sea and off North America.

The observations of bats at sea resulted in an interest in monitoring bat activity with passive acoustic detectors at locations where bat migration can be expected. In 2011, bat calls were logged the coast at Lauwersoog and at the island Rottumeroog. Since 2012 bat activity is monitored at the Offshore Wind Farm Egmond aan Zee (OWEZ) and at Prinses Amalia Wind Farm (PAWP). In 2014, the IJmuiden meteorological mast (75 km offshore) and a coastal location near Egmond aan Zee were added to the monitoring network (Jonge Poerink *et al.* 2013; Lagerveld *et al.* in prep.).

#### 4.2.7.2 On the ecology of bats

Global annual cycle

Most bats in temperate climate areas are nocturnal insectivores. Because insects are not available during winter, they hibernate in buildings, caves or trees. Their active period starts in March or April when the winter roosts are abandoned. During this period they sleep during daylight hours and leave their roost at dusk and fly to their foraging areas, often along specific flyways (Dietz *et al.* 2007).

Bats use a network of roosts and move frequently from one to another. In early summer the females start forming maternity colonies, which may contain up to hundreds of individuals. Males use different roosts, individually or in small groups. Females usually produce only one young per year, but some (migratory) species have two. Juveniles become independent within 4-8 weeks and after that the maternity colonies are abandoned. The mating season for most species of bat starts late summer when males move to their mating roosts, which are often located near their winter roosts or along migration routes. During autumn bats store fat reserves enabling them to survive the winter. Late autumn or early winter bats move to their winter roosts where they enter hibernation again. Individual bats can live more than 20 years, but the population growth is rather limited due to the slow reproduction rate.

Migration

Several species of bats in northern Europe show seasonal migrations between their summer roosts and winter quarters (Figure 4.45). Most of them travel short or moderate distances, but some species like Nathusius' Pipistrelle *Pipistrellus nathusii*, Noctule *Nyctalus noctula*, Leisler's Bat *Nyctalus leisleri* and Parti-coloured Bat *Vespertilio murinus* are long distance migrants, travelling from northern and eastern Europe to more temperate areas and *vice versa* (Hutterer *et al.* 2005; Krapp & Niethammer 2011).

Some bats migrate individually while others migrate in groups, sometimes even large groups (Dietz *et al.* 2007). Most migrating bats that have been observed during coastal migration counts and surveys at sea, were single individuals. Groups were seen only on two occasions: one group of two and one group of four unidentified bats (probably Nathusius' Pipistrelle) at Camperduin flying in from sea (pers. obs. Sander Lagerveld).

Generally, bat migration occurs at night, but some species like Noctule have been seen migrating in the morning or just before dusk (Randler 2001; Mostert 2012; Gerjon Gelling *in litt*.).

*Figure 4.45. Migrating Noctule at the coast near The Hague, 4 October 2010 © Gerjon Gelling*

Little is known about their migration routes. At least some specific flyways are used, such as river valleys (Furmankiewicz & Kucharska 2009) and coasts (Petersons 2004; Dietz *et al.* 2007; Masing 2011; Šuba *et al.* 2012). Bats do not migrate exclusively over land. Large lakes, the Baltic Sea and the North Sea are crossed (Ahlén *et al.* 2007, 2009; Rydell *et al.* 2014; Morris 2014), but most bats seem to prefer to follow the shore (McGuire *et al.* 2012). Compared to birds, bats migrate relatively slow: on average 55 km/day for Nathusius' Pipistrelle, including stopovers (Rydell *et al.* 2014) and possibly less for other migrating species (30-50 km per day; Dietz *et al.* 2007).

Navigation

All bats in N Europe use echolocation. Echolocating bats emit ultrasonic pulses to gain information about their environment by listening to the returning echoes. This enables them to navigate during flight and to catch prey (Dietz *et al.* 2007). Some species of bat have distinctive species-specific echolocation calls, while others can be very hard to tell apart. Echolocation is not only used during darkness. A study in northern Scandinavia during summer showed that bats always used echolocation near the roosts and in areas further away (Speakman *et al.* 2000). Bats also use echolocation during nocturnal flight over sea (Ahlén *et al.* 2009), but it is apparently not known whether bats use echolocation during the day when flying over sea.

Bats are also able to navigate visually, although their eyesight is in general poorly developed. In most species of bat, vision depends on the exposure to light. In subdued lights at dawn and dusk their vision tends to be sharpest but it diminishes when it gets brighter (Orbach & Fenton 2010). Their vision is better beyond, than within the range of their echolocation (Suthers & Wallis 1970) and therefore visual cues are used to avoid obstacles and for navigation over larger distances, for example while commuting between foraging areas or during migration (Eklöf 2003).

Like many other animals, at least some species of bat are able to detect the earth magnetic field and it is likely that they use it for orientation during homing or migration by calibrating it with sunset cues (Holland *et al.* 2006, 2008, 2010). In addition, stars may be used as well for orientation (Childs & Buchler 1981).

### Weather and bat activity

The weather has a profound effect on bat activity. During nights with low temperatures, rain or strong winds, bat activity decreases significantly due to the reduced availability of prey (Winkelman *et al.* 2008) and increasing energetic costs by rain (Voigt *et al.* 2011).

Passive acoustic monitoring at the Offshore Wind farm Egmond aan Zee (OWEZ) and Prinses Amalia Wind Farm (PAWP) showed that most bat activity at sea occurred during nights with wind speeds up to 5 m/s (Jonge Poerink *et al.* 2013; Lagerveld *et al.* in prep.). This result exactly matches the pattern observed in southern Sweden by Ahlén *et al.* (2007, 2009). In addition, most bats were only observed during nights with high ambient pressures and no precipitation, shortly after periods with unfavourable weather conditions. This also corresponds with the findings of Ahlén *et al.* (2007, 2009) who observed that migrating bats aggregate at coastal locations and wait for favourable conditions to cross over sea.

### 4.2.7.3 Bat species at the southern North Sea

Some species of bat are more likely to occur at sea than others. In order to get insight into which species are relevant to consider for this study, we gathered information on the occurrence of all the species which occur on a regular basis in the countries bordering the southern North Sea. Main information sources for occurrence at the North Sea were Boshamer & Bekker (2008), Jonge Poerink *et al.* (2013), Lagerveld *et al.* in prep., Petersen *et al.* (2014), Walter *et al.* (2007), the North Sea Bird Club (www.abdn.ac.uk/nsbc/), www.waarneming.nl and www.observado.org. Records of migrating individuals along the coastline and observations of individuals flying in from sea were retrieved from www.trektellen.nl, www.waarneming.nl and Jonge Poerink (unpublished data). Information concerning offshore occurrence at the Baltic Sea were found in Ahlén *et al.* (2007, 2009).

Table 4.9 shows the species of bats which occur on a regular basis in England, Scotland, Belgium, the Netherlands, Germany and Denmark (Dietz *et al.* 2007). For each species it was established whether observations have been done in the study area, and whether offshore observations are known from the southern North Sea, the Wadden Sea or Baltic Sea. We consider species potentially relevant when they are either recorded within the study area but also when they have been exclusively recorded from one or more other offshore locations.

*Table 4.9. Offshore occurrence of bats at the southern North Sea and other areas (northern North Sea, Waddensea & Baltic Sea) and the relevance for this study.*

| **Species** | **Records** | | **Relevant in this study** |
|---|---|---|---|
| | **Study area** | **Other areas** | |
| Lesser Horseshoe Bat (*Rhinolophus hipposideros*) | No | No | No |
| Greater Horseshoe Bat (*Rhinolophus ferrumequinum*) | No | No | No |
| Daubenton's Bat (*Myotis daubentonii*) | No | Yes | Yes |
| Pond Bat (*Myotis dasycneme*) | No | Yes | Yes |
| Brandt's Bat (*Myotis brandtii*) | No | No | No |
| Whiskered Bat (*Myotis mystacinus*) | No | No | No |
| Alcathoe Whiskered Bat (*Myotis alcathoe*) | No | No | No |
| Natterer's Bat (*Myotis nattereri*) | No | No | No |
| Geoffroy's Bat (*Myotis emarginatus*) | No | No | No |
| Bechstein's Bat (*Myotis bechsteinii*) | No | No | No |
| Greater mouse-eared Bat (*Myotis myotis*) | No | No | No |
| Noctule (*Nyctalus noctula*) | Yes | Yes | Yes |
| Leisler's Bat (*Nyctalus leisleri*) | Yes | Yes | Yes |
| Common Pipistrelle (*Pipistrellus pipistrellus*) | Yes | Yes | Yes |
| Soprano Pipistrelle (*Pipistrellus pygmaeus*) | No | Yes | Yes |
| Nathusius' Pipistrelle (*Pipistrellus nathusii*) | Yes | Yes | Yes |
| Parti-coloured Bat (*Vespertilio murinus*) | Yes | Yes | Yes |
| Serotine Bat (*Eptesicus serotinus*) | Yes | Yes | Yes |
| Northern Bat (*Eptesicus nilssonii*) | Yes | Yes | Yes |
| Barbastelle (*Barbastella barbastellus*) | No | No | No |
| Brown Long-eared Bat (*Plecotus auritus*) | No | Yes | Yes |
| Grey Long-eared Bat (*Plecotus austriacus*) | No | No | No |

In the next section we describe the (potential) occurrence in the southern North Sea of each relevant species in more detail.

Daubenton's Bat *Myotis daubentonii*

Daubenton’s Bats commonly occur in the countries bordering the southern North Sea and beyond. This species is either a resident or a short-distance migrant. Usually it does not travel for more than 150 km between the summer roosts and the winter quarters. The majority of the individuals of this species forages over water or near water, but forests are also used. Foraging areas are usually located within a few kilometres from their roosts (Dietz *et al.* 2007). This species has been observed at the Baltic Sea off the Swedish coast (Ahlén *et al.* 2007, 2009), but has apparently never been recorded at the southern North Sea. We cannot rule out the possibility that it occasionaly may use the coastal sea as foraging area, but its occurrence further offshore seems unlikely.

Pond Bat *Myotis dasycneme*

The Pond Bat is an uncommon species on the European mainland and does not occur in the UK. It is a medium-distance migrant, travelling up to 300 kilometres between its summer and winter roosts (Dietz *et al.* 2007). Pond Bats prefer to forage over large freshwater bodies like lake IJsselmeer in the Netherlands, but also forage over the Wadden Sea (pers. comm. Anne-Jifke Haarsma). Passive acoustic monitoring at Lauwersoog produced various records during July – September 2011 (Figure 4.46 and 4.47). Additionally, there have been regular observations of this species at the Baltic Sea off the Swedish coast (Ahlén *et al.* 2007, 2009). It has never been observed offshore at the North Sea and therefore it seems unlikely that it migrates over sea. However, we cannot exclude the possibility that it uses the coastal sea as foraging area.

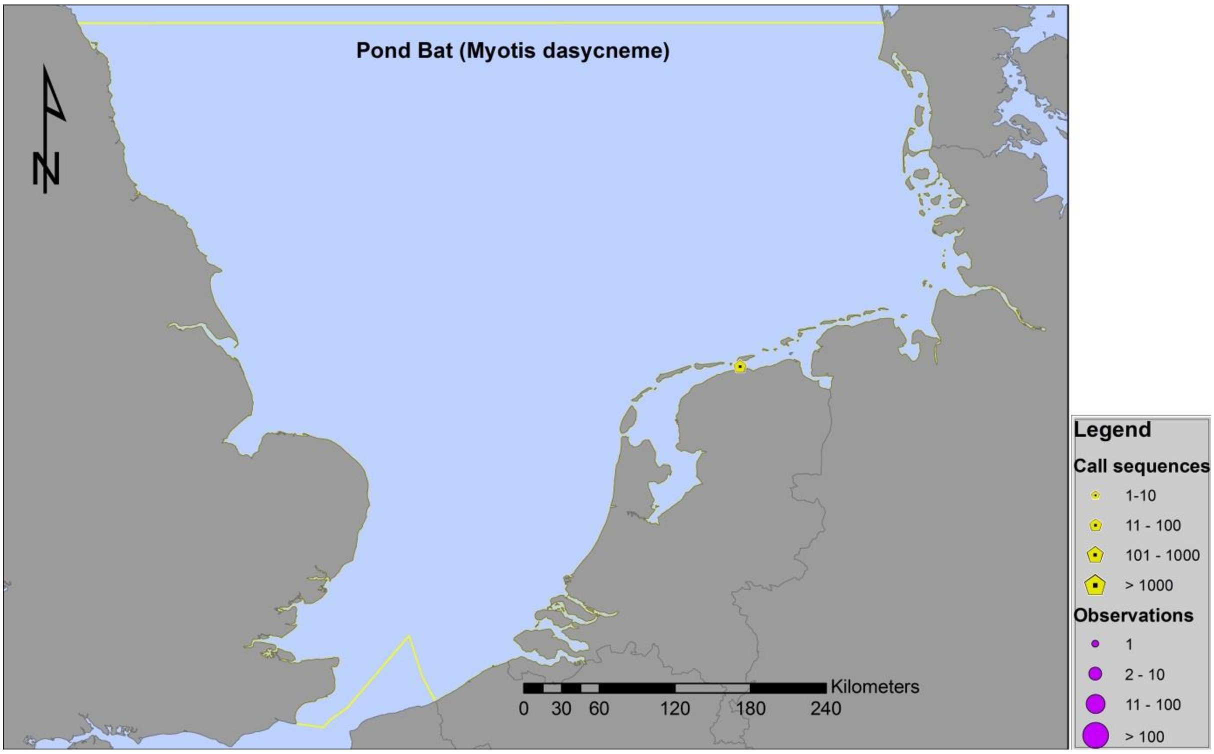


*Figure 4.46. Spatial occurrence of Pond Bat.*

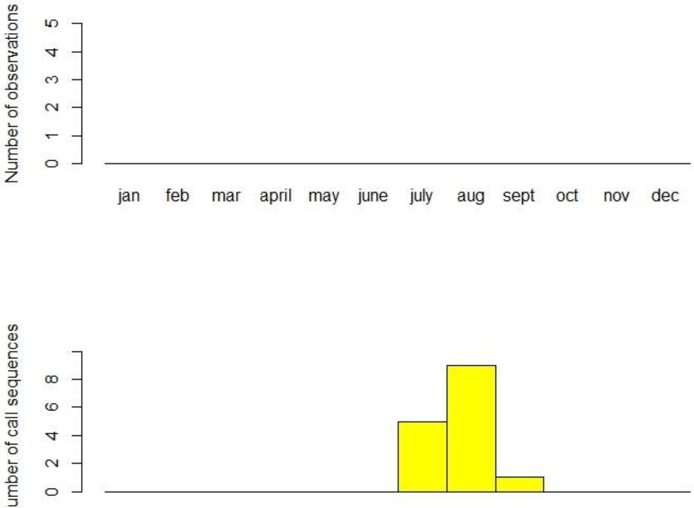


*Figure 4.47. Temporal occurrence of Pond Bat.*

Noctule *Nyctalus noctula*

The Noctule commonly occurs in the countries bordering the southern North Sea and further south and east. It is a tree-roosting bat and populations in NE Europe migrate over large distances in southwesterly direction in late summer / autumn. The maximum known distance is 1546 kilometre (Dietz *et al.* 2007). Noctules are frequently seen during migration counts during the day (www.trektellen.nl). The maximum daily count included 116 individuals migrating south along the coast near The Hague in 1.5 hours before dusk at 4 October 2010 (Mostert 2012, Gerjon Gelling *in litt*.). Noctules have also been recorded with acoustic detectors along the coast. Figure 4.48 shows the observed pattern of occurrence at Lauwersoog in 2012 (Jonge Poerink unpublished data). Note the obvious peak late August.

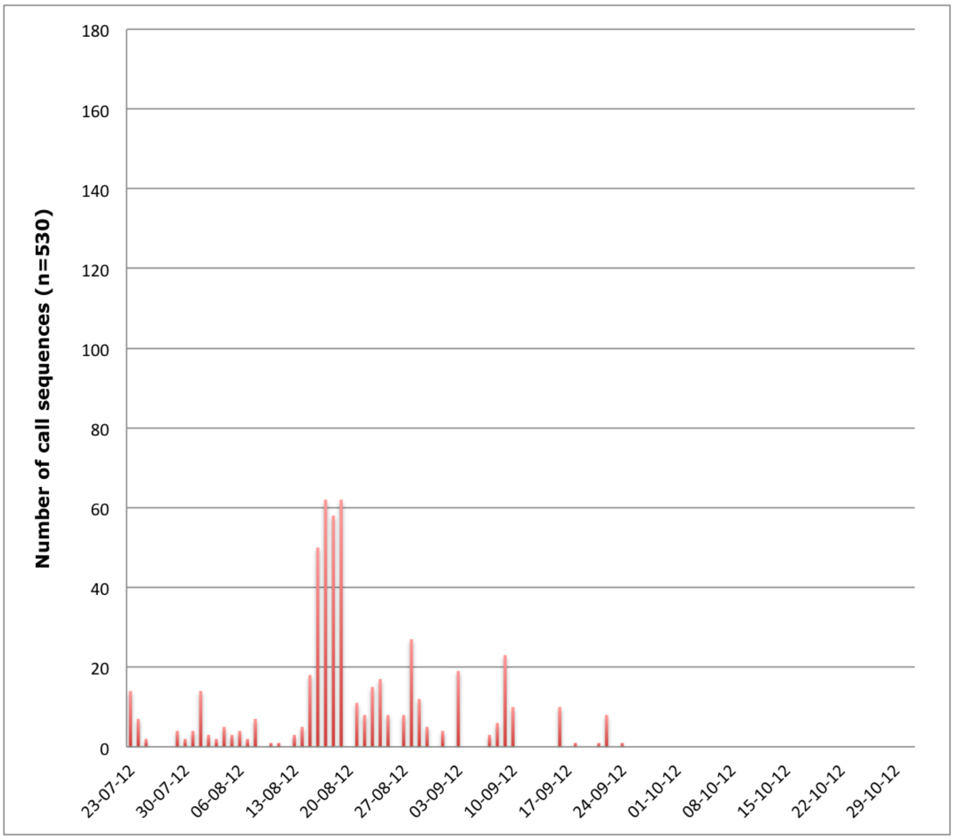


*Figure 4.48. Number of recorded call sequences of Noctule at Lauwersoog autumn 2012.*

In addition to coastal records, Noctules have been found on oil rigs and ships (Boshamer & Bekker 2008, Walter 2007) and were recorded with acoustic detectors at OWEZ (Jonge Poerink *et al.* 2013). Noctules have also been observed at the northern North Sea, Shetland and Orkney (Petersen *et al.* 2014) and are commonly recorded in the Baltic off southern Sweden (Ahlén *et al*. 2007, 2009). Figure 4.49 shows the locations where Noctules have been recorded over the southern North Sea. Most visual observations have been made during October whereas most acoustic activity (at the coast at Lauwersoog) was during August (Figure 4.50). All records of Noctule at sea and along the coast have been in the migration season and therefore it seems likely that Noctule is a regular migrant over the North Sea.

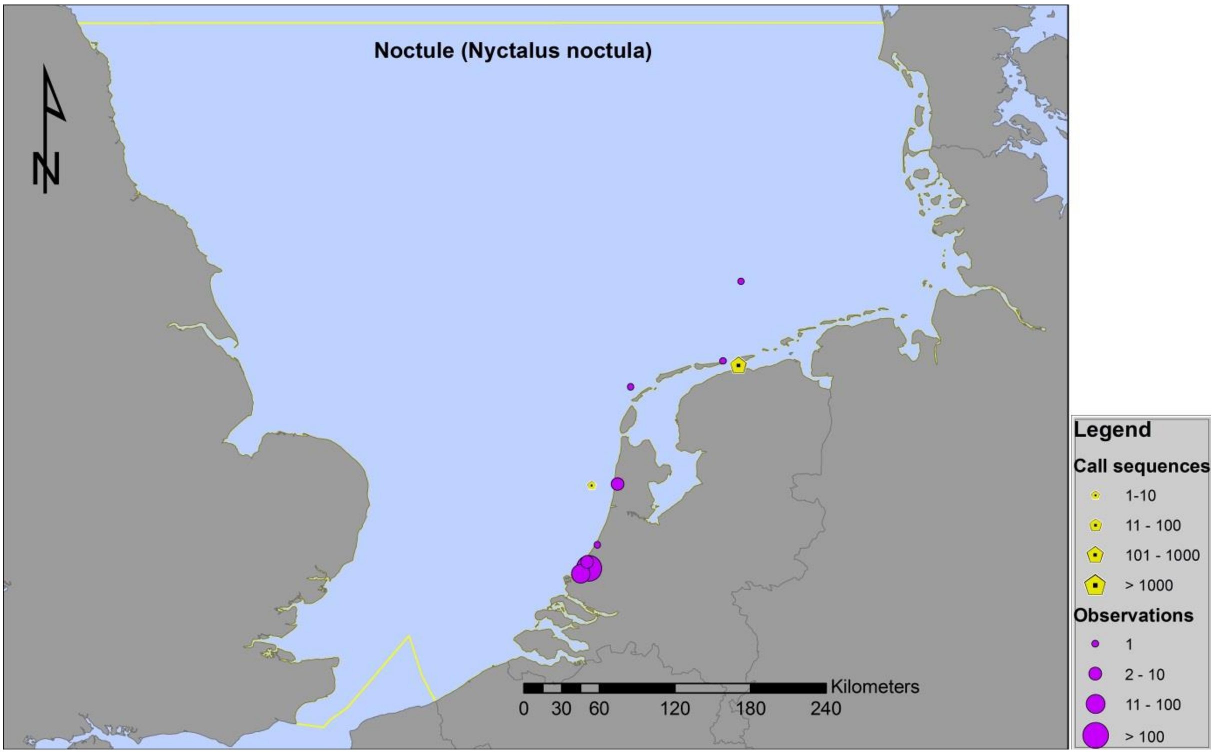


Figure 4.49. Spatial occurrence of Noctule.

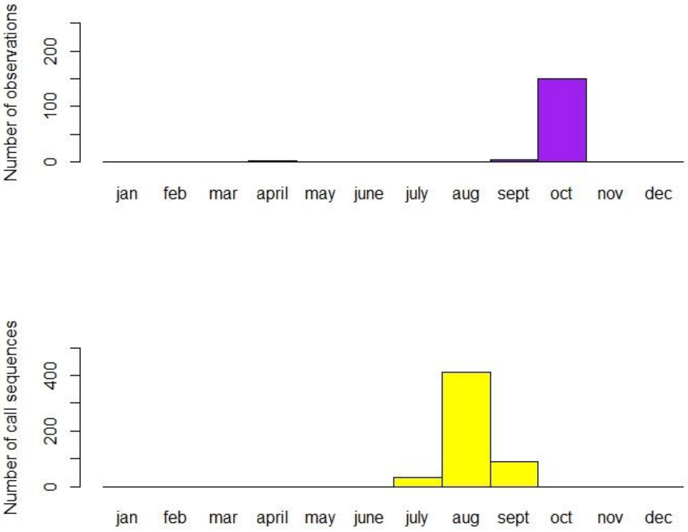


Figure 4.50. Temporal occurrence of Noctule.

Leisler's Bat *Nyctalus leisleri*

Leisler's Bat is rare in England, Belgium and the Netherlands. In Germany and further east it is more common. It does not occur on a regular basis in Denmark and in Scandinavia. It is a tree-roosting bat of which eastern populations show long-distance migrations up to 1500 kilometres in a SW ⇔ NE direction They seem to be more or less sedentary in the western part of their range (Dietz *et al.* 2007). There are a few records in the North Sea area (www.observado.org; North Sea Bird Club; Petersen *et al.* 2014) and therefore this species may be considered an occasional visitor to the North Sea. Figures 4.51 shows the geographical location of the sole record within the study area in October 2010.

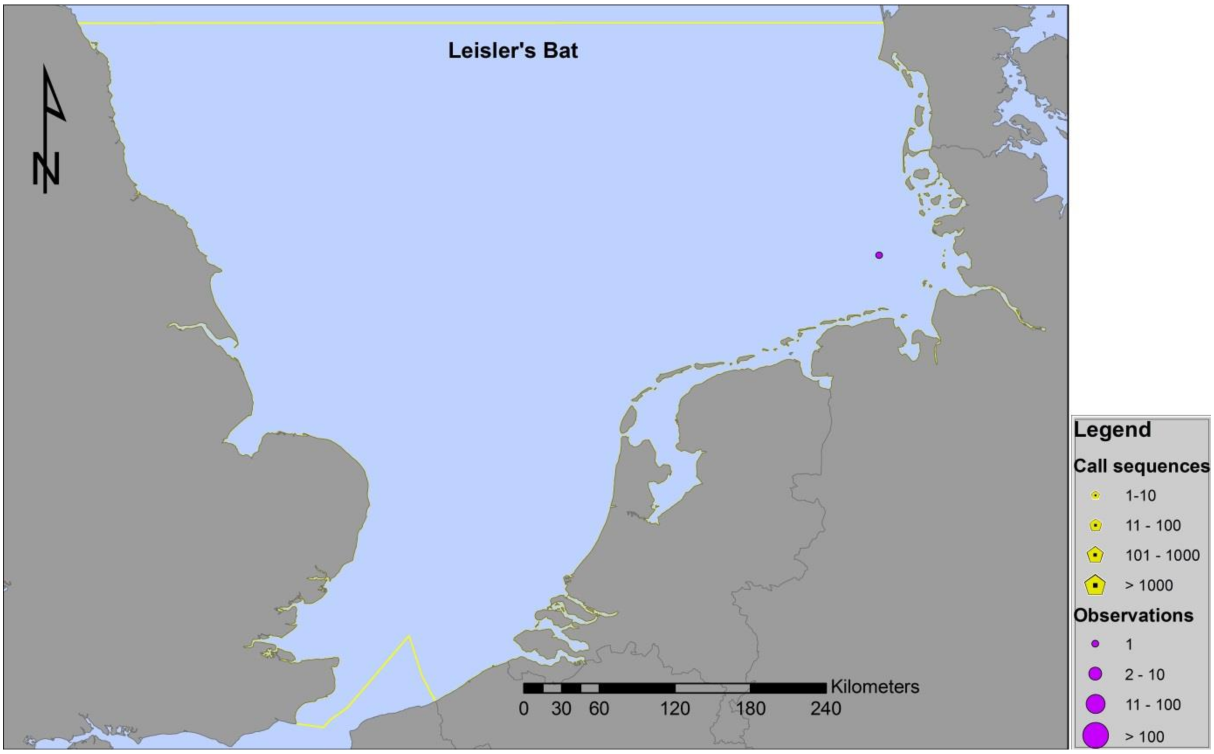


*Figure 4.51. Spatial occurrence of Leisler's Bat.*

Common Pipistrelle *Pipistrellus pipistrellus*

Common Pipistrelle is a common species in the countries bordering the southern North Sea. It is considered a a resident as seasonal displacements usually do not exceed 20 kilometers (Dietz *et al.* 2007). There are a few offshore observations from the Baltic Sea off Sweden (Ahlèn *et al.* 2009). No reliable records are known from the northern North Sea, with the exception of observations from Orkney which apparently refer to a local population (Petersen *et al.* 2014). There is apparently only one record from the southern North Sea of an individual which was recorded late August 2013 at OWEZ (Lagerveld *et al.* in prep.). The geographical location of the observation is shown in Figure 4.52. Therefore it seems likely that Common Pipistrelle is an uncommon or rare visitor to the southern North Sea.

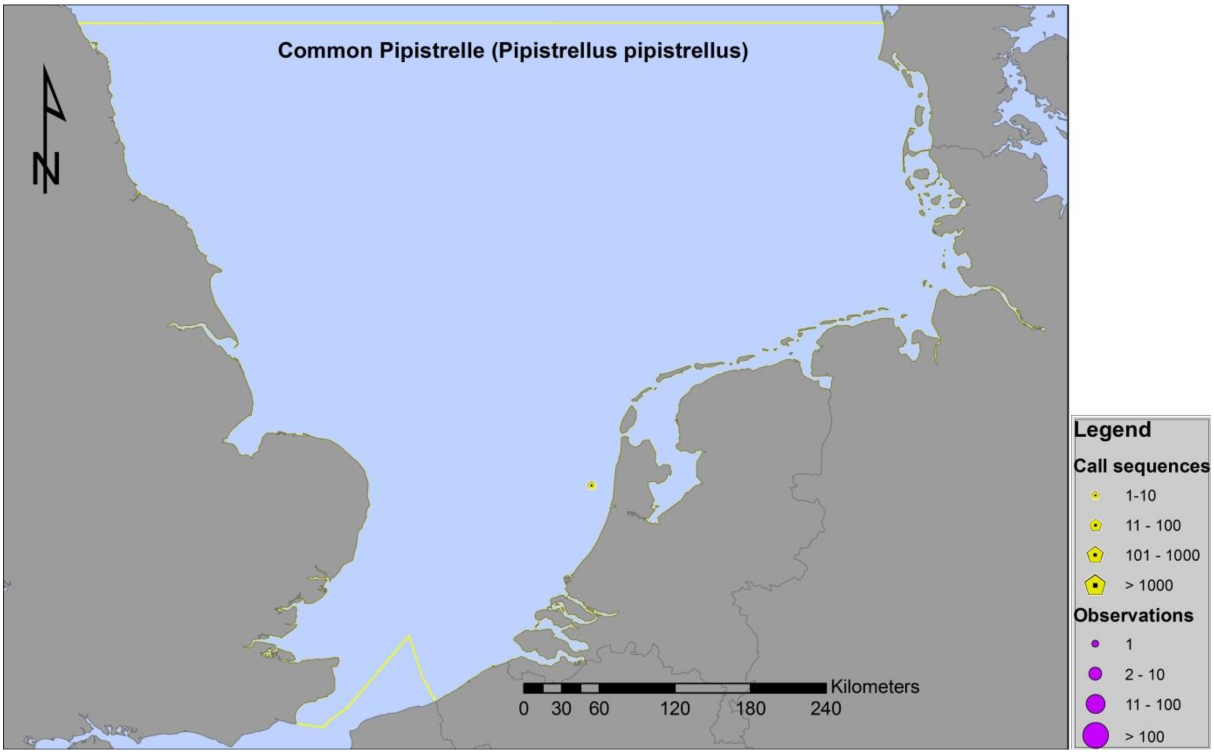


*Figure 4.52. Spatial occurrence of Common Pipistrelle.*

Soprano Pipistrelle *Pipistrellus pygmaeus*

The Soprano Pipistrelle occurs commonly in the UK (www.bats.org.uk), but is rare in Belgium and the Netherlands. It is considered a partial migrant, at least in the eastern part of its range, and it has been observed frequently at the Baltic Sea off Sweden (Ahlén *et al.* 2009). To date there are no records at the North Sea.

Nathusius' Pipistrelle *Pipistrellus nathusii*

Nathusius' Pipistrelle has an extensive range throughout Europe and the main reproduction areas are located in the northeast. It is a long-distance migrant with known displacements of up to 1905 km (Dietz *et al.* 2007). Migratory movements generally follow a NE ⇔ SW direction (Hutterer *et al.* 2005). Compared to other migrating bat species, Nathusius' Pipistrelle is by far the most common migratory species in the Netherlands. In autumn large numbers of Nathusius' Pipistrelle migrate along the Dutch coastline. An important migration route runs along the Afsluitdijk which connects the Dutch provinces of Friesland and Noord-Holland. Migration peaks coincide with favourable weather conditions: low wind speeds, no precipitation and high temperatures. Figure 4.53 shows the observed migration pattern at Lauwersoog during autumn 2012 (Jonge Poerink, unpublished data).

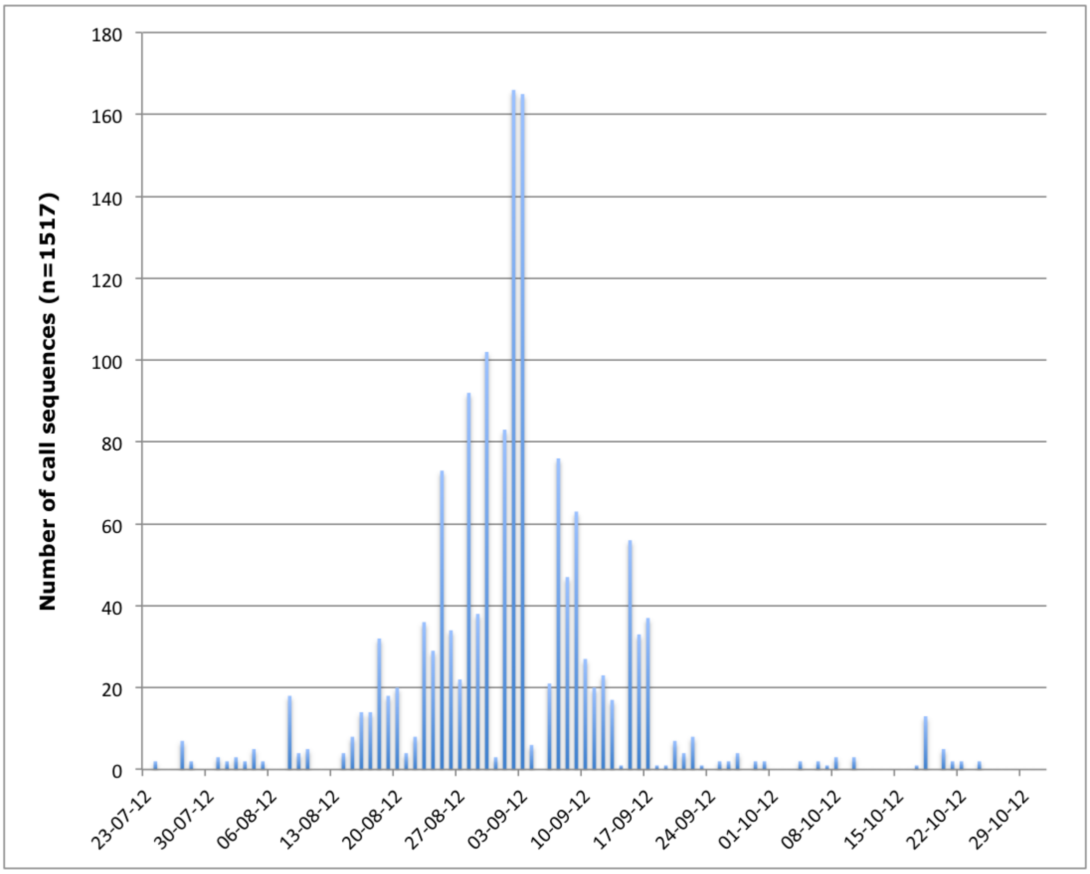


*Figure 4.53 Number of recorded call sequences of Nathusius' Pipistrelle at Lauwersoog autumn 2012.*

The abundance of migrating Nathusius' Pipistrelle along the Dutch coast corresponds with the number of observations offshore. Most observations at the North Sea of bats identified to species are Nathusius' Pipistrelles (Walter *et al.* 2007, Boshamer & Bekker 2008, Petersen *et al.* 2014, the North Sea Bird Club (www.abdn.ac.uk/nsbc/), www.waarneming.nl, www.observado.org). Monitoring with acoustic detectors at sea also revealed that Nathusius' Pipistrelle is by far the most recorded species at sea (Jonge Poerink *et al.* 2013, Lagerveld *et al.* in prep.).

Figures 4.54 and 4.55 show respectively the geographical locations and temporal occurence of the observations and acoustic detections at the southern North Sea. Both the observations and the acoustic detections are strongly linked with the migration season.

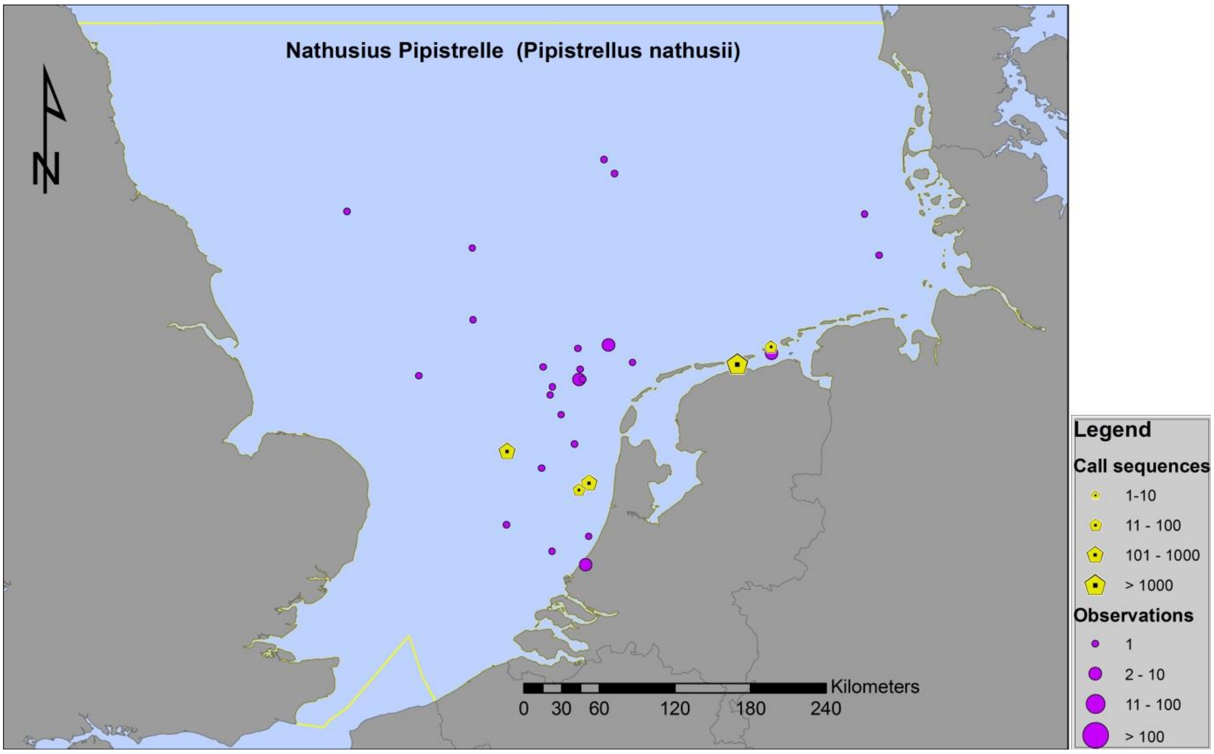


*Figure 4.54. Spatial occurrence of Nathusius' Pipistrelle.*

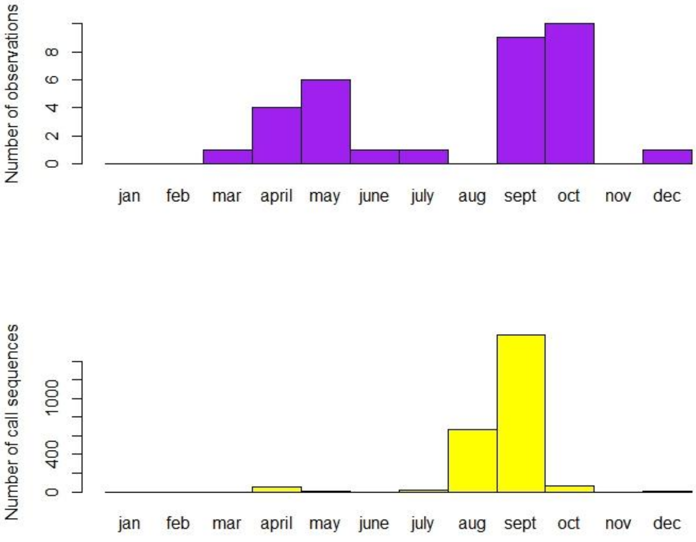


*Figure 4.55. Temporal occurrence of Nathusius' Pipistrelle.*

Serotine Bat *Eptesicus serotinus*

The Serotine Bat is an uncommon species in southern England (www.bats.org.uk), but is fairly common in the Netherlands and Belgium (www.waarneming.nl, www.waarnemingen.be). It is considered a sedentary species (Dietz *et al.* 2007). There is one record in the study area of a juvenile female which was found on an oil platform in July 2005 (Boshamer & Bekker 2008). The geographical location is shown in Figure 4.56. The Serotine Bat is probably only a rare visitor to the southern North Sea.

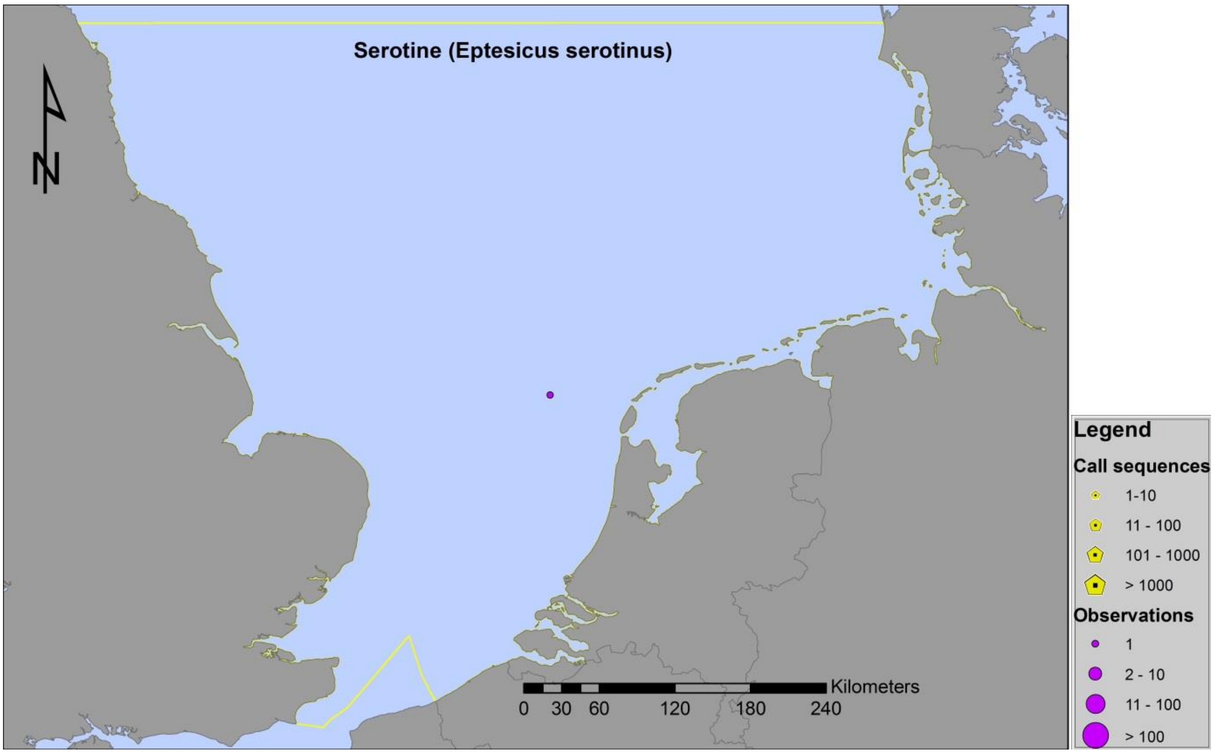


*Figure 4.56. Spatial occurrence of Serotine Bat.*

Northern Bat *Eptesicus nilssonii*

Northern Bat is a sedentary species that occurs from Scandinavia throughout central and eastern Europe. It does not occur on the mainland around the southern North Sea (Dietz *et al.* 2007). There are two records in the study area: one in June and one in September (Boshamer & Bekker 2008), and a few records further north (Petersen *et al.* 2014). We consider this species an occasional visitor to the southern North Sea. The geographical locations of the observations in the study area are shown in Figure 4.57.

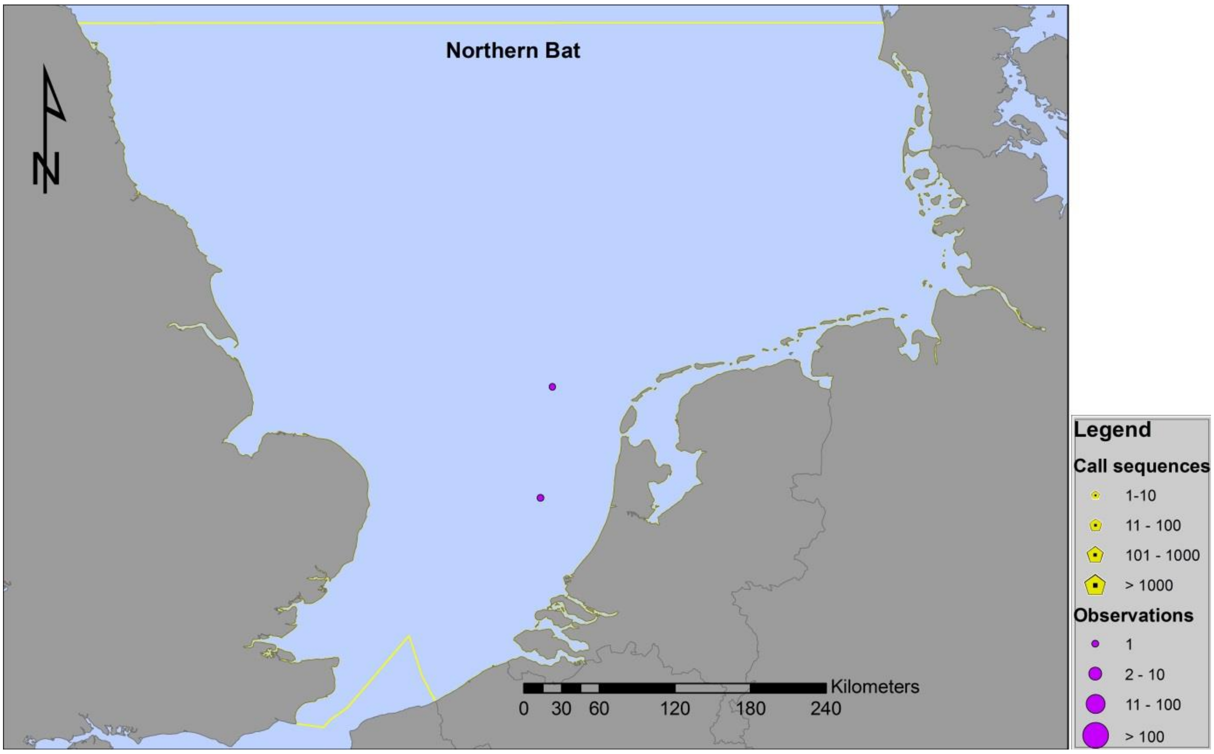


*Figure 4.57. Spatial occurrence of Northern Bat.*

Parti-coloured Bat *Vespertilio murinus*

Parti-coloured Bats do not occur in the UK and are rare in Belgium, the Netherlands and Denmark. The species occurs mainly in central and eastern Europe and in southern Scandinavia. It is a long-distance migrant, at least in the eastern part of its range (Dietz *et al.* 2007). In the Netherlands it is frequently observed during the migration period. Figure 4.58 shows the number of recordings in autumn 2012 at Lauwersoog (Jonge Poerink, unpublished data).

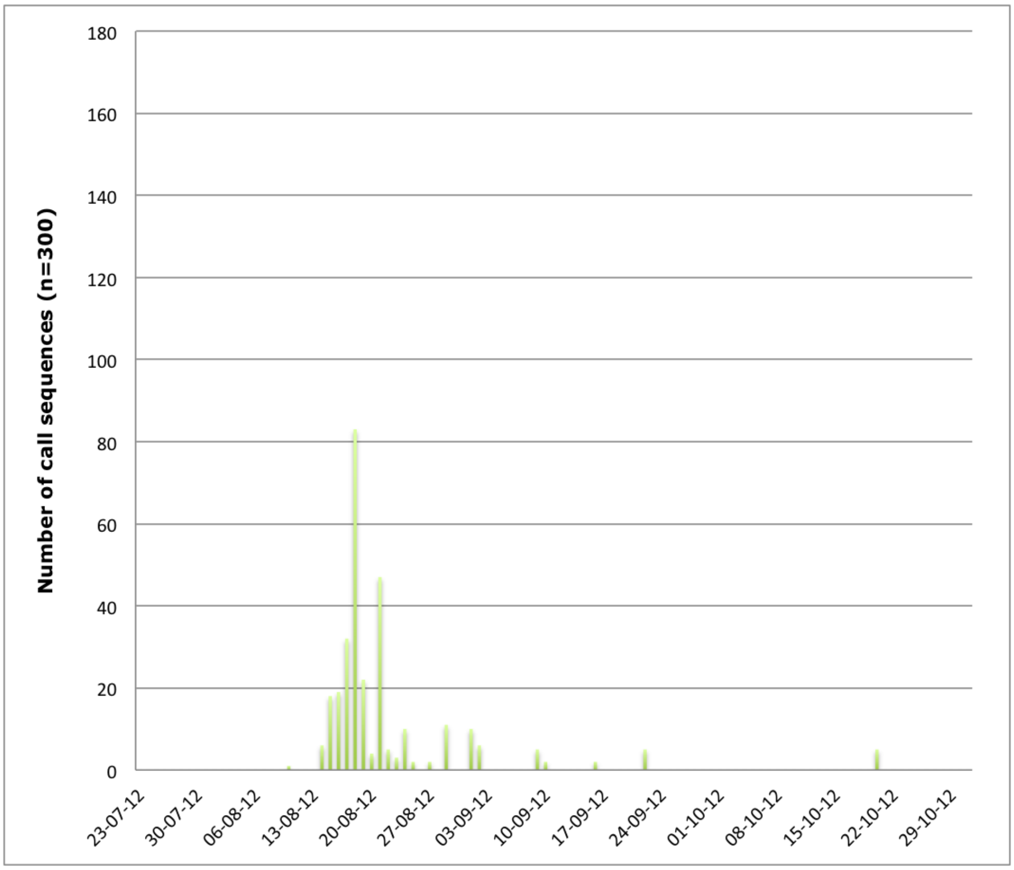


*Figure 4.58. Number of call sequences of Parti-coloured Bat per night at Lauwersoog autumn 2012.*

Several Parti-coloured Bats have been found on oil platforms at the southern North Sea (Boshamer & Bekker 2008, North Sea Bird Club) and it has also been observed quite frequently at the northern North Sea (Petersen *et al.* 2014). Probable Parti-coloured Bats were recorded during three nights at PAWP in autumn 2013 (Lagerveld *et al.* in prep.).

Figure 4.59 and 4.60 show respectively the geographical locations and temporal occurence of the observations and acoustic detections of Parti-coloured Bats at the southern North Sea. Interestingly, this species has also been recorded in winter.

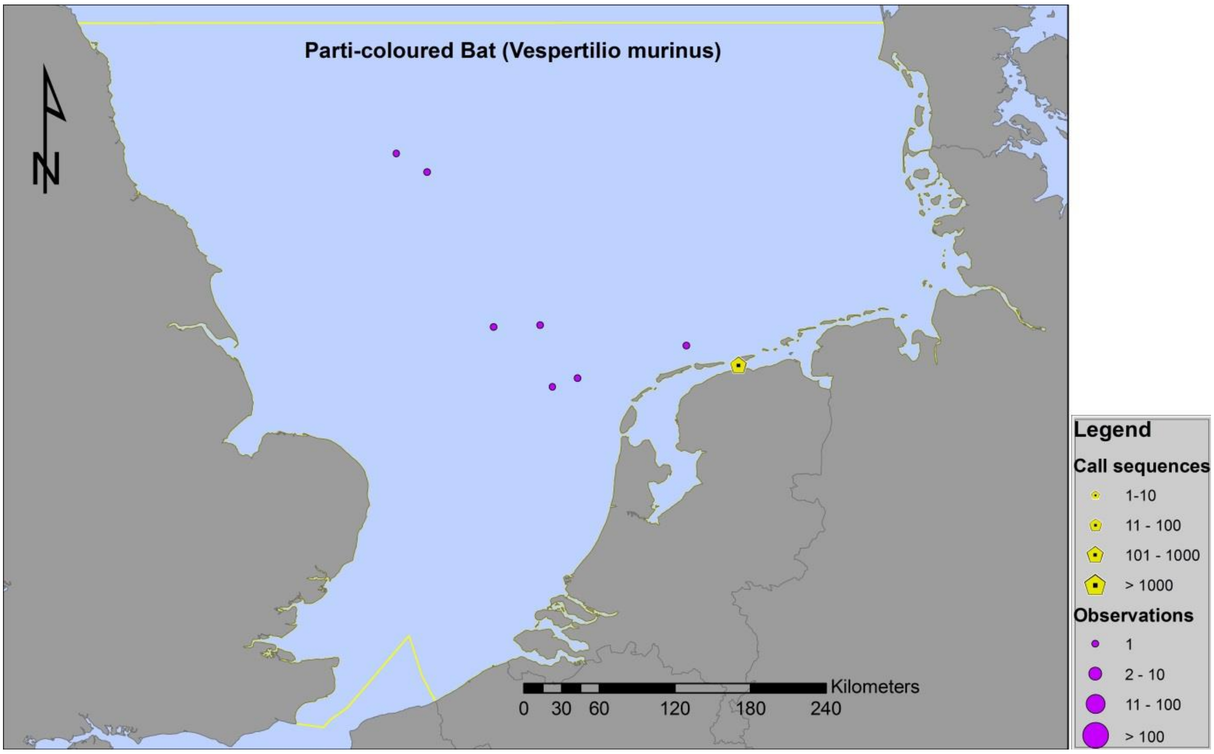
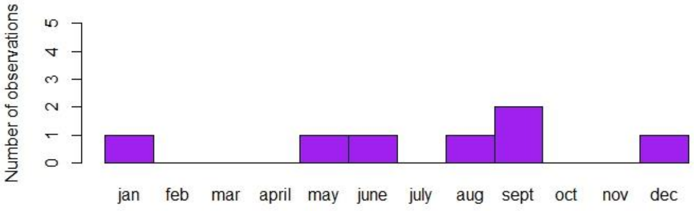


*Figure 4.59. Spatial occurrence of Parti-coloured Bat.*



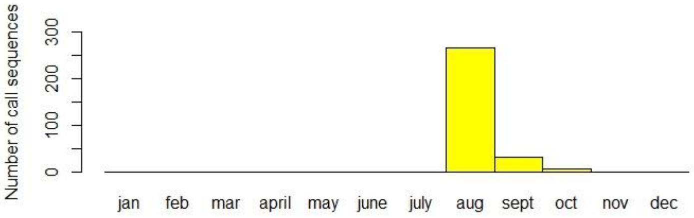


*Figure 4.60.Temporal occurrence of Parti-coloured Bat.*

Brown Long-eared Bat *Plecotus auritus*

The Brown Long-eared Bat occurs throughout Europe and is a resident species (Dietz *et al.* 2007). There are a few records from Shetland and Orkney (Petersen *et al.* 2014) and off Sweden (Ahlén *et al.* 2009), but it has apparently never been observed at the southern North Sea.

Unidentified bats

There are quite a few observations of unidentified bats at the southern North Sea and at its shores (North Sea Bird Club, www.trektellen.nl, www.waarneming.nl, Walter 2007). Most of these are described as Pipistrelle spec., and therefore likely to be Nathusius' Pipistrelle. Figures 4.61 and 4.62 show respectively their spatial and temporal occurrence. Note the peaks during the migration seasons.

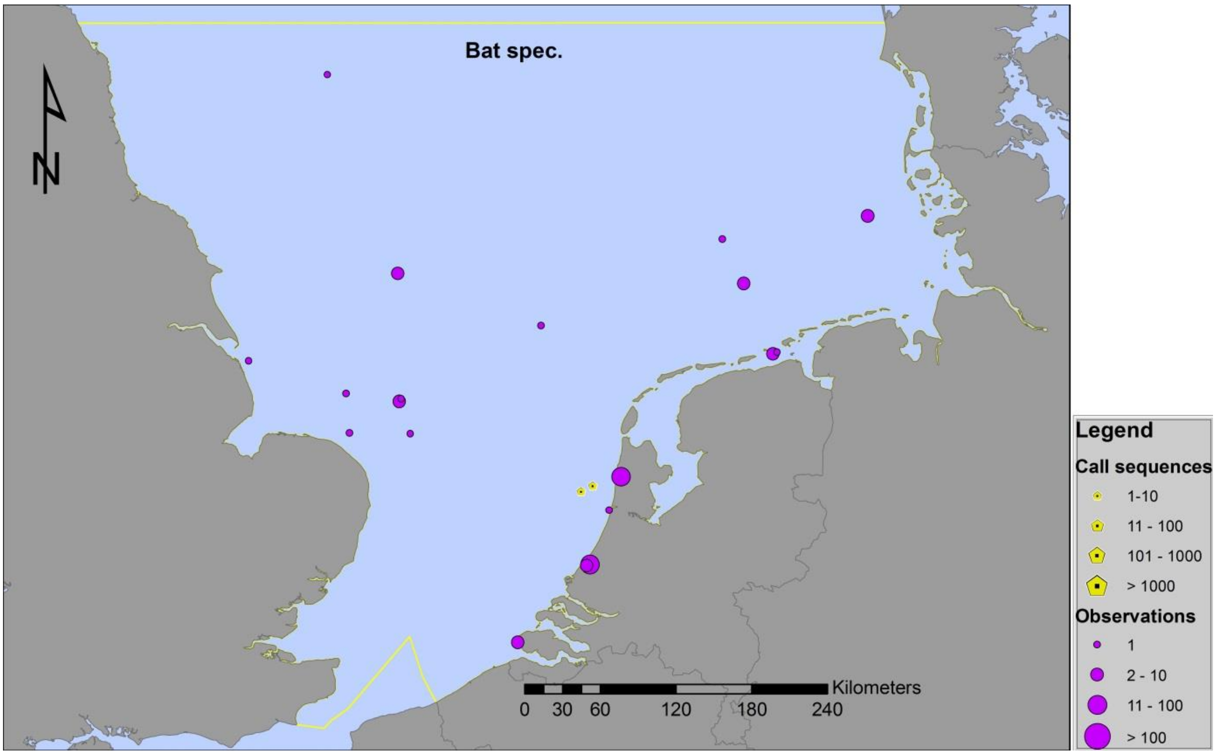


*Figure 4.61 Spatial occurrence of unidentified bats.*

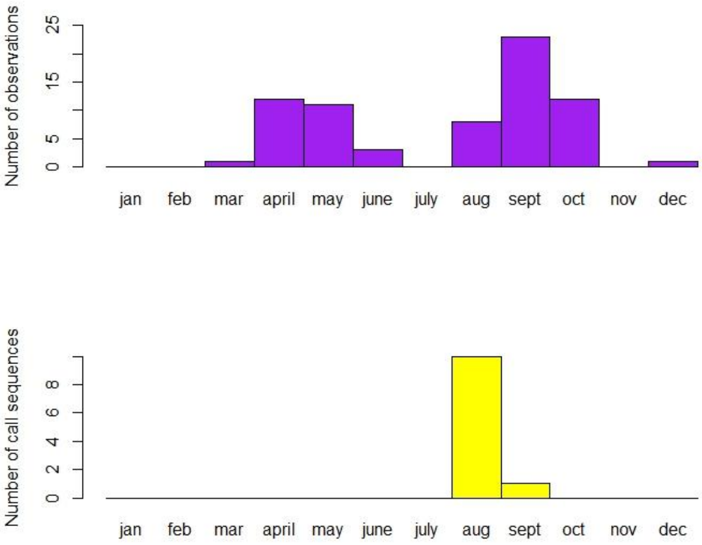


*Figure 4.62. Temporal occurrence of unidentified bats.*

Summary

The most common bat species at the southern North Sea is Nathusius' Pipistrelle. Noctule and Parti-coloured Bats also probably occur regularly at sea. Their pattern of occurrence clearly coincides with the migration seasons. Pond Bats have not been recorded offshore but might use the coastal sea as foraging area.

Common Pipistrelle, Northern Bat, Serotine Bat and Leisler's Bat have all been observed at the southern North Sea, but are likely to be just occasional visitors or vagrants.

A map of all bat sightings is shown in Figure 4.63 and their temporal occurrence in Figure 4.64. Even without genuine monitoring, observations have been made spread out over the entire (southern) North Sea. As most observations were made incidentally, these records cannot be corrected for monitoring effort.

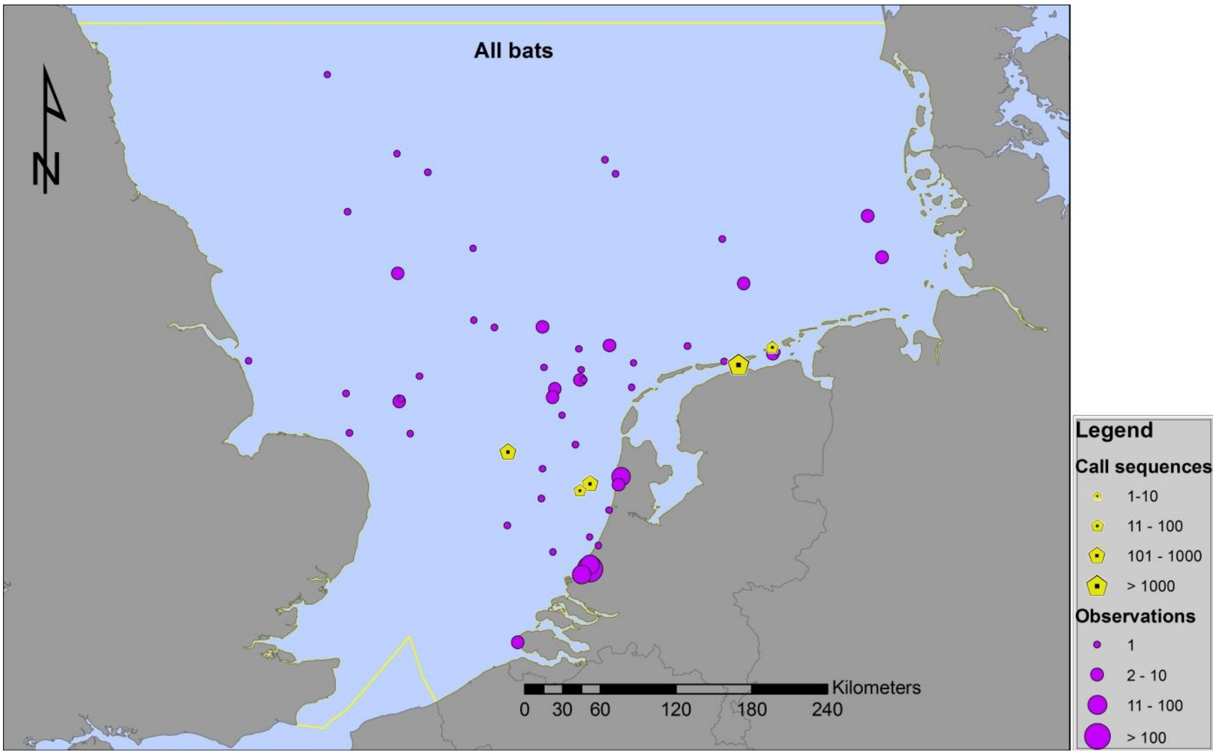


*Figure 4.63. Geographical locations of all bat observations and acoustic recordings. Note that most observations have been done on oil and gas platforms, and that these platforms are scarce in the southern part of the study area.*

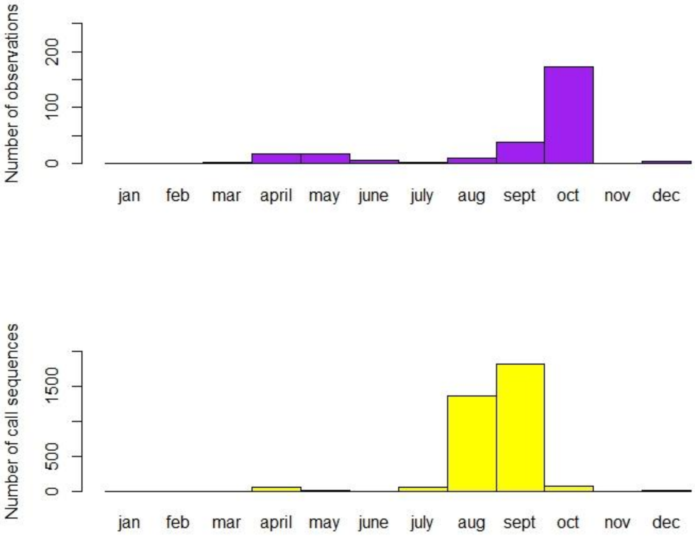


*Figure 4.64. Temporal occurrence of all bat observations and acoustic recordings.*

### 4.2.7.4 Estimation of bat fatalities

The main detrimental effect of wind energy developments on bats is increased mortality. Bat fatalities occur because of direct collisions with the rotor blades or pressure changes near them, called barotrauma (Bearwald *et al.* 2008). Disturbance does not seem to play a role, in fact bats are attracted to wind turbines (Cryan *et al.* 2014). The reason why bats are attracted to turbines is not yet fully understood. The most likely explanation is that bats are foraging on migrating insects that congregate at the top of wind turbines at the end of summer / beginning of fall (Rydell *et al.* 2010b). Most fatalities on land occur in this time of the year, also in non-migratory species. Bat species that have been reported as fatality in wind farms are mostly belonging to the genera *Pipistrellus*, *Nyctalus* and *Vespertilio*. These bats are aerial insectivorous bats. Their wing morphology and echolocation characteristics are suitable for foraging in wide open areas. Bats adapted to foraging in cluttered environments or low above the ground / water (*Myotis* and *Plecotus*) can be considered low risk species (Rydell *et al.* 2010a). With the exception of Noctules (juveniles overrepresented; Seiche *et al.* 2008; Lehnert *et al.* 2014), mortality does not seem skewed according to sex or age. Bat activity at rotor height is highest during calm (wind speed < 5m/s), warm and dry nights in August and September.

All species of bats in EU countries are strictly protected, in concurrence with their place on appendix IV of the European Habitats Directive. One way of reducing bat mortality in wind farms would be by raising the cut-in speed and reducing the speed of rotor blades during the freewheel phase.

Several factors affect the likelihood of bats to become victim of an offshore wind turbine. For the assessment of the species specific sensitivity to offshore wind turbines we assessed the parameters mentioned in Table 4.10, which were derived from Winkelman *et al.* (2008), Jones *et al.* (2009) and Rydell *et al.* (2010a).

*Table 4.10 Factors which affect the likelihood to become victim of offshore wind turbines.*

| **Symbol** | **Parameter** |
|---|---|
| a | Migration |
| b | Foraging range |
| c | Foraging habitat |
| d | Foraging strategy |
| e | Attraction |
| f | Flight height |
| g | Flight Speed |
| h | Manoeuvrability |

To determine the species-specific sensitivity (SSS) we used the following formula:

SSS = a x b x c x d x e x f x g x h

And the offshore wind turbine sensitivity index (OWTSI) was calculated based on the SSS in combination with the relative abundance (RA) at sea:

OWTSI = SSS x RA

with RA = 1 for species which have been observed occasionaly, 2 for species which might occur regularly and 3 for species which have been recorded frequently (see paragraph 8.1.7).

In the next section we describe each factor of the SSS in detail, including the classification criteria of the risk scores. For each criterion a value of 1, 2 or 3 is scored, where a high score indicates a high likelihood to become a victim of offshore wind turbines.

Migration (a)

Migratory bats may encounter offshore wind turbines during migration, while sedentary species will not cross over sea. We therefore consider the following risk levels in relation to migration:

1. Low for sedentary or short distance migrants.
2. Medium for species with seasonal migrations up to 500 km.
3. High for long-distance migrants (>500 km).

The classification of the migratory behaviour of the different species is based on Hutterer *et al.* (2005) and Krapp & Niethammer (2011).

Foraging range (b)

Bat species foraging only at short distance from their roost are not expected to fly to offshore wind farms. Species with foraging ranges over 15 km may reach offshore wind farms during foraging flights. The associated risk to foraging range is considered:

1. Low with foraging distances up to 5 km.
2. Medium with foraging ranges between 5 and 15 km
3. High with foraging ranges over 15 km

The classification of foraging range of the different bat species is based on Dietz *et al.* (2007) and Krapp & Niethammer (2011).

Foraging habitat (c)
Bats foraging in cluttered environments like forests are not expected to forage in offshore wind farms, wheareas bat species that are normally hunting in open areas are more likely to forage in offshore wind farms (Ahlén *et al.* 2009). We consider the fatality risk in relation to foraging habitat:

1. Low, when hunting in forests
2. Medium, when hunting in semi-open areas
3. High, when hunting in open areas

The classification of the foraging habitat of the different bat species is based on Limpens *et al.* (1997) , Kapteyn (1995) and Krapp & Niethammer (2011).

Foraging strategy (d)
Those bat species that are normally hunting in open areas will not specifically forage near wind turbines. Other bat species (*Pipistrellus* species) are known to fly up along the tower of the windturbine and come within reach of the blades easily. The fatality risk in relation to foraging strategy is therefore considered:

1. Low, when foraging away from (habitat) structures
2. Medium, when foraging near (habitat) structures
3. High, when foraging near habitat structures includes 'towering'

The classification of the foraging strategy of the different bat species is based on Limpens *et al.* (1997) , Kapteyn (1995) and Krapp & Niethammer (2011).

Attraction to offshore wind turbines (e)
Bats can be attracted to (offshore) wind turbines for various reasons. We consider the fatality risk:

1. Low, when attraction is absent
2. Medium, in case of limited attraction
3. High, in case of attraction, **or** when unknown

Currently there are no publications on the species specific attraction of offhore wind turbines for bats available. Therefore all bats were classified in the highest risk level. A result of this assumption is that the criterion of attraction does not contribute to differences in species specific sensitivities.

Flight height (f)
Only bats flying near turbine blades are at risk by either a collision or barotrauma. Some bats prefer to fly high while others stay low. We consider the risk in relation to flight height:

1. Low, when flying usually below 15 m
2. Medium, in case of no specific preference
3. High, when flying usually above 15 m

The classification of the flight height of the different bat species is based on Rodrigues *et al.* (2008) and Krapp & Niethammer (2011).

Flight speed (g)
The fatality risk of bats flying at high speeds is lower compared to slower flying species. We therefore consider the risk level in relation to flight speed:

1. Low, when flying at high speeds
2. Medium, when flying at moderate speeds
3. High, when flying at low speeds

The classification of the flight height of the different bat species is based on Norberg *et al.* (1987) and Baagoe (1987).

Manoeuvrability (h)
Broad-winged bats are more manoeuvrable than slender-winged bats and are better in avoiding collisions We consider the fatality risk:

1. Low, in case of a high manoeuvrability (broad-winged)
2. Medium, in case of a medium manoeuvrability
3. High, in case of a high manoeuvrability (slender-winged)

The classification of the manoeuvrability of the different bat species is based on Norberg *et al.* (1987) and Baagoe (1987).

Table 4.11 provides an overview of the species-specific risk scores per factor, as well as the overall species-specifc sensitivity.

Therefore, Nathusius' Pipistrelle, Parti-coloured Bat and Noctule are not just species that regularly occur over the southern North Sea, but also appear to be the most vulnerable species based on this risk assessment.

*Table 4.11. Species-specific risk scores for factors a – h (1=low, 2=medium, 3=high) and the overall species specific sensitivity index. The highest possible score is $3^8$= 2187 and the lowest 3 (since all species score 3 for attraction).*

| **Species** | **Migration** | **Foraging range** | **Foraging habitat** | **Foraging strategy** | **Attraction** | **Flight height** | **Flight speed** | **Manoeuvrability** | **Sensitivity index** |
|---|---|---|---|---|---|---|---|---|---|
| Daubenton's Bat *Myotis daubentonii* | 1 | 2 | 2 | 1 | 3 | 2 | 3 | 3 | 216 |
| Pond Bat *Myotis dasycneme* | 2 | 3 | 2 | 1 | 3 | 2 | 2 | 3 | 432 |
| Noctule *Nyctalus noctula* | 3 | 3 | 3 | 1 | 3 | 3 | 1 | 3 | 729 |
| Leisler's Bat *Nyctalus leisleri* | 3 | 3 | 3 | 1 | 3 | 3 | 1 | 2 | 486 |
| Common Pipistrelle *Pipistrellus pipistrellus* | 1 | 1 | 2 | 3 | 3 | 2 | 2 | 2 | 144 |
| Soprano Pipistrelle *Pipistrellus pygmaeus* | 3 | 2 | 3 | 3 | 3 | 2 | 2 | 2 | 1296 |
| Nathusius' Pipistrelle *Pipistrellus nathusii* | 3 | 2 | 3 | 3 | 3 | 2 | 2 | 2 | 1296 |
| Parti-coloured Bat *Vespertilio murinus* | 3 | 3 | 3 | 1 | 3 | 3 | 1 | 3 | 729 |
| Serotine Bat *Eptesicus serotinus* | 2 | 2 | 3 | 2 | 3 | 2 | 2 | 2 | 576 |
| Northern Bat *Eptesicus nilssonii* | 2 | 3 | 2 | 1 | 3 | 3 | 1 | 2 | 216 |
| Brown Long-eared Bat *Plecotus auritus* | 1 | 1 | 1 | 2 | 3 | 2 | 3 | 1 | 36 |

Eventually the offshore wind turbine sensitivity index (OWTSI) can be derived when combining the species-specifc sensitivity (SSS) with the relative abundance (RA) at the southern North Sea, with

- RA = 1 for species which have not been observed yet
- RA = 2 for species which occasional observations
- RA = 3 for species which have been recorded regularly, or may be expected regularly (Table 4.12)

*Table 4.12 The relative density (RA), species-specific sensitivity (SSS) and offshore wind turbine sensitivity index (OWTSI).*

| **Species** | **RA** | **SSS** | **OWTSI** |
|---|---|---|---|
| Daubenton's Bat *Myotis daubentonii* | 1 | 216 | 216 |
| Pond Bat *Myotis dasycneme* | 3 | 432 | 1296 |
| Noctule *Nyctalus noctula* | 3 | 729 | 2187 |
| Leisler's Bat *Nyctalus leisleri* | 2 | 486 | 972 |
| Common Pipistrelle *Pipistrellus pipistrellus* | 2 | 144 | 288 |
| Soprano Pipistrelle *Pipistrellus pygmaeus* | 1 | 1296 | 1296 |
| Nathusius' Pipistrelle *Pipistrellus nathusii* | 3 | 1296 | 3888 |
| Parti-coloured Bat *Vespertilio murinus* | 3 | 729 | 2187 |
| Serotine Bat *Eptesicus serotinus* | 2 | 576 | 1152 |
| Northern Bat *Eptesicus nilssonii* | 2 | 216 | 432 |
| Brown Long-eared Bat *Plecotus auritus* | 1 | 36 | 36 |

#### 4.2.7.5 Preliminary assessment of bat fatalities at sea

In this section we make a quantitative assessment of the cumulative effects of offshore wind farms in the Southern North Sea on the bats that seem to be most vulnerable: Nathusius' Pipistrelle, Parti-coloured Bat and Noctule.

To determine the potential impacts on total populations of bats in Europe we give an overview of what is known on the potential catchment area of bats occurring at the North Sea. Subsequently we estimate the population sizes in the different countries around the North Sea and further to the east, to scale the potential impacts on these populations (via preliminary Potential Biological Removal calculations).

Nathusius' Pipistrelle, Parti-coloured Bat and Noctule seem to be the most vulnerable species in the study area. They are found regularly as fatalities in onshore wind farms (Dürr 2013). These species can therefore be considered as overall risk species with regard to wind energy developments. Up until now, the number of locations where bat activity has been measured in North Sea wind farms is limited. Furthermore, bat activity as measured by bat detectors generally shows large variability. This makes it difficult to directly compare offshore wind farms with those onshore (with known fatality rates). Ahlén *et al.* (2009) reported high levels of bat activity offshore, but as their study sites were located in narrow sea channels between two bodies of land, their study is not representative for wind farms located more than 10 km from the coast in the North Sea. Bat activity at offshore wind farms measured several meters above the water surface is generally low compared to the activity at onshore wind farms near ground level. In some offshore wind farms in the North Sea, such as OWEZ, more than 100 recordings of bats were made during a single month (Jonge Poerink *et al.* 2013, Lagerveld *et al.* in prep.) In contrast to sites with only a few recordings, we cannot exclude the possibility that bat fatalities do occur here more than incidentally.

Catchment area

- Nathusius' Pipistrelle

The geographic provenance of bats over the North Sea has never been studied. In both the Netherlands and the UK, reproduction of the Nathusius' Pipistrelle is very rare. Very few maternity roosts have been recorded so far (Bat Conservation Trust 2010, Kapteyn 1995). Based on mark-recapture studies of banded Natusius' Pipistrelles, it seems likely that they originate from the Baltic States, Scandinavia, Poland and Germany (Hutterer *et al.* 2005). Recently, the provenance of a dozen Nathusius' Pipistrelles killed at wind farms in Germany was studied by using stable hydrogen isotopes in fur (Voigt *et al.* 2012): they originated from Estonia or Russia.

- Noctule

To determine the most likely origin of Noctules migrating over the North Sea, a different approach is required. Noctules are known to reproduce in many of the countries surrounding the North Sea (e.g. UK, Belgium, the Netherlands) but the populations in the UK seem largely sedentary (Jones *et al.* 2009). The same applies to the Netherlands, where most Noctules hibernate relatively close to their maternity roost (Bells 1952; Sluiter & Van Heerdt 1966). Noctules occurring far offshore may not originate from these largely sedentary populations. It seems far more likely that long-distance migrants originate from areas with colder winters, such as northeastern Europe. This is confirmed by the general southwesterly migration direction of marked Noctules in Europe during autumn (Hutterer *et al.* 2005) and stable hydrogen isotope studies (Voigt *et al.* 2012; Lehnert *et al.* 2014). The latter demonstrate the provenance of Noctules killed at onshore wind farms in eastern Germany as both local (Germany and Poland) and distant (Baltic States, Belarus and Russia). To summarise, the most likely origin of migrating Noctules in the North Sea is: the Baltic States, Belarus, Russia, Poland, Germany and possibly Scandinavia.

- Parti-coloured Bat

Parti-coloured Bats reproduce in low numbers in several countries surrounding the North Sea: the Netherlands, Denmark, Germany. In the UK this species is a vagrant with around two records a year.

Maternity roosts have not been recorded in the UK so far. In Denmark the species seems to be sedentary (Dietz *et al.* 2007). Long-distance migration is known from populations in Russia, Belarus and the Baltic States (Hutterer *et al.* 2005). These bats migrate in a southwestern direction in autumn. The origin of Parti-coloured Bats in the North Sea is possibly northeastern Europe but more data are needed to substantiate this.

Population size

Compared to birds, little is known about the size of bat populations. Estimates for population sizes are usually based on expert opinion rather than extrapolated counts. The European Topic Centre on Biological Diversity summarises these population estimates for EU countries (http://bd.eionet.europa.eu/article17/reports2012/). Both population sizes and trends are given. We used these data to determine the most likely catchment area (see the paragraphs on catchment area for the three species).

*Table 4.13. Population estimates and trend for Nathusius' Pipistrelle in EU countries that are situated in the catchment area of North Sea wind farm fatalities (source: European Topic Centre on Biological Diversity). N/A = missing value.*

| | Population estimate | Trend |
|---|---|---|
| Denmark | N/A | N/A |
| Estonia | N/A | + |
| Finland | N/A | N/A |
| Germany | N/A | = |
| Latvia | 10 000 – 50 000 | + |
| Lithuania | 40 000 – 50 000 | = |
| Poland | 5100 | N/A |
| Sweden | 3000 – 6500 | + |

*Table 4.14. Population estimates and trend for Noctule in EU countries that are situated in the catchment area of North Sea wind farm fatalities (source: European Topic Centre on Biological Diversity). N/A = missing value.*

| | Population estimate | Trend |
|---|---|---|
| Denmark | N/A | N/A |
| Estonia | N/A | + |
| Finland | N/A | N/A |
| Germany | N/A | + |
| Latvia | 5000 – 10 000 | N/A |
| Lithuania | N/A | = |
| Poland | 50 000 | = |
| Sweden | 55 000 – 95 000 | = |

*Table 4.15. Population estimates and trend for Parti-coloured Bat in EU countries that are situated in the catchment area of North Sea wind farm fatalities (source: European Topic Centre on Biological Diversity). N/A = missing value.*

| | Population estimate | Trend |
|---|---|---|
| Denmark | N/A | N/A |
| Estonia | N/A | N/A |
| Finland | N/A | N/A |
| Germany | N/A | N/A |
| Latvia | 1000 – 5000 | N/A |
| Lithuania | N/A | = |
| Poland | N/A | = |
| Sweden | 600 – 1500 | = |

The total bat population from the catchment area is much higher than the numbers presented in Tables 4.13-4.15. for several reasons: data are missing; especially from (large) non-EU countries. In particular, Russia probably has large populations of both Noctules and Nathusius' Pipistrelles because of its large territory. Population size estimates from several EU countries are also missing, namely: Denmark, Germany and Estonia. Estonia is one of the Baltic countries with large populations of Nathusius' Pipistrelle. Except for knowing that significant numbers of Noctules occur in Germany, detailed information on population sizes is missing. Furthermore, it is unknown whether data at the sub-region level are available.

Population trends

Of the five relevant countries, a population trend is reported for Nathusius' Pipistrelle (Table 4.16). For Germany a stable trend is given, whereas for Sweden and the two Baltic States a positive trend is reported. Furthermore, it is known that the range of this species is expanding (Dietz *et al.* 2007). A total of five countries reported a population trend for the Noctule (Table 4.15). In Germany and Estonia this trend is positive, whilst in Sweden, Lithuania and Poland the trend is stable. There are insufficient data available (Table 4.15) to draw conlusions about population size and trend for the Parti-coloured Bat.

Life history characteristics

Life history characteristics of Nathusius' Pipistrelle and Noctule are described by Dietz *et al.* (2007) and summarised in Table 4.16.

*Table 4.16. Life history characteristics from Nathusius Pipistrelle and Noctule. Source: Dietz et al. 2007.*

| | Nathusius' Pipistrelle | Noctule |
|---|---|---|
| Mortality of adults (per year) | 0.32 – 0.34 | 0.44 |
| Observed average age of animals that survived their first year | 2.6 – 2.9 | 2.2 – 2.3 |
| Maximum age (expected in 1% of population) | 11 - 13 | 8.5 - 9 |
| Observed maximum age | 14 | 12 |
| Age at first breeding | 1-2 | 1-2 |
| Observed birth rate (per year) | 1.8 | 1.4 – 1.5 |

These values are derived from studies in Brandenburg, Germany (Noctule; Heise & Blohm 2003) and (Nathusius' Pipistrelle; Schmidt 1994). Obviously, these characteristics can differ between populations (e.g. birth rate probably lower in outer range of distribution) or vary over time. Therefore, uncertainty exists as to whether these values can be safely applied to bats occurring in offshore wind farms in the North Sea.

Life history characteristics of the Parti-coloured Bat seem to be similar to those of the Noctule. The birth rate is relatively high because most females give birth to twins. Observed maximum age is 12 (Dietz *et al.* 2007). There is insufficient data available to complete Table 4.16 for this species.

Towards an expert opinion on the number of fatalities offshore

There are no estimates of the number of bat fatalities in offshore wind farms to date. This is not surprising, since it is virtually impossible to search and find bat carcasses at sea. Bat carcasses can sink or will be displaced from the wind turbine by the seawater currents. Brinkmann *et al.* (2011) developed a method to predict the number of bat fatalities for wind farms based on the number of bat recordings from the nacelle of wind turbines. They measured bat activity and conducted fatality searches in more than 30 onshore wind farms in Germany. Most of the studied wind farms are located in areas with potential roost sites (buildings, trees) in the vicinity. Using their model (BMU/Oikostat model) the number of fatalities in offshore wind farms could theoretically be predicted.
For the following reasons the BMU/Oikostat model could not be used to estimate the number of fatalities in the North Sea based on the acoustic data:

1. The model is based on onshore studies and it is uncertain whether it can be applied to offshore areas. A large dataset from NW Germany containing mostly migratory bat species suggests that there is no correlation between acoustic bat activity and bat mortality in the flat, open and windy areas along the North Sea (Bach *et al.* 2014). The possibility exists that due to a difference in behaviour (e.g. lesser use of echolocation or bats mostly flying above the nacelle within rotor height) a larger proportion of bats remain undetected rendering the model not suitable for migrating bats and thus for application in the North Sea.
2. Data in North Sea wind farms were measured 15 m above the water level. The BMU/Oikostat model requires data measured from the nacelle of wind turbines. Data collected at ground level have a very weak correlation with the number of fatalities and cannot be used to make accurate estimates.

For this reason we use a preliminary estimate based on expert opinion on the number of fatalities instead of using the BMU/Oikostat model. This estimate is not based on registered/observed bat fatalities in offshore wind farms since this information is currently not available. It should therefore be regarded as a best educated guess.

A total of 95.7% of the recorded offshore bat activity over the North Sea consists of Nathusius' Pipistrelle and 2.6% of Noctule (the remaining small percentage consisting of Parti-coloured Bat (1.2%) and Common Pipistrelle (0.5%), based on 417 recordings of bats at sea (Jonge Poerink *et al.* 2012, Lagerveld *et al.* in prep.). Noctules and Parti-coloured Bats use calls at a lower peak frequency than Nathusius Pipistrelle. As calls at lower frequencies are less attenuated by the atmosphere they can be detected over larger distances. When taking the detection probability of both species into account (Eurobats) the percentage of Noctules and Parti-coloured Bats are likely to be even slightly lower than respectively 2.6 % and 1.2%.

The total number of recordings in offshore wind farms in the North Sea is low compared to the number of recordings in wind farms in intensively used agricultural areas on the Dutch mainland. As stated before, the possibility exists that bat activity in offshore wind farms is systematically underestimated by bat detectors. Nonetheless, the number of fatalities in offshore wind farms is likely to be lower than on the mainland for the following reasons:

- Non-migratory bats, such as Common Pipistrelle *Pipistrellus pipistrellus*, are virtually absent offshore. Onshore, Common Pipistrelle is one of the most common species.
- Only a small proportion of Noctules in onshore wind farms are long distance migrants (Lehnert *et al.* 2014). In offshore wind farms most bats are likely to be long distance migrants.

- Bat activity offshore is generally limited to periods with calm weather that is suitable for long distance migration. Onshore, bats are recorded during a wider range of weather conditions (e.g. higher wind speeds and from various directions).
- Outside the migration period, bat fatalities occur at onshore wind farms (although in relatively low numbers). At offshore wind farms, nearly all activity is limited to the migration period.

Wind farms in large, open intensively used agricultural areas show the lowest number of fatalities onshore. The number is typically around 1 fatality per turbine per year (Rydell *et al.* 2010a; Limpens *et al.* 2013). Based on the current knowledge, a rough estimate for the number of fatalities in offshore wind farms is somewhere between 0 and 1 fatalities per turbine per year.

Assuming a total of 1 victim per turbine per year, based on the number of 8 000 turbines to be installed in the Southern North Sea in the near future, we logically arrive at a maximum of 8 000 potential fatalities. A worst case scenario would be to use this number for each species, but based on the available information on recorded bat activity in the North Sea it is more realistic to divide this number based on the relative abundance of the three species, thus arriving at estimates of 7700 (95.7%) fatalities on a yearly basis for Nathusius' Pipistrelle, 200 (2.6%) for Noctule and almost 100 (1.2%) for Parti-coloured Bat.

Potential Biological Removal for bat collisions at sea

From the viewpoint that part of a biogeographical population will fly over the North Sea we can ask the question: At what level (expressed by number of victims, increased mortality) is the effect on a bat population unacceptable?

To answer this question, we used the Potential Biological Removal (PBR) approach followed by Lebreton (2005), Niel & Lebreton (2005) and Dillingham & Fletcher (2008). It is a calculation based on a species-specific maximum population growth rate and a minimum population estimate, to calculate the total number of victims possible without the population becoming at risk. With regard to bats at sea we applied this method only on Nathusius' Pipistrelle and Noctule, as there is insufficient data available for the Parti-coloured Bat. The PBR approach is further explained in section 4.3.3.

Bellebaum & Wendeln (2011) repeated the simulations of Wade (1998) to determine the appropriate rf to use for species with growth rates higher than those of marine mammals (generally <1.2), like for instance small birds and also bats. They found that for growth rates ($\lambda_{max}$) between 1.4 and 2.4 the recommended rf for stable populations would be 0.2 and 0.1 for threatened and/or declining populations (precautionary). As we strive to calculate conservative values for the PBR we therefore applied the rf value of 0.2 here for bats, following Bellebaum & Wendeln (2011) for other short-lived species (average lifespan of 4 years or shorter), which generally have a maximum annual population growth rate ($\lambda_{max}$) higher than 1.4.

*Table 4.17. PBR- Potential Biological Removal level for two bat species for assumed catchment populations occurring in the Dutch part of the North Sea. Both species have a stable or increasing population trend, so a recovery PBR factor of rf = 0.5 would apply and indicated with light purple. Here we have indicated with dark purple the PBR of populations of least concern with unstable or decreasing population trend (recovery PBR factor rf = 0.2).* $R_{max}$ *calculated based on parameters in Table 4.10.*

| Species | Nathusius' Pipistrelle | Noctule Bat |
|---|---|---|
| Region | Population estimate for Latvia, Lithuania, Poland and Sweden | Population estimate for Latvia, Poland and Sweden |
| Population estimate Nmin individuals | 58100 | 110000 |
| Rmax | 0,33 | 0,37 |
| PBR rf=0.1 | 953 | 2045 |
| PBR rf=0.2 | 1905 | 4089 |
| PBR rf=0.5 | 4763 | 10224 |
| Remarks | Population is increasing | Population seems to be stable or increasing |

Just adding up only the known population sizes for each species of bat (arriving at an incomplete total of the biogeographical population), and the species-specific life history characteristics and assuming stable or positive trends in both species of bats (taking the factor rf for the PBR calculation as 0.2 or 0.5), the first preliminary calculations show that for Nathusius’ Pipistrelles the calculated number of collision victims is well above the limit of the Potential Biological Removal level with rf = 0.2 (400%), and also above the limit of the Potential Biological Removal level with rf = 0.5 (7,700 fatalities on a yearly basis is 160% of this latter PBR value). It is clear that the incompleteness of population data prevents a proper evaluation of the number of estimated fatalities according to a worst case scenario (assumed catchment population now consisting of Latvia, Lithuania, Poland and Sweden). For Noctule the estimated number of 200 collision victims according to a worst case scenario is well within the limit of the Potential Biological Removal level for the assumed catchment population of Latvia, Poland and Sweden (with 5% of the PBR level with Rf = 0.2). In case we would regard the species as 'near-threatened', the calculated number of 200 potential collision victims based on that status would still be within the PBR value for this species for the assumed population (10% of that level).

Because for Nathusius’ Pipistrelles the number of collision victims in a worst case scenario is above the limit of the Potential Biological Removal level, we also calculated the population size for which 7700 fatalities, on a yearly basis, would be on the level of the PBR value. With an rf = 0.2 this population size is 235 000 animals and with an rf = 0.5 this is 98 000 animals.

Our main conclusion is that, due to a lack of accurate information on the exact locations and sizes of the catchment populations, it is not possible to rule out that a worst case scenario of fatalities in both species of bats will have serious impacts on total populations. Based on the assumptions made, the least important impact is to be expected for Noctule. The knowledge gaps in Nathusius’ Pipistelle regarding the size of the catchment populations prevent any reliable evaluation of the number of estimated worst case scenario.

## 4.3 Methods to assess seabird species sensitivity to OWF

The method used for assessing seabird sensitivity to collisions with wind turbines and displacement from wind farm areas largely follows the method of Bradbury *et al.* (2014). First, we will explain the method proposed by Bradbury *et al.* (2014) and second, the method for the current cumulation project will be described showing the resemblance to and deviations from the method of Bradbury *et al.* (2014).

### 4.3.1 Bradbury method

Bradbury *et al.* (2014) used four factors that are related to conservation and six factors that are directly related to species vulnerability, that together determine species sensitivity to offshore wind farms (see Table 4.18). The scoring criteria for each factor and the scores for each marine bird species were based on evidence and a large body of reviewed literature.

*Table 4.18. Parameters used by Bradbury et al. (2014) for seabird sensitivity for offshore windfarms.*

| Symbol | Parameter | Element of | Relevance for the current method |
|---|---|---|---|
| a | Score for highest percentage of biogeographic population in England in any season | Conservation importance | No |
| b | Adult survival rate | Conservation importance | No # |
| c | UK threat status score | Conservation importance | No |
| d | Bird Directive score | Conservation importance | No |
| e | Estimated percentage at blade height | Collision | Yes |
| f | Flight manoeuvrability | Collision | Yes |
| g | Percentage of time spent flying | Collision | Yes |
| h | Nocturnal activity | Collision | Yes |
| i | Disturbance susceptibility | Displacement | Yes |
| j | Habitat specialization | Displacement | Yes |

# adult survival rate is part of the PBR models

Most factors were scored on a scale of 1 to 5, with the conservation importance or anticipated negative impact increasing with increasing number on the scale. Only estimated flight at blade height was scored as a percentage (1 - 100) instead of a number on the 1–5 scale. Further details of the scoring in the Bradbury *et al.* (2014) paper are given in the Annex E of this report.

Bradbury *et al.* (2014) scored separately for collision and for displacement risks. For collision risk, they gave a high weighting to flight altitude (e), and lower weightings to manoeuvrability (f), percentage of time flying (g), and nocturnal flight activity (h): (Equation 1).

$$\text{Collision risk score} = (e \bullet (f + g + h)) / 3 \bullet (a + b + c + d) \qquad (1)$$

For displacement they proposed a vulnerability index according to equation 2 where i represents disturbance by wind farm structures, ship and helicopter traffic, and j the impact of habitat specialisation respectively. They do combine these two measures only in a semi-quantitative way with conservation parameters (Table 4.18; equation 2). Bradbury et al. do not use the absolute outcomes of these calculations in a quantitative way, but use these only to rank the vulnerabilities of the various species.

$$\text{Displacement score} = ((i \times j) \bullet (a + b + c + d)) / 10 \qquad (2)$$

Application of an extra factor of 0.10 (which results from the factor 10 in the denominator) in the equation for displacement (equation 2) means that at maximum 10% of the displaced birds may die. This is an arbitrary choice, made by Bradbury *et al.* (2014), and a better guess is not possible at the moment.

For species sensitivity mapping the sensitivity to wind farm collision and displacement, Bradbury *et al.* (2014) applied scores to a function of the density of those species in each 3 km × 3 km grid cell across their study area (English territorial waters). The natural logarithm of the density has been used as this enabled better scaling for comparison between species and areas. So for each species' sensitivity to wind farm impacts, the expression took the form of equations (3), (4) and (5):

$$\text{Windfarm Sensitivity Index collision} = LN(\text{density species} + 1) \times SSIcollision \qquad (3)$$

$$\text{Windfarm Sensitivity Index displacement} = LN(\text{density species} + 1) \times SSIdisplacement \qquad (4)$$

$$\text{Windfarm Sensitivity Index windfarm} = \text{LN}(\text{density species} + 1) \times \text{SSImax (collision, displacement)} \qquad (5)$$

Where SSI is the Species Sensitivity Index to either wind farm collision or displacement.

Bradbury *et al.* (2014) derived overall sensitivity to wind farms in each 3 km × 3 km grid cell by using the highest values for either collision or displacement risk. Scores of 5 were assigned to those considered to be 'Very High Risk', 4 to 'High Risk', 3 to 'Moderate Risk', 2 to 'Low Risk' and 1 to 'Very Low Risk'. They chose not to assign the top rank 'Very High Risk' for displacement concern in order to acknowledge the lower risk to populations compared to collision risks. This is an arbitrary choice.

### *4.3.2 CUMULEO-framework*

We adjusted the formulas of Bradbury *et al.* (2014) in order to calculate absolute numbers of potential bird victims of wind farms (extended-Bradbury method). The differences of our method compared to the method by Bradbury are:

- omitting the conservation importance scores (factors a-d), as these do not impact actual numbers of casualties;
- scaling the relative risk factor for collision between 0 and 1 in which the maximum score of 1 is assigned to a species with the theoretical maximum score for all parameters resulting in 100% mortality. The underlying maximum score is 1500 for collision
- scaling the relative risk factor for displacement between 0 and 0.1 in which the maximum score of 0.1 is assigned to a species with the theoretical maximum score for all parameters resulting in 10% mortality. The underlying maximum score is 25 for displacement. As a worst case approach the factor 0.10 can be omitted and the risk factor for displacement is scaled between 0 and 1 assuming mortality of all displaced birds (100%). This worst case calculation will be carried out next too and in case the PBR criterion is exceeded, implications will be discussed;
- combine relative collision risk score and relative displacement risk score for each species in a relative species risk score;
- calculating absolute risk scores for species by using the density instead of the natural logarithm of the density.

This resulted in the following set of formula for species sensitivity: equations (6) and (7) and (8) and for windfarm sensitivity mapping: equations (9), (10) and (11):

1. Species sensitivity

For collision

$$\text{Relative collision risk score} = (e \bullet (f + g + h)) / 1500 \qquad (6)$$

For displacement

$$\text{Relative displacement risk score} = (i \bullet j) / 250 \qquad (7)$$

For the combination of collision and displacement

$$\text{Relative species risk score} = \text{Relative collision score} + \text{relative displacement score} - \text{Relative collision score} \times \text{relative displacement score} \qquad (8)$$

Note that in this last formula a correction for the "double effect" is applied.

2. Windfarm sensitivity mapping

$$\text{Absolute collision risk score} = \text{Density species} \times \text{Relative collision risk score} \qquad (9)$$

$$\text{Absolute displacement risk score} = \text{Density species} \times \text{Relative displacement risk score} \qquad (10)$$

$$\text{Absolute windfarm risk score} = \text{Density species} \times \text{Relative species risk score} \qquad (11)$$

For the selected seabird species, the scores for parameters e to j are taken from Bradbury *et al.* (2014) and listed in Table 4.19. We decided to adjust the score for the parameter i: disturbance susceptibility for the great cormorant from 4 (high) to 1 (very low) based on own observations. This was only done with regard to offshore wind farms and not with regard to shipping. Cormorant are not displaced from offshore windfarms (Lindeboom *et al.* 2011; Leopold *et al.* 2013a), but do react to shipping at quite long distances, by flying off.

Furthermore, we added 3 bird groups; small loon spec., large gull spec. and "commic tern". The scores are averages of the scores of the separate species within these groups, but without the very rare species. The small loon spec. score is based on Red-throated Loon that was far more abundant in the study area than Black-throated Loon. The gull spec. score is based on all included gull species. The commic tern score is based on Common and Arctic Tern scores.

The results for the relative collision risk, relative displacement risk and the total risk are calculated with the formulas listed above and shown in Tables 4.21, 4.22 and 4.23, respectively.

It should be noted that each seabird species is considered sensitive to displacement, although some have very low scores (see Table 4.21). Zero scores for displacement are not possible due to the method used, because the lowest scores of the two parameters involved (i and j) are both 1. On the other hand zero sensitivity for collision is found for three seabird species, all shearwater species. This is methodologically possible because the parameter: "estimated percentage at blade height" (e) can be zero.

For the displacement sensitivity of seabird species to shipping, the same approach and scores were used as for offshore wind farms. This applies only to two parameters (i and j) for disturbance susceptibility and habitat specialization. No collision mortality was assumed from shipping.

*Table 4.20. Scores used in assessing sensitivity of seabird species to collision and displacement from offshore wind farms. Data from Bradbury et al. (2014). The symbols e - j are explained in Table 4.18.*

| **ESAS code** | **Name EN** | **Name NL** | **Scientific Name** | **e** | **f** | **g** | **h** | **i** | **j** |
|---|---|---|---|---|---|---|---|---|---|
| 20 | Red-throated Loon | Roodkeelduiker | *Gavia stellata* | 5 | 5 | 2 | 1 | 5 | 4 |
| 30 | Black-throated Loon | Parelduiker | *Gavia arctica* | 5 | 5 | 3 | 1 | 5 | 4 |
| 50 | White-billed Loon | Geelsnalvelduiker | *Gavia adamsii* | 5 | 5 | 2 | 1 | 5 | 4 |
| 90 | Great Crested Grebe | Fuut | *Podiceps cristatus* | 2 | 4 | 3 | 2 | 3 | 4 |
| 220 | Northern Fulmar | Noordse Stormvogel | *Fulmarus glacialis* | 1 | 3 | 2 | 4 | 1 | 1 |
| 430 | Sooty Shearwater | Grauwe Pijlstormvogel | *Puffinus griseus* | 0 | 3 | 3 | 3 | 1 | 1 |
| 460 | Manx Shearwater | Noordse Pijlstormvogel | *Puffinus puffinus* | 0 | 3 | 3 | 3 | 1 | 1 |
| 462 | Balearic Shearwater | Vale Pijlstormvogel | *Puffinus mauretanicus* | 0 | 3 | 3 | 3 | 1 | 1 |
| 520 | European Storm-petrel | Stormvogeltje | *Hydrobates pelagicus* | 2 | 1 | 3 | 4 | 1 | 1 |
| 550 | Leach's Storm-petrel | Vaal Stormvogeltje | *Oceanodroma leucorhoa* | 2 | 1 | 3 | 4 | 1 | 1 |
| 710 | Northern Gannet | Jan van Gent | *Morus bassanus* | 12 | 3 | 3 | 2 | 2 | 1 |
| 720 | Great Cormorant | Aalscholver | *Phalacrocorax carbo* | 8 | 4 | 2 | 1 | 1 | 3 |
| 800 | European Shag | Kuifaalscholver | *Phalacrocorax aristotelis* | 8 | 3 | 2 | 1 | 3 | 3 |
| 2060 | Common Eider | Eidereend | *Somateria mollissima* | 2 | 4 | 2 | 3 | 3 | 4 |
| 2130 | Common Scoter | Zwarte Zee-eend | *Melanitta nigra* | 3 | 3 | 2 | 3 | 5 | 4 |
| 2150 | Velvet Scoter | Grote Zee-eend | *Melanitta fusca* | 3 | 3 | 2 | 3 | 5 | 3 |
| 5670 | Parasitic Jaeger | Kleine Jager | *Stercorarius parasiticus* | 10 | 1 | 5 | 1 | 1 | 2 |
| 5690 | Great Skua | Grote Jager | *Stercorarius skua* | 10 | 1 | 4 | 1 | 1 | 2 |
| 5780 | Little Gull | Dwergmeeuw | *Larus minutus* | 15 | 1 | 3 | 2 | 1 | 3 |
| 5820 | Black-headed Gull | Kokmeeuw | *Larus ridibundus* | 20 | 1 | 1 | 2 | 2 | 2 |
| 5900 | Mew Gull | Stormmeeuw | *Larus canus* | 25 | 1 | 2 | 3 | 2 | 2 |
| 5910 | Lesser Black-backed Gull | Kleine Mantelmeeuw | *Larus fuscus* | 30 | 1 | 2 | 3 | 2 | 1 |
| 5920 | European Herring Gull | Zilvermeeuw | *Larus argentatus* | 35 | 2 | 2 | 3 | 2 | 1 |
| 6000 | Great Black-backed Gull | Grote Mantelmeeuw | *Larus marinus* | 35 | 2 | 2 | 3 | 2 | 2 |
| 6020 | Black-legged Kittiwake | Drieteenmeeuw | *Rissa tridactyla* | 15 | 1 | 3 | 3 | 2 | 2 |
| 6110 | Sandwich Tern | Grote Stern | *Sterna sandvicensis* | 10 | 1 | 5 | 1 | 2 | 3 |
| 6140 | Roseate Tern | Dougalls Stern | *Sterna dougallii* | 8 | 1 | 5 | 1 | 2 | 3 |
| 6150 | Common Tern | Visdief | *Sterna hirundo* | 10 | 1 | 5 | 1 | 2 | 3 |
| 6160 | Arctic Tern | Noordse Stern | *Sterna paradisaea* | 5 | 1 | 5 | 1 | 2 | 3 |
| 6240 | Little Tern | Dwergstern | *Sterna albifrons* | 10 | 1 | 5 | 1 | 2 | 4 |
| 6340 | Common Murre | Zeekoet | *Uria aalge* | 1 | 4 | 1 | 2 | 3 | 3 |
| 6360 | Razorbill | Alk | *Alca torda* | 0.5 | 4 | 1 | 1 | 3 | 3 |
| 6380 | Black Guillemot | Zwarte Zeekoet | *Cepphus grylle* | 0.5 | 4 | 1 | 1 | 3 | 4 |
| 6470 | Little Auk | Kleine Alk | *Alle alle* | 0.5 | 3 | 1 | 1 | 2 | 2 |
| 6540 | Atlantic Puffin | Papegaaiduiker | *Fratercula arctica* | 0.5 | 3 | 1 | 1 | 2 | 3 |
| 6270 | Black tern | Zwarte stern | *Chilidonias niger* | 10 | 1 | 4 | 1 | 2 | 3 |
| 5660 | Pomarine Skua | Middelste Jager | *Stercorarius pomarinus* | 10 | 1 | 5 | 1 | 1 | 2 |
| 5670 | Long-tailed Jaeger | Kleine Jager | *Stercorarius longicaudus* | 10 | 1 | 5 | 1 | 1 | 2 |
| 59 | Loon spec. | Duiker spec. | *Gavia* spec. | 5 | 5 | 2 | 1 | 5 | 4 |
| NA | Large gull spec. | Grote meeuw spec. | *Larus* spec. | 31 | 2 | 2 | 3 | 2 | 2 |
| NA | Commic tern spec. | Stern spec. (”Noordse dief”) | *Sterna* spec. | 8 | 1 | 5 | 1 | 2 | 3 |

*Table 4.20. Collision risk scores for seabird species. Results of calculations with the data from Table 4.19.*

| **Vernicular name** | **Scientific name** | **Relative Collision Risk Score** |
|---|---|---|
| European Herring Gull | *Larus argentatus* | 0.163 |
| Great Black-backed Gull | *Larus marinus* | 0.163 |
| Large gull spec. | *Larus* spec. | 0.145 |
| Lesser Black-backed Gull | *Larus fuscus* | 0.120 |
| Mew Gull | *Larus canus* | 0.100 |
| Black-legged Kittiwake | *Rissa tridactyla* | 0.070 |
| Northern Gannet | *Morus bassanus* | 0.064 |
| Little Gull | *Larus minutus* | 0.060 |
| Black-headed Gull | *Larus ridibundus* | 0.053 |
| Parasitic Jaeger | *Stercorarius parasiticus* | 0.047 |
| Sandwich Tern | *Sterna sandvicensis* | 0.047 |
| Common Tern | *Sterna hirundo* | 0.047 |
| Little Tern | *Sterna albifrons* | 0.047 |
| Pomarine Skua | *Stercorarius pomarinus* | 0.047 |
| Long-tailed Jaeger | *Stercorarius longicaudus* | 0.047 |
| Great Skua | *Stercorarius skua* | 0.040 |
| Black tern | *Chilidonias niger* | 0.040 |
| Great Cormorant | *Phalacrocorax carbo* | 0.037 |
| Roseate Tern | *Sterna dougallii* | 0.037 |
| Commic tern spec. | *Sterna* spec. | 0.037 |
| European Shag | *Phalacrocorax aristotelis* | 0.032 |
| Black-throated Loon | *Gavia arctica* | 0.030 |
| Red-throated Loon | *Gavia stellata* | 0.027 |
| White-billed Loon | *Gavia adamsii* | 0.027 |
| Loon spec. | *Gavia* spec. | 0.027 |
| Arctic Tern | *Sterna paradisaea* | 0.023 |
| Common Scoter | *Melanitta nigra* | 0.016 |
| Velvet Scoter | *Melanitta fusca* | 0.016 |
| Great Crested Grebe | *Podiceps cristatus* | 0.012 |
| Common Eider | *Somateria mollissima* | 0.012 |
| European Storm-petrel | *Hydrobates pelagicus* | 0.011 |
| Leach's Storm-petrel | *Oceanodroma leucorhoa* | 0.011 |
| Northern Fulmar | *Fulmarus glacialis* | 0.006 |
| Common Murre | *Uria aalge* | 0.005 |
| Razorbill | *Alca torda* | 0.002 |
| Black Guillemot | *Cepphus grylle* | 0.002 |
| Little Auk | *Alle alle* | 0.002 |
| Atlantic Puffin | *Fratercula arctica* | 0.002 |
| Sooty Shearwater | *Puffinus griseus* | 0 000 |
| Manx Shearwater | *Puffinus puffinus* | 0 000 |
| Balearic Shearwater | *Puffinus mauretanicus* | 0 000 |

*Table 4.21. Relative displacement risk scores for seabird species. Results of calculations with the data from Table 4.19.*

| **Vernicular name** | **Scientific name** | **Relative Displacement Risk Score** |
|---|---|---|
| Black-throated Loon | *Gavia arctica* | 0.080 |
| Red-throated Loon | *Gavia stellata* | 0.080 |
| White-billed Loon | *Gavia adamsii* | 0.080 |
| Loon spec. | *Gavia* spec. | 0.080 |
| Common Scoter | *Melanitta nigra* | 0.080 |
| Velvet Scoter | *Melanitta fusca* | 0.060 |
| Great Crested Grebe | *Podiceps cristatus* | 0.048 |
| Common Eider | *Somateria mollissima* | 0.048 |
| Black Guillemot | *Cepphus grylle* | 0.048 |
| European Shag | *Phalacrocorax aristotelis* | 0.036 |
| Common Murre | *Uria aalge* | 0.036 |
| Razorbill | *Alca torda* | 0.036 |
| Little Tern | *Sterna albifrons* | 0.032 |
| Sandwich Tern | *Sterna sandvicensis* | 0.024 |
| Common Tern | *Sterna hirundo* | 0.024 |
| Black tern | *Chilidonias niger* | 0.024 |
| Roseate Tern | *Sterna dougallii* | 0.024 |
| Commic tern spec. | *Sterna* spec. | 0.024 |
| Arctic Tern | *Sterna paradisaea* | 0.024 |
| Atlantic Puffin | *Fratercula arctica* | 0.024 |
| Great Black-backed Gull | *Larus marinus* | 0.016 |
| Large gull spec. | *Larus* spec. | 0.016 |
| Mew Gull | *Larus canus* | 0.016 |
| Black-legged Kittiwake | *Rissa tridactyla* | 0.016 |
| Black-headed Gull | *Larus ridibundus* | 0.016 |
| Little Auk | *Alle alle* | 0.016 |
| Little Gull | *Larus minutus* | 0.012 |
| Great Cormorant | *Phalacrocorax carbo* | 0.012 |
| European Herring Gull | *Larus argentatus* | 0.008 |
| Lesser Black-backed Gull | *Larus fuscus* | 0.008 |
| Northern Gannet | *Morus bassanus* | 0.008 |
| Parasitic Jaeger | *Stercorarius parasiticus* | 0.008 |
| Pomarine Skua | *Stercorarius pomarinus* | 0.008 |
| Long-tailed Jaeger | *Stercorarius longicaudus* | 0.008 |
| Great Skua | *Stercorarius skua* | 0.008 |
| European Storm-petrel | *Hydrobates pelagicus* | 0.004 |
| Leach's Storm-petrel | *Oceanodroma leucorhoa* | 0.004 |
| Northern Fulmar | *Fulmarus glacialis* | 0.004 |
| Sooty Shearwater | *Puffinus griseus* | 0.004 |
| Manx Shearwater | *Puffinus puffinus* | 0.004 |
| Balearic Shearwater | *Puffinus mauretanicus* | 0.004 |

*Table 4.22. Relative species risk scores for seabird species for offshore wind farms. Results of calculations with the data from Table 4.19.*

| **Vernicular name** | **Scientific name** | **Relative OWF Risk** |
|---|---|---|
| Great Black-backed Gull | *Larus marinus* | 0.177 |
| European Herring Gull | *Larus argentatus* | 0.170 |
| Large gull spec. | *Larus* spec. | 0.158 |
| Lesser Black-backed Gull | *Larus fuscus* | 0.127 |
| Mew Gull | *Larus canus* | 0.114 |
| Black-throated Loon | *Gavia arctica* | 0.108 |
| Red-throated Loon | *Gavia stellata* | 0.105 |
| White-billed Loon | *Gavia adamsii* | 0.105 |
| Loon spec. | *Gavia* spec. | 0.105 |
| Common Scoter | *Melanitta nigra* | 0.095 |
| Black-legged Kittiwake | *Rissa tridactyla* | 0.085 |
| Little Tern | *Sterna albifrons* | 0.077 |
| Velvet Scoter | *Melanitta fusca* | 0.075 |
| Northern Gannet | *Morus bassanus* | 0.071 |
| Little Gull | *Larus minutus* | 0.071 |
| Sandwich Tern | *Sterna sandvicensis* | 0.070 |
| Common Tern | *Sterna hirundo* | 0.070 |
| Black-headed Gull | *Larus ridibundus* | 0.068 |
| European Shag | *Phalacrocorax aristotelis* | 0.067 |
| Black tern | *Chilidonias niger* | 0.063 |
| Roseate Tern | *Sterna dougallii* | 0.060 |
| Commic tern spec. | *Sterna* spec. | 0.060 |
| Great Crested Grebe | *Podiceps cristatus* | 0.059 |
| Common Eider | *Somateria mollissima* | 0.059 |
| Parasitic Jaeger | *Stercorarius parasiticus* | 0.054 |
| Pomarine Skua | *Stercorarius pomarinus* | 0.054 |
| Long-tailed Jaeger | *Stercorarius longicaudus* | 0.054 |
| Black Guillemot | *Cepphus grylle* | 0.050 |
| Great Cormorant | *Phalacrocorax carbo* | 0.049 |
| Great Skua | *Stercorarius skua* | 0.048 |
| Arctic Tern | *Sterna paradisaea* | 0.047 |
| Common Murre | *Uria aalge* | 0.040 |
| Razorbill | *Alca torda* | 0.038 |
| Atlantic Puffin | *Fratercula arctica* | 0.026 |
| Little Auk | *Alle alle* | 0.018 |
| European Storm-petrel | *Hydrobates pelagicus* | 0.015 |
| Leach's Storm-petrel | *Oceanodroma leucorhoa* | 0.015 |
| Northern Fulmar | *Fulmarus glacialis* | 0.010 |
| Sooty Shearwater | *Puffinus griseus* | 0.004 |
| Manx Shearwater | *Puffinus puffinus* | 0.004 |
| Balearic Shearwater | *Puffinus mauretanicus* | 0.004 |

## Calculation of the risk of OWF for birds

The risk of offshore wind farms (OWF) for birds will be presented in Tables and in GIS maps. The potential number of birds exposed to future OWF will be combined with the relative sensitivity of birds for OWF to produce the potential number of birds dying due to collision, displacement and the combination of both using the formulas listed previously. The results can be presented in different ways: per species, per specific offshore windfarm, all offshore wind farms together, for shipping, per pressure type (collision, displacement), along the time line of expected realisation of the OWF.

Specific attention will be paid to the bird species with an estimated mortality exceeding the Potential Biological Removal (PBR). A comparison will be made with the alternative assessment methods that are also used for validation purposes. Furthermore, a more in-depth analysis may be carried out, targeted at options for mitigation including optimizing the choice of OWF locations, the order of OWF development in time, but no data on OWF configurations are presently available . Note that future monitoring and research may generate more reliable information which may reveal that actual risks are substantially different than estimated in the current study. This is also why the order of development of the various wind farms might be important: if wind farms that presumably will cause high mortality would be developed last, there may be time to develop mitigational measures or even to decide not te develop the last, most critical wind famrs at all.

## Discussion

Bradbury *et al.* (2014) have produced SeaMaST, a GIS tool that will inform current and future impact assessment and marine spatial planning in England, as well as providing a framework for mapping sensitivity in other geographic areas where there is high demand for wind farms. In addition, an updated compilation of seabird sensitivity scores in relation to the potential impacts of turbine collision and displacement and disturbance from wind farms has been made to reflect current knowledge. In this report, we have taken up the challenge to address areas other than (just) English waters and extended the Bradbury method to arrive at estimated mortalities.

Bradbury *et al.*'s method of wind farm sensitivity scoring used was built on previous peer-reviewed approaches, especially those of Garthe & Hüppop (2004) and Furness *et al.* (2013), to produce species rankings and relative scores separately for collision and displacement impacts. Garthe & Hüppop (2004) combined both collision risk and habitat loss considerations into a single score. On the other hand Furness *et al.* (2013) and Bradbury *et al.* (2014) chose to separate collision and displacement concerns. In the current study we do both: a separate estimation of collision and displacement, but also a combination. For that purpose we applied a correction for the overlap in effect. Birds that die from collision cannot be affected by displacement, and *vice versa*.

Bradbury *et al.* (2014) state that impacts on populations through displacement (disturbance) are poorly understood. They did not include macro avoidance due to insufficient data. Some bird species may be attracted by wind farms with a potentially increased collision risk, while other species may avoid wind farm with a suppressed collision risk. Currently research on macro avoidance is running and when information becomes available it may be possible to include relative macro-avoidance rates in future species sensitivity scoring.

For displacement we calculated a vulnerability index according to Equation (3) where i and j represent disturbance by wind farm structures, ship and helicopter traffic, and habitat specialisation respectively. Furness *et al.* (2013) divided the outcome by 10 (an arbitrary value) to recognise that the displacement impact of seabirds is likely to have a considerably lower effect on survival than direct mortality from collisions. Therefore, they suggested that the two scales should not be compared in a quantitative way. However, in order to assess the full impact of OWFs on seabird population, total mortality, in relation to Potential Biological Removal has to estimated and we endeavoured to do this.

Furness *et al.* (2013) and Bradbury *et al.* (2014) considered flight altitude to be the most important factor in calculating collision risk for marine bird species at offshore wind farms. This was reflected in the calculating the collision risk for species.

Furness *et al.* (2013) and Bradbury *et al.* (2014) mention that their vulnerability index is based upon a limited set of factors and that there are other potential impacts that are not necessarily covered. An example of an additional factor that cannot easily be considered but should not be ignored is the possibility that e.g. weather conditions may affect collision risk for marine birds. For example fog or heavy rain, may obscure wind turbines and over-ride any species-specific differences in vulnerability.

### *4.3.3 Potential Biological Removal*

To assess the effect of the exploitation of multiple wind farms in the southern North Sea on populations of (migratory) birds and bats, the first step is to estimate the level of additional mortality. The second step would be to compare this estimated level of mortality with the level of human-caused mortality that can be sustained (each year) by the corresponding bird populations. Wade (1998) developed such a mortality limit (termed the Potential Biological Removal, PBR) to calculate the allowable human-caused mortality of marine mammals (Cetaceans and Pinnipeds).

Many (sea)bird species have similar life histories to Pinnipeds and Cetaceans, characterized by long life, delayed maturity, and low fecundity. Because of this, the model developed by Wade (1998) for Cetaceans and Pinnipeds is also relevant for (sea)birds (Dillingham & Fletcher 2008; Richard & Abraham 2013). The results of Milner-Gulland & Akçakaya (2001) suggest that the PBR approach could also be applied to a variety of other bird species as well (non-seabird species), such as moderately lived game birds. For short-lived species such as songbirds it may be less appropriate (Dillingham & Fletcher 2008), but following Bellebaum and & Wendeln (2011) an adaptation was made so that also for passerines a sufficiently precautionary approach of the PBR could be applied. In this study, we also used the Potential Biological Removal (PBR) approach followed by Lebreton (2005), Niel & Lebreton (2005) and Dillingham & Fletcher (2008) to estimate effects of offshore wind farms on bats at sea.

Recently, the Potential Biological Removal approach has been used in several studies in which the effects of additional mortality caused by collisions with wind turbines on bird populations were assessed (Watts 2010; Poot *et al*. 2011a; Sugimoto & Matsuda 2011; Bellebaum *et al*. 2013). PBR is based on harvest theory to estimate mortality limits. The results of these studies underline that the PBR is a useful tool to predict whether an additional source of mortality is unsustainable and point out bird populations worthy of careful future monitoring or indicate situations in which mortality-mitigation effort should be initiated (Wade 1998; Niel & Lebreton 2005). In assessing the effect of a specific source of human caused mortality, it is important to understand that the PBR includes all sources of human-caused mortality (Wade 1998; Dillingham & Fletcher 2008). Recently Richard & Abraham (2013) have put forward that "the PBR was not envisioned as a means to set levels of human-caused mortality, but as a tool to detect levels of fishing-related mortality that fail the management criterion. This approach is closer to current management process, which is setting priorities for the species that are the most at risk, rather than setting the level of fisheries-related mortalities for each species". Like in Wats (2010) we herewith use PBR "as the biological framework for evaluating limits to human- caused mortality that comply with the objective to determine the levels of incidental take that will not jeopardize the focal population. As with harvest, sustained levels of incidental take have the potential to drive populations to extinction, hold populations below carrying capacity, or to change recovery trajectories".

In this study we calculated the Potential Biological Removal for 61 bird species that migrate over the (southern) North Sea. The Potential Biological Removal is calculated as:

$$PBR = 0.5 * R_{max} * N_{min} * rf \qquad (1)$$

Where $R_{max}$ is the maximum annual recruitment rate, $N_{min}$ is a conservative estimate of population size and rf is a recovery factor between 0.1 and 1.0 (Wade 1998; Dillingham & Fletcher 2008). In the absence of harvest or incidental take, the growth curve of a population follows a sigmoid-shape towards the carying capacity level, due to density-dependency; when the population is still small it grows rapidly at first with halfway the steepest growth and then slows down as it approaches carrying capacity due to the influence of decreasing reproductive rates. The 0.5 in equation (1) is based on the point in the growth curve according a logistic model where the highest growth rate is reached and indicates the halfway point of the population size relative to the carying capacity level. On this point the sustainable level of harvest or incidental take reaches it's maximum and is equal to 0.5 times the maximum recruitment rate of a population. The relationship between sustainable harvest or incidental take and the equilibrium population size for a population growing according to the logistic model is a parabola with the maximum sustainable yield being equal to the maximum recruitment rate/2 and the population experiencing this harvest rate will be held at the carying capacity/2. For further explanations, see Watts (2010).

$R_{max}$ and maximum annual population growth rate ($\lambda_{max}$) are related through:

$$R_{max} = \lambda_{max} - 1. \qquad (2)$$

According to Wade (1998), one half of $R_{max}$ should be a conservative estimate of the current net production rate of a depleted population. If sufficient demographic information is available, matrix population models can be constructed to estimate $\lambda_{max.}$ If sufficient data is lacking, the 'demographic invariant method' (DIM) developed by Niel & Lebreton (2005) can be used to estimate $\lambda_{max}$ based on age at first reproduction ($\alpha$) and adult survival (s):

$$\lambda_{max} \approx \frac{(s\alpha - s + \alpha + 1) + \sqrt{((s - s\alpha - \alpha - 1)^2 - 4s\alpha^2)}}{2\alpha} \qquad (3)$$

Niel & Lebroton (2005) proposed to use this method to assess mortality levels for species where minimal information is available, such as seabirds. They estimated $\lambda_{max}$ for 13 bird species using both Eq. 3 and matrix models and compared the resulting estimates of $\lambda_{max}$. The strongest differences between $\lambda_{max}$ obtained from the matrix models and $\lambda_{max}$ obtained from Eq. 3 concerned the two passerine species (Great Tit *Parus major* and Rock Sparrow *Petronia petronia*). The $\lambda_{max}$ calculated by Eq. 3 appeared to be underestimated. Niel & Lebreton (2005) suggested to correct the calculated $\lambda_{max}$ for short-lived bird species using the following equation:

$$\lambda_{max} = \exp[(\alpha + \frac{s}{\lambda_{max} - s})^{-1}] \qquad (4)$$

In this study we defined short-lived species as species with an average lifespan of 4 years or shorter and calculated $\lambda_{max}$ for these species using both Eq. 3 and Eq. 4.

As presented in Figure 4.65, the most conservative estimate of the PBR is derived using high estimates of adult survival and age at first breeding (Richard & Abraham 2013). In this study we therefore selected the highest available estimates of adult survival and age at first breeding. For many species, data on adult survival were derived from the BTO BirdFacts website (http://www.bto.org/about-birds/birdfacts). In some cases an estimate of adult survival was available as well as the corresponding standard deviation. In that case the average estimate of adult survival plus the standard deviation was used in the calculations to obtain conservative estimates of the PBR.

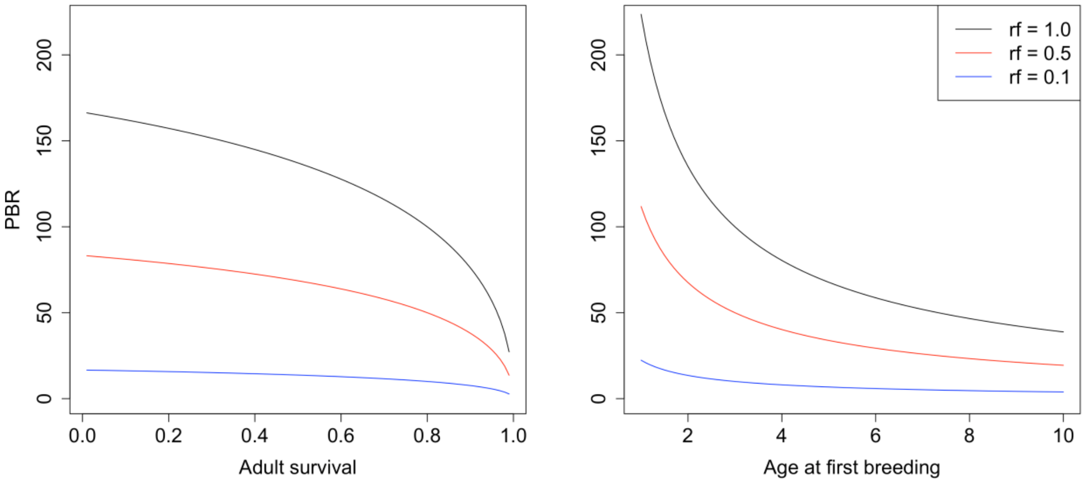


*Figure 4.65. PBR for rf =0.1 (blue), rf = 0.5 (red) and rf = 1.0 (black) plotted against the adult survival (left) and the age at first breeding (right). High estimates of the adult survival and age at first breeding lead to conservative estimates of the PBR.*

$N_{min}$ is a conservative estimate of the population size, suggested by Wade (1998) to be the lower bound of the 60% confidence interval, to be regarded as an important precautionary step to compensate for eventual bias in the few data used in the PBR approach. However, for birds virtually non of the available population estimates are based on samples with known variance estimates (Watts 2010). In this study we therefore used a point estimate of the minimum population size derived from literature or, if an upper and lower bound of the population size was available, $N_{min}$ was calculated following the equations described in Dillingham & Fletcher (2008).

$N_{min}$ can be estimated in terms of breeding pairs or individuals. To convert estimations of numbers of breeding pairs to individuals (and vice versa) a conversion factor of 2.5 was used. Dillingham & Fletcher (2011) developed a modified PBR value making use of simple population models, to calculate the PBR for albatrosses and petrels when only estimates of the number of breeding pairs are available. Results from this study show that in case of albatrosses and petrels the population size per breeding pair is (much) larger than 2.5. The same was shown by Richard & Abraham (2013). We assume that this is generally the case and that the conversion factor of 2.5 used in this study leads to a conservative estimate of the population size (number of individuals) and therefore a conservative estimate of the PBR.

The factor rf is a management factor, rf = 0.1 provides a minimal increase in recovery time for a depleted population or near-threatened population (IUCN criterion), to maintain a population size close to carrying capacity or to minimize the extinction risk for a population with a limited range. A value of rf = 1.0 could be used to maintain a growing population at or above its maximum net production level, recommended to use for a population with a least-concern status with a stable or increasing population trend (Dillingham & Fletcher 2008). Rf = 0.5 is an arbitrary intermediate stage for species with a least-concern status but with an unstable or decreasing population trend (Wade 1998; Dillingham & Fletcher 2008). Generally an rf < 1.0 is used (mostly rf = 0.5), to account for possible biases in for instance the estimates of $R_{max}$, $N_{min}$ or mortality (Wade 1998). In this study we used rf = 0.1 for species with an endangered or near-threatened status and/or a clearly declining population, rf = 0.5 as a default for species with a least-concern status and rf = 1.0 for those species with a least-concern status and a population that is clearly increasing. In some cases two different values of rf can be used (Annex D4). Bellebaum & Wendeln (2011) repeated the simulations of Wade (1998) to determine the appropriate rf to use for species with growth rates higher than those of marine mammals (generally <1.2), like for instance small birds. They found that for growth rates ($\lambda_{max}$) between 1.4 and 2.4 the recommended rf for stable populations would be 0.2 and 0.1 for threatened and/or declining populations (precautionary). As we strive to calculate conservative values for the PBR, we applied the rf value of 0.2 for short-lived species (average lifespan of 4 years or shorter), which generally have a maximum annual population growth rate ($\lambda_{max}$) higher than 1.4.

1% Additional annual mortality criterion in the Netherlands

In the framework of the Dutch nature legislation, criteria have been developed for acceptable effects on wildlife. For mortality of wind turbines on birds and bats the 1% additional annual mortality criterion has been proposed in procedures and is currently the generally accepted limit. This criterion has been developed in the framework of the European Bird Directive on sustainable hunting (European Commission 2008) and says that "the overall annual mortality is an appropriate parameter to quantify small numbers because it takes population size, status and population dynamics into account. Within this framework "small numbers" should be considered as being any taking of around 1% of the annual mortality for species which may be hunted, it being understood that conformity with Article 9 of the Directive depends in any event on compliance with the other provisions of the Article".

In Tables 4.23 and 4.24 the total number of collisions per seabird species and migrant bird species, respectively, in the southern North Sea is compared with the applicable Potential Biological Removal level (based on the status of the population, see Annex D4) and the Ornis committee criterion of 1% of the annual mortality. For the Lesser and the Great Black-backed Gull the numbers of collision victims exceed the PBR level. For the European Herring Gull, Northern Gannet, Black-legged Kittiwake and Great Skua the percentage relative to the PBR level are above 10% when presumed effects of all projected wind farms are added up. Like in the Lesser and Great Black-backed Gull species, these percentages for the European Herring Gull and Black-legged Kittiwake should probably be regarded as extreme worst case, because of a potential overestimation of numbers.

For none of the migrant species the numbers of collision victims exceed the PBR level (Table 4.9). For the Tundra Swan, Red Knot, Sanderling, Eurasian Curlew, Black Tern and Common Starling the percentage relative to the PBR level are above 10%.

*Table 4.23. Total number of collisions per* ***seabird species*** *in the southern North Sea, compared with the applicable Potential Biological Removal level (based on the status of the population, see Annex D4) and the Ornis committee criterion of 1% of the annual mortality. This criterion is used to determine whether the calculated number of victims can be regarded as a true impact in terms of increased mortality. In the last column those species are indicated with green of which the number of collision victims is below this 1% of the annual mortality and for these species the number of collision has no impact on the populations. In the third column the percentage is presented of the total number of collision victims relative to the PBR level. For the species for which the percentage are above 100 % are indicated with red and above 10% with orange.*

| Species | Total n collisions southern North Sea | applicable PBR | % collision/PBR | Ornis criterion 1% of annual mortality | % collision/1% annual mortality |
|---|---|---|---|---|---|
| Common Eider | 1735 | 22082 | 7,86% | 293 | 592,15% |
| Common Scoter | 13 | 27730 | 0,05% | 1167 | 1,11% |
| Velvet Scoter | 0 | 408 | 0,00% | 50 | 0,00% |
| Loon spec. | 130 | 1550 | 8,39% | 53 | 245,28% |
| Great Crested Grebe | 1 | 10705 | 0,01% | 402 | 0,25% |
| Northern Fulmar | 8 | 5934 | 0,13% | 99 | 8,08% |
| Sooty Shearwater | 0 | ++ | - | - | - |
| Manx Shearwater | 0 | ++ | - | - | - |
| European Storm Petrel | 0 | ++ | - | - | - |
| Leach's Storm Petrel | 0 | ++ | - | - | - |
| Northern Gannet | 2631 | 5245 | 50,16% | 143 | 1839,86% |
| Great Cormorant | 3 | 4919 | 0,06% | 144 | 2,08% |
| European Shag | 0 | ++ | - | - | - |
| Parasitic Jaeger | 2 | 812 | 0,25% | 28 | 7,14% |
| Great Skua | 12 | 120 | 10,00% | 5 | 240,00% |
| Black-headed Gull | 43 | 58986 | 0,07% | 1180 | 3,64% |
| Little Gull | 295 | 3971 | 7,43% | 139 | 212,23% |
| Mew Gull | 1524 | 22534 | 6,76% | 724 | 210,50% |
| Lesser Black-backed Gull | 23674 | 7560 | 313,15% | 220 | 10760,91% |
| European Herring Gull | 3381 | 4184 | 80,81% | 531 | 636,72% |
| Great Black-backed Gull | 5441 | 4144 | 131,30% | 107 | 5085,05% |
| Black-legged Kittiwake | 5930 | 16473 | 36,00% | 493 | 1202,84% |
| Little Tern | 1 | 39 | 2,56% | 5 | 20,00% |
| Sandwich Tern | 133 | 2378 | 5,59% | 52 | 255,77% |
| Tern spec. | 288 | 10076 | 2,86% | 295 | 97,63% |
| Common Murre | 13 | 26641 | 0,05% | 681 | 1,91% |
| Razorbill | 29 | 7129 | 0,41% | 249 | 11,65% |
| Little Auk | 1 | ++ | - | - | - |
| Atlantic Puffin | 0 | ++ | - | - | - |

*Table 4.24. Total number of collisions per* ***migrant bird species*** *in the southern North Sea, compared with the applicable Potential Biological Removal level (based on the status of the population, see Annex D4) and the Ornis committee criterion of 1% of the annual mortality. This criterion is used to determine whether the calculated number of victims can be regarded as a true impact in terms of increased mortality. In the last column those species are indicated with green of which the number of collision victims is below this 1% of the annual mortality and for these species the number of collision has no impact on the populations. In the third column the percentage is presented of the total number of collision victims relative to the PBR level. The species for which the percentages are above 10 %, are highlighted in orange (like inTable4.9; the colour red is not used as no level of above 100% was reached).*

| species | Total n collisions southern North Sea | applicable PBR | % collision of PBR | Ornis criterion 1% of annual mortality (individuals) | % collision/1% annual mortality |
|---|---|---|---|---|---|
| Tundra Swan | 58 | 131 | 44,0% | 27 | 213,5% |
| Pink-footed Goose | 27 | 1169 | 2,3% | 41 | 65,3% |
| Barnacle Goose | 36 | 13075 | 0,3% | 323 | 11,2% |
| Brent Goose | 155 | 6056 | 2,6% | 95 | 163,3% |
| Common Shelduck | 158 | 3447 | 4,6% | 74 | 213,2% |
| Tufted Duck | 124 | 10885 | 1,1% | 436 | 28,5% |
| Greater Scaup | 126 | 4392 | 2,9% | 470 | 26,8% |
| Eurasian Wigeon | 124 | 128325 | 0,1% | 6124 | 2,0% |
| Great Cormorant | 1 | 4919 | 0,0% | 144 | 0,7% |
| Great Crested Grebe | 3 | 10705 | 0,0% | 402 | 0,7% |
| Western Osprey | 1 | 436 | 0,2% | 15 | 6,8% |
| Peregrine Falcon | 1 | 156 | 0,5% | 4 | 20,1% |
| Grey Plover | 38 | 4337 | 0,9% | 105 | 35,9% |
| Northern Lapwing | 1047 | 33999 | 3,1% | 2534 | 41,3% |
| Red Knot | 654 | 6099 | 10,7% | 372 | 175,8% |
| Sanderling | 377 | 1770 | 21,3% | 48 | 790,5% |
| Dunlin | 405 | 59643 | 0,7% | 1988 | 20,4% |
| Common Snipe | 365 | 20542 | 1,8% | 2068 | 17,6% |
| Eurasian Woodcock | 395 | 27996 | 1,4% | 2564 | 15,4% |
| Bar-tailed Godwit | 434 | 6737 | 6,4% | 203 | 213,9% |
| Eurasian Curlew | 543 | 901 | 60,3% | 148 | 367,2% |
| Common Redshank | 617 | 14590 | 4,2% | 559 | 110,3% |
| Ruddy Turnstone | 80 | 3322 | 2,4% | 80 | 99,5% |
| Black-headed Gull | 519 | 58986 | 0,9% | 1180 | 44,0% |
| Black Tern | 23 | 43 | 52,2% | 6 | 356,9% |
| Short-eared Owl | 1 | 663 | 0,2% | 15 | 6,8% |
| Common Swift | 358 | 47155 | 0,8% | 2283 | 15,7% |
| Goldcrest | 1453 | 1099504 | 0,1% | 59351 | 2,4% |
| Eurasian Skylark | 13557 | 404804 | 3,3% | 38456 | 35,3% |
| Barn Swallow | 35 | 424754 | 0,0% | 21939 | 0,2% |
| Willow Warbler | 47 | 4470384 | 0,0% | 235603 | 0,0% |
| Common Starling | 17210 | 139577 | 12,3% | 26678 | 64,5% |
| Common Blackbird | 3464 | 2365907 | 0,1% | 102326 | 3,4% |
| Song Thrush | 3408 | 917732 | 0,4% | 42590 | 8,0% |
| Redwing | 2731 | 488221 | 0,6% | 24680 | 11,1% |
| European Robin | 3200 | 1674271 | 0,2% | 84507 | 3,8% |
| Northern Wheatear | 30 | 158368 | 0,0% | 7881 | 0,4% |
| Western Yellow Wagtail | 28 | 65545 | 0,0% | 3055 | 0,9% |
| Meadow Pipit | 358 | 287413 | 0,1% | 12283 | 2,9% |
| Common Chaffinch | 14547 | 3486037 | 0,4% | 158056 | 9,2% |
| Total | 66737 | 16452995 | 0,4% | 839712 | 7,9% |

# 5 Application of assessment and validation methods within CUMULEO

## 5.1 Impact of OWF relative to PBR, using the extended-Bradbury method

Combining estimated mortalities from all projected offshore wind farms in the North Sea allows an evaluation of the sustainability of these wind farms in relation to seabirds, using the Potential Biological Removal (PBR) limits derived for these birds. The estimated mortalities per seabird species, with the appropriate PBR values are listed in Table 5.1 and Annex H. If we stack all mortality estimates, per species, for all projected wind farms, it turns out that, with the possible limitations of the method used, no single seabird species will receive wind farm caused mortality that approaches PBR. In other words, although mortality is projected to occur, this remains within safe biological limits, for all species. Figure 5.1 depicts the cumulative effects of all projected wind farms in the southern North Sea (by the year 2023, as can currently be envisaged), for the ten species for which wind farm mortality is closest to their respective PBR values (scaled at 100%). Wind farms have been grouped by national territory (underlying part of the southern North Sea). The outcome, on a nation by nation basis, is largely governed by differences in specific seabird densities in the North Sea, and the area covered by future wind farms in each national North Sea sector. The results show, that wind farms in the German sector are expected to have a relatively large impact on large gulls and loons, while UK wind farms impact auks, Black-legged Kittiwake and Northern Gannet most. Wind farms in the Netherlands and Belgium have most effects on large gulls, and the Belgian wind farms pose a relatively high risk to Northern Gannets.

*Table 5.1. Estimated mortalities per seabird species and appropriate PBR values.*

| ESAScode | Species | PBR | Collision numbers | Displacement mortality numbers | Total mortality numbers | Collision/PBR (%) | Displacement mortality/PBR (%) | Total mortality/PBR (%) |
|---|---|---|---|---|---|---|---|---|
| 20 | Red-throated Loon | 1378 | 37 | 111 | 145 | 3% | 8% | 10% |
| 30 | Black-throated Loon | 179 | 5 | 12 | 17 | 3% | 7% | 9% |
| 50 | White-billed Loon | very low | 0 | 0 | 0 | ? | ? | ? |
| 90 | Great Crested Grebe | 10705 | 0 | 1 | 1 | 0% | 0% | 0% |
| 220 | Northern Fulmar | 5934 | 160 | 107 | 266 | 3% | 2% | 4% |
| 430 | Sooty Shearwater | very high | 0 | 0 | 0 | 0% | 0% | 0% |
| 460 | Manx Shearwater | high | 0 | 1 | 1 | 0% | 0? | 0? |
| 462 | Balearic Shearwater | very low | 0 | 0 | 0 | ? | ? | ? |
| 520 | European Storm-petrel | high | 0 | 0 | 0 | 0% | 0% | 0% |
| 550 | Leach's Storm-petrel | high | 0 | 0 | 0 | 0% | 0% | 0% |
| 710 | Northern Gannet | 5245 | 837 | 105 | 935 | 16% | 2% | 18% |
| 720 | Great Cormorant | 4919 | 14 | 5 | 18 | 0% | 0% | 0% |
| 800 | European Shag | medium | 0 | 0 | 0 | 0% | 0% | 0% |
| 2060 | Common Eider | 22082 | 10 | 40 | 49 | 0% | 0% | 0% |
| 2130 | Common Scoter | 27730 | 13 | 67 | 80 | 0% | 0% | 0% |
| 2150 | Velvet Scoter | 409 | 0 | 0 | 0 | 0% | 0% | 0% |
| 5660 | Pomarine Skua | medium | 0 | 0 | 0 | ? | ? | ? |
| 5670 | Parasitic Jaeger | 812 | 8 | 1 | 9 | 1% | 0% | 1% |
| 5680 | Long-tailed Jaeger | medium | 0 | 0 | 0 | ? | ? | ? |
| 5690 | Great Skua | 120 | 14 | 3 | 16 | 11% | 2% | 13% |
| 5780 | Little Gull | 3971 | 92 | 18 | 109 | 2% | 0% | 3% |
| 5820 | Black-headed Gull | 58986 | 41 | 12 | 53 | 0% | 0% | 0% |
| 5900 | Mew Gull | 22534 | 412 | 66 | 471 | 2% | 0% | 2% |
| 5910 | Lesser Black-backed Gull | 7560 | 3686 | 246 | 3902 | 49% | 3% | 52% |
| 5920 | Herring Gull | 4184 | 882 | 43 | 918 | 21% | 1% | 22% |
| 6000 | Great Black-backed Gull | 4144 | 1008 | 99 | 1090 | 24% | 2% | 26% |
| 6020 | Black-legged Kittiwake | 16473 | 3197 | 731 | 3877 | 19% | 4% | 24% |
| 6110 | Sandwich Tern | 2378 | 22 | 11 | 33 | 1% | 0% | 1% |
| 6140 | Roseate Tern | very low | 0 | 0 | 0 | 0? | 0? | 0? |
| 6150 | Common Tern | 4930 | 64 | 33 | 96 | 1% | 1% | 2% |
| 6160 | Arctic Tern | 5146 | 32 | 33 | 64 | 1% | 1% | 1% |
| 6240 | Little Tern | 39 | 1 | 0 | 1 | 2% | 1% | 3% |
| 6340 | Common Murre | 26641 | 449 | 3464 | 3896 | 2% | 13% | 15% |
| 6360 | Razorbill | 7129 | 31 | 550 | 580 | 0% | 8% | 8% |
| 6380 | Black Guillemot | medium | 0 | 0 | 0 | 0% | 0% | 0% |
| 6470 | Little Auk | very high | 1 | 12 | 13 | 0% | 0% | 0% |
| 6540 | Atlantic Puffin | very high | 3 | 47 | 50 | 0% | 0% | 0% |

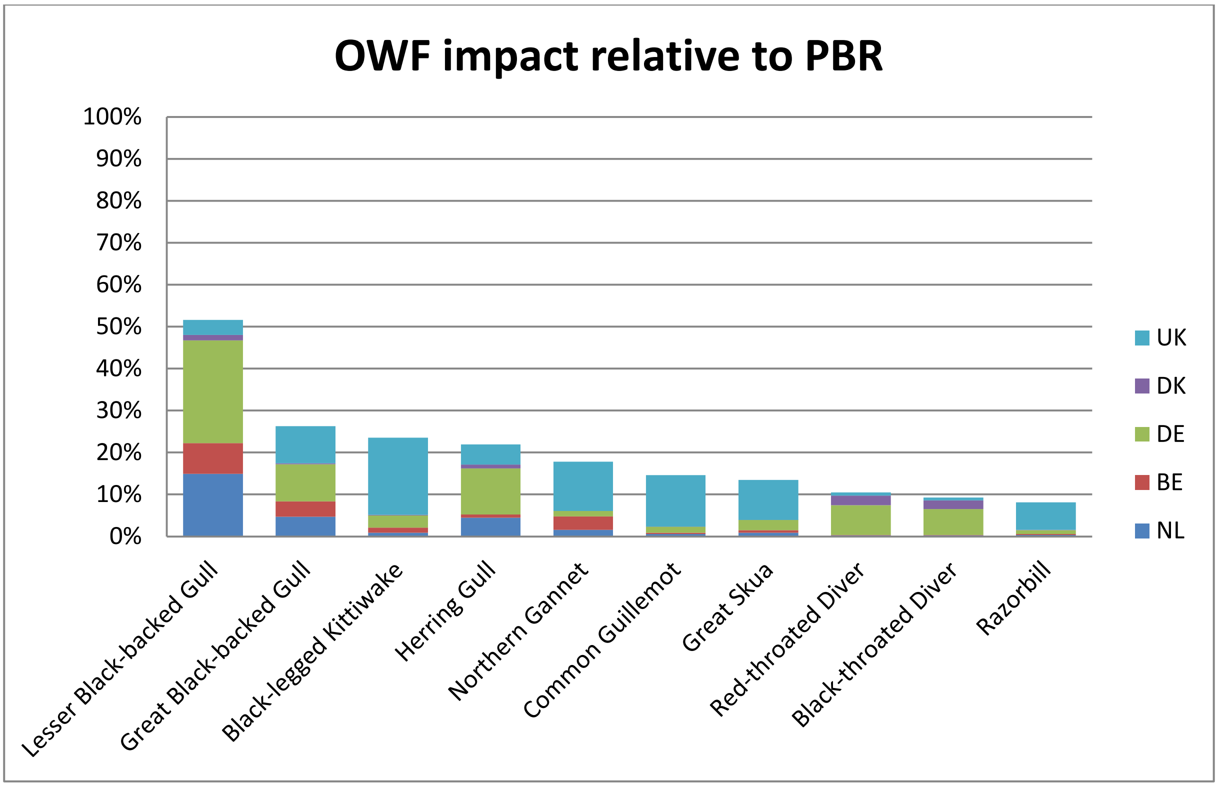


*Figure 5.1. Cumulative effects of all projected wind farms in the southern North Sea (as envisaged by 2023) for the ten species, for which the wind farm mortality value is closest to their respective PBR values (scaled at 100%) and the origin of the involved national exclusive zones.*

## 5.2 Cross-check

A major risk of the extended-Bradbury method is, that the basic method has only been published recently (September 2014) and that the necessary extension of the method, to arrive at absolute mortality numbers was developed for this report, and has not been rigorously tested.

The Bradbury method has two mortality components: direct mortality from collisions and indirect mortality from displacement and subsequent habitat loss. The first component, collision mortality, can be independently estimated (that is: using the same on-site seabird density estimates) by using the SOSS Band model (Band 2012; see paragraph 4.2.6.4).

The Bradbury method can only be applied to seabirds, and the Band model only to collisions, so only the estimates for seabird collisions can be cross-checked by either model. We isolated the collision mortality component from the Bradbury estimates, per seabird species and per offshore wind farm in the southern North Sea, and estimated the same values using the Band model. The outcomes are compared in Figure 6.1 and 6.2. It turnes out that the Band model estimates are often considerably higher than the Bradbury estimates and even surpass, for two species (the Lesser Blacked Gull and the Great Black-backed Gull), PBR, the safe biological limit. We do not know which model predicts actual mortality best. However, the Band model is very sensitive to high fluxes of seabirds estimated to fly through wind farms, and the fluxes used were derived from local densities. In a number of (projected) wind farm locations, very high gull densities were estimated. This was the result of a few very high counts of large numbers of gulls around fishing vessels and extrapolation of such point counts to entire wind farm surface areas. Such high input values are regarded unrealistic for two reasons: first, no trawling is likely to be allowed in future wind farms, so these peak-densities will occur only outside the wind farms and second, high point counts appear to have been extrapolated too liberally onto entire wind farms. Real mortality rates, in either model, are thus probably much lower for species attracted to fishing vessels, than currently estimated.

The second part of total wind farm mortality, stemming from displacement, cannot yet be validated by an independent method. An alternative approach would require values for the actual displacement factors (proportions of birds displaced) and for the resulting mortality of the displaced birds (not tentatively set at 10%). Displacement factors have been estimatated in various impact studies, but only for a few species and the outcomes of different studies were found to vary. This is likely to be due to different lay outs of the wind farms studied. This problem is further studied by Zuur (Annex C) for the most widely spread species, the Common Murre (or Common Guillemot). However, other factors, such as dependence on a given area (an "area effect"), season, or background density may also play a role and this needs more study.

## 5.3 Integrated seabird species sensitity to OWF (updated Windspeed map)

The first North Sea map of integrated seabird sensitity to offshore wind farms was published by Garthe & Hüppop (2004). They used distribution data for all (relevant) seabird species, and multiplied the yearly average for each species with a species-specific wind farm sensitivity index (WSI). By adding up the resulting values for each pre-defined grid cell, a value of wind farm sensitivity across all relevant seabird species was derived, that was geo-referenced. Using additional survey data, and slightly amended WSI factors, an update of this first map was provided in Leopold & Dijkman (2011). Maps like these can be used for spatial planning of offshore wind farms.

In the current exercise, again more survey data have become available, and also a new method to use these data to generate integrated sensitivity values across the study area. We used the Bradbury *et al.* (2014) approach, extended to derive absolute numbers of casualties for all seabirds considered in this report to generate a new seabirds sensitivity map (Figure 5.2).

Similar to the 'Windspeed map' presented in Leopold & Dijkman (2011), three categories are used in the new map: areas of less concern (green), areas of concern (yellow) and areas of high concern (red). The cut-off points between categories are based on the frequency distribution of the summed sensitivity values across the 5x5 km grid cells in the southern North Sea, as used elsewhere in this report. The 60% grid cells with the lowest values are considered to be of less concern; the 20% grid cells with the highest values are of high concern and the 20% grid cells with intermediate values are of concern.

The map shows broad areas of high concern in a broad band along mainland Europe, and off England's Northumberland, Tyne & Wear, Cleveland, North Yorkshire and Humberside (c.f. Mitchell *et al.* 2004, Figure 4), and a band of concern/high concern across the North Sea bridging these two. This bridge follows the zone of tidal fronts between summer-stratified waters in the north, and non-stratified waters in the south (Pingree & Griffiths 1978), including the Natura 2000 Site Frisian Front in the Dutch sector. The map is entirely based on seabirds at sea survey data, but has a clear hydrographical basis, where coastal and frontal waters show up in red (high concern, i.e., high general densities of sensitive seabirds), and may thus be considered robust. Note that values depicted in the Wadden Sea (high concern) are not based on seabirds data, but on extrapolation from high values in North Sea coastal waters.

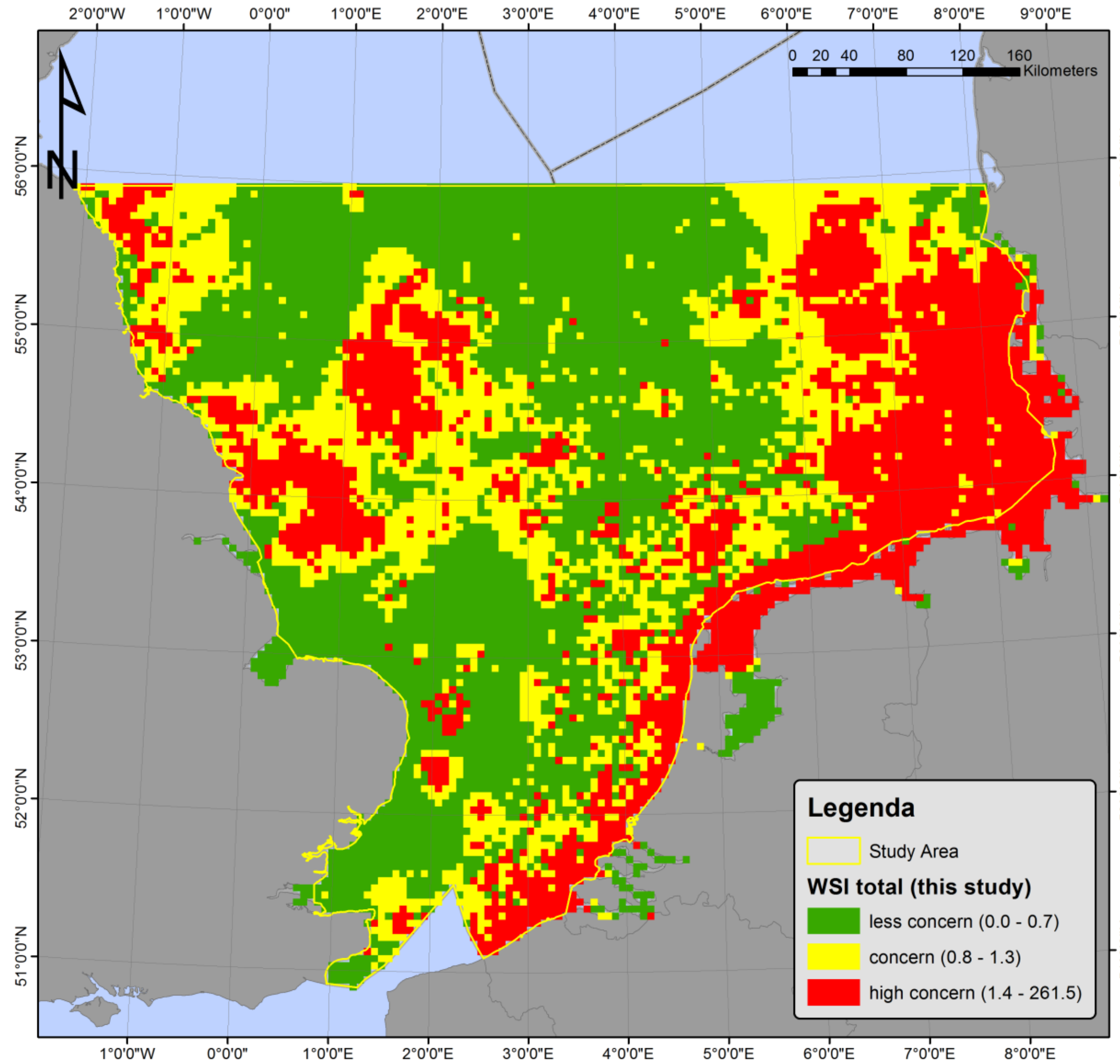


*Figure 5.2. Integrated seabirds wind farm sensitivity map for the southern North Sea. Seabird sensitivity summed for relevant seabird species in plotted on a 5x5 km grid, using density-weighted species-specific vulnerability assessments (following the extended-Bradbury method; Bradbury et al. 2014) based on presumed collision and displacement risks.*

## 5.4 Integrating other activities in CUMULEO

Offshore wind farms are not the only human activity in the southern North Sea that might impact birds and bats. Wind turbines are the major feature that will kill these animals directly, through collisions. For seabirds, at least three other sources of direct mortality are known: hunting and harvesting (including egg collecting), pollution (particularly oil spills) and drowning in fishing nets. Indirect mortality may result from direct competion for food, if for instance fisheries target the same organisms and sizes as targetted by the birds, and grossly overfish these resources; more subtle pollutants that do not cause direct mortality (such as oil spills) but rather cause reduced fecundity.

There are, in fact, two types of pressures that impact seabird populations. Pressures that cause direct mortality are often spectacular, but their effects can to a large extent be compensated by density-dependent 'correction mechanisms'. As such, direct mortality from e.g. oil spills, hunting, drowning in fishing nets, but also from wind farm collisions, do not necessarily impact population size. This is not true for pressures that increase (discarding, eutrophication, removing competition, colony protection) or decrease (discards-ban, de-eutrophication, introduction of predators, habitat loss) on a more structural basis. In some cases, there is a fine line between 'incidental' and 'structural' pressures. For instance, an oil spill near a very important breeding colony, also at regional scale, may gravely impact populations e.g., the Prestige oil spill near the last remaining colony of Iberian Guillemots (Common Murres), or the Amoco Cadiz break-up near the few remaining French Atlantic Puffin colonies. Also, a very large hunting pressure or egg collecting may structurally reduce seabird populations. By and large, however, more structural pressures are more likely to impact seabird population sizes than more incidental mortality factors. Therefore, in the long run, habitat loss resulting from offshore wind farm development may have a more severe impact on the seabirds than the wind farm associated collisions, even though actual mortality rates may be higher in the latter.

In contrast to offshore wind farms such sources of mortality cannot easily be geographically referenced, and estimating cumulative effects of all these factors on seabirds would require more study. Indirect mortality from fisheries is also a highly complex issue, and by and large, fisheries have been probably mostly benificial to the North Sea seabirds, by removing competion (large predatory fish) and supplying an ample supply of discards and offal as food for an estimated one million seabirds in the North Sea (Camphuysen *et al.* 1995b). Conversely, a future discards ban will turn part of this development around, leading to lower ("more natural") numbers of seabirds in the North Sea. No studies have yet addressed this issue in full and no estimates for loss of numbers are yet available. However, in contrast to loss of life caused by wind farm collision, these losses will be structural, as lost birds cannot be replaced by density dependent processes. Therefore, food supply related changes in seabirds numbers are structural and will lower seabirds numbers, whereas losses from wind farms, due to collisions, if lower than PBR, will be compensated for.

Regarding migrant birds, direct mortality is known from gas flares, that might kill many tens of thousands of birds in a single night if conditions are exactly 'right' (see Lensink *et al*. 1999 and Wiese *et al*. 2001 for reviews). Estimates of total numbers of victims are made by Bruinzeel *et al*. (2009) and Bruinzeel & Van Belle (2010), but are likely to decrease as fewer flares with be used in the future, due to dwindling gas reserves and protective measures regarding migrant birds.

As an example of how the issue of cumulation with other activities can be addressed, we analysed the presumed effect of (existing) shipping. We considered ships more or less equal to wind turbines, in that they will displace birds from their immediate surroundings. As explained in section 4.2.4, shipping has only a temporary effect at any one location, but the effect may become permanent if enough ships pass by, through busy shipping lanes. This makes that shipping lanes and in fact all shipping when corrected for this temporal effect, can be compared to the displacement effect of offshore wind farms. Note that we only consider displacement here, as ships are very unlikely to cause collision mortality. Freak collisions events between seabirds and ships are known (e.g. Dick & Donaldson 1978), but considered to be so rare that they need not further be addressed here. We estimated that ship-related (displacement) mortality in the southern North Sea is at least an order of magnitude less important than offshore wind farm-induced mortality (Annex F).

From the recent literature (Schwemmer *et al*. 2011; Furness *et al*. 2013; Bradbury *et al*. 2014) about the sensitivity of birds for shipping it can be concluded that the sensitivity is influenced by:

- shipping density (especially loons and seaduck are very sensitive to high shipping densities)
- habituation, which may occur in some species as time progresses
- interspecific differences
- flock size (increase in sensitivity with flock size)
- season in the year
- types of ships
- temporal habitat loss (differs per species)
- food availabilty (lower sensitivity for displacement by ships in areas with high food availabililty)

We have used the Bradbury seabird sensitivity scores developed for offshore wind farms to assess the impact of shipping on seabirds as well, but only used the part on displacement, as collisions are unlikely to occur.  We assume that the sensitivities of birds for turbines and ships are similar. The footprint of a wind turbine is geographically small, but constant in time. The footprint of one ship is geographically large but occurs over a short period at any one spot, because a ship is moving by. A correction for the time factor of shipping is already applied in the MARIN map used as input for our modelling. This map shows the average shipping density over a year (Figure 4.5). Five shipping density classes are used (Table 5.2). Only the two classes connected to major shipping lanes (highest density) are used in our calculations.

*Table 5.2. Shipping density classes and space taken up by shipping lanes in the southern North Sea.*

| Class | Interpretation of density class | Route bound? | Density (n/1000 km2) | Area (km2) |
|---|---|---|---|---|
| 1 | less than one per month | No | 0.03 | 55500 |
| 2 | one per month to one per week | No | 0.25 | 29175 |
| 3 | one per week to one per day | No | 1.36 | 63200 |
| 4 | one per day to one per hour | Yes | 17.5 | 41825 |
| 5 | more than one ship per hour | Yes | 85.6 | 28375 |
| 4+5 | | | | 70200 |
| all | | | | 218075 |

The impact of non-route bound shipping is not included for two reasons:

1. a considerable part of this shipping is fishing vessels and their impact on birds is very different from most other vessels (attraction for some, not for others etc.)
2. options exist and to some extent are already used to control the impact on the ecosystem (a.o. birds) by having seasonal and other closures. This also influences recreational vessels.

The number of seabirds displaced by shipping is estimated by multiplying the density of seabirds in the shipping lanes with the sensitivity factors derived by Bradbury *et al*. (2014). The results are listed in Annex F.

Comparison of shipping and OWF

The numbers of seabirds affected by shipping are much lower than the numbers affected by OWF. The difference cannot explained by the difference in affected area. The area influenced by route-bound shipping is about a factor 7 higher than the area by OWF. The number of wind turbines (8282 estimated to be operational by 2023) is about a factor 7.5 higher than the average number of ships in shipping routes (estimated at 1160, on average).

Possible reasons for the lower impact by shipping are:

1. On average shipping is located further away from the shore than OWF, i.e. away from the areas used most intensively by seabirds.
2. Shipping have already displaced many birds, resulting in low densities as input for the modelling of the effect of shipping lanes.

# 6 Discussion

## 6.1 Uncertainties

This study explores new aspects of presumed bird and bat mortalities, related to offshore wind farms. As yet, few impact studies, in relation to the projected number of offshore wind farms have been conducted and analyses of displacement and collision rates were complex, because these were often hampered by insufficient sample sizes for many species and large numbers of zero counts in combination with counts of large numbers of birds. Methods used for the analyses differed between studies, and the outcomes, to some extent, were contradictory in several cases. Moreover, studies of truly offshore wind farms, e.g. in the Dogger Bank area, have not been conducted, for the simple reason that such wind farms do not yet exist. Species that do not occur in a sufficient number of (potential) wind farm areas, such as Northern Fulmars, Atlantic Puffins, or Little Auks can only be considered by extrapolation. Projected mortalities for such species should be regarded with great caution and it should be noted that many future wind farms are projected in core habitats of these species in the North Sea.

By definition, few data are available for the rarer seabirds, that nevertheless might be impacted. The most extreme uncertainties concern the bats, for which we do not even know the approximate numbers flying across the North Sea. The same is true for many migrant birds, for which we must rely on estimated population sizes, migration routes and values for flight behaviour. For all birds, collision modelling greatly depends on assumptions on micro- and macro avoidance rates, and likewise, rough estimates need to be used to estimate the carry-on effects of displacement.

Population estimates for geese, ducks and waders are quite accurate. Since these species are relatively large in size, live in flocks, and/or use communal roosting sites, and live in open landscapes, they are easy to find and count. Data on bird numbers are gathered in the wintering areas, or on stopover sites during migration with relatively great accuracy. Population estimates for songbirds are less accurate. Estimates are mainly based on samples of breeding bird densities, followed by an extrapolation to arrive at a total number. In large areas in northern Europe and further east in Russia information is scarce. Therefore, population estimates for passerines in large areas in boreal regions are just rough estimates.

Birds crossing the North Sea according to the two-directional hypothesis (Buurma 1987), might cross the line between South Norway and the entrance of the Channel several times; e.g. a bird leaving Norway in the direction SW will after some hours change towards SE and will arrive later on at the Dutch Wadden coast from the NW. From there these birds can leave for England towards SW. Since the estimated flux is based on one passage by a bird of the imaginary line, the figures presented here potentially are an underestimation of reality.

For collision rate modelling, the estimated fluxes were modelled as a single movement in autumn. Spring fluxes were taken as 0.6 of the autumn flux. The proportion of birds at rotor height was taken from Cook *et al*. (2012) and Wright *et al*. (2012a,b).

For the seabirds, we note that not all at-sea survey data were available for this study (i.e., forwarded to the ESAS database). This is particularly true for UK WWT aerial survey data (see: Bradbury *et al.* 2014), and probably for data from other countries as well. Estimates of numbers of seabirds present in such areas can thus possibly be improved by including these missing data. A fuller assesment of wind farm related effects can therefore only be made in a more international setting, drawing experts from all North Sea nation states, and their data. The present study should thus be seen as a first step only in the road to a full analysis of offshore wind farm effects on North Sea seabirds, given the wind farms currently considered to be developed by 2023.

A caveat arising from this study were unrealistically higher than expected densities of some birds in some areas, particularly in some projected future offshore wind farms. This is probably the result of concentrations of these birds counted around fishing vessels, or flocks of passing migrants, followed by standard data interpolation. We used IDW (inverse distance weighing) to “dampen” large peaks in seabird count data, and to fill in areas where no actual counts were made. A side-effect of this –and any other- interpolation technique is that observations impact a larger area than just the location where the birds were actually observed. This is not unrealistic in most cases, where a high observation is indicative of similar high observations being expected in the vicinity. But if the birds that joined the concentration had been drawn in from a larger, surrounding area, the peaks, although genuine for that moment in time, are very different from other moments, and from the surrounding area. For these situations adding an extra step while preparing the data for interpolated (or extrapolated) density maps would be necessary. As the association with fishing vessels is well-known and predictable behaviour, the recorded counts also come with information relevant to this specific case. However, the final result may be that a rather large area may “receive” very high bird numbers, if birds in the concentration came from a much larger area than used in the IDW. If a future wind farm is projected in such a smeared concentration of birds, unrealistically high numbers of birds will be assigned to this future wind farm for two reasons: 1) because the birds came from a larger area around the point count than appreciated by IDW and 2) because such bird concentrations were often related to fishing activities, that will be banned from future wind farms. A clear example of a ‘constructed’ concentration of birds resulting from the IDW data treatment may be seen in the Northern Fulmar map for February/March, off the Danish coast, but also in maps for relatively rare birds, such as skuas and jaegers. Similar problems may arise from groups of migrants (such as Common Eiders) being counted incidentally at locations were they normally would be rarely seen.

## 6.2 Extended-Bradbury method versus Band model outcomes

Based on a comparison of the total number of collision victims per wind farm (Figure 6.1) and on species level (Figure 6.2), the outcomes via the Band model are around a factor 5 times higher than the Bradbury method.

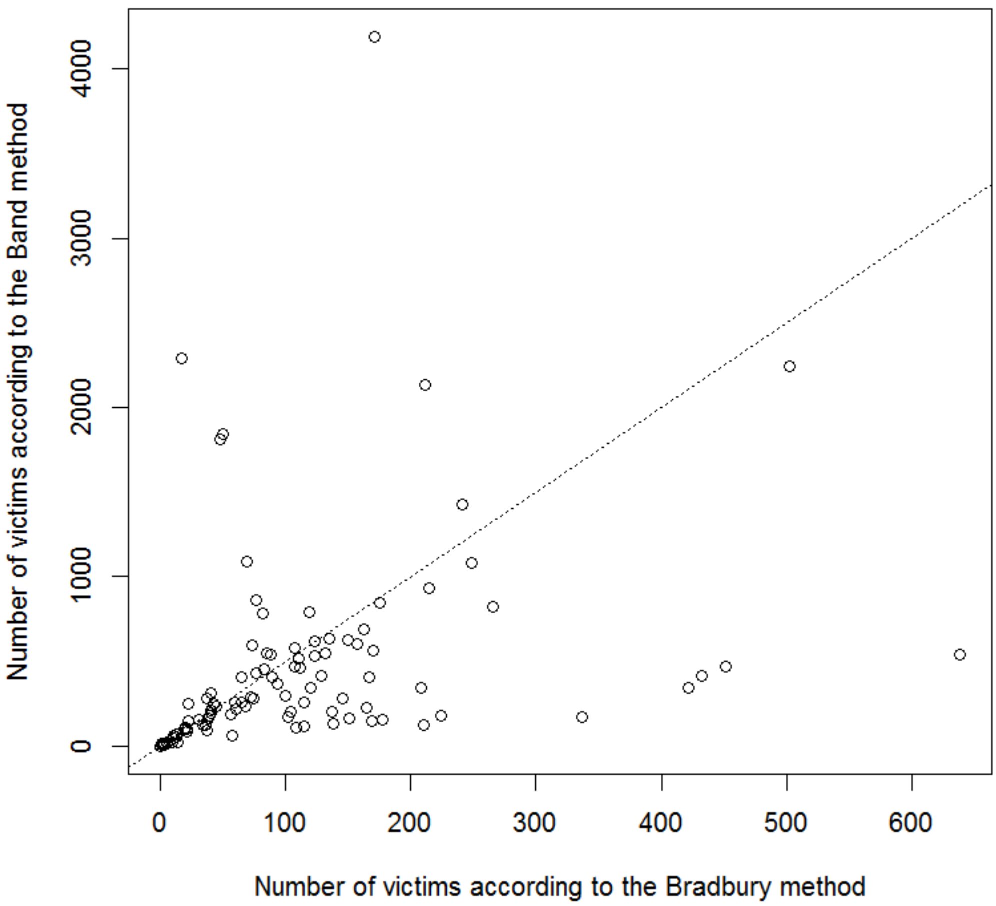


*Figure 6.1. Total number of collision victims in seabirds for > 100 individual wind farms at the southern North Sea calculated with the extended Band model compared with the number calculated according the extended-Bradbury method. The lined drawn depicts the proportional relationship according the factor 5.*

Short-lived concentrations may occur in any seabird, for a number of reasons, anywhere, and we feel that the peak counts resulting from these will have biased model outcomes in this study in some cases. Bird densities used as input for our modelling work may have been too high for some individual wind farm locations. With the approach used here, the model output will be positively biased and real numbers of wind farm victims will be lower than predicted. Therefore, we feel that the high mortality estimates for some gulls are over-extrapolations. This problem needs further study, however.

Exploring the cumulation of mortality between different wind farms was the main aim of this study. Clearly, other mortality factors are at play and have only been touched upon in this report.

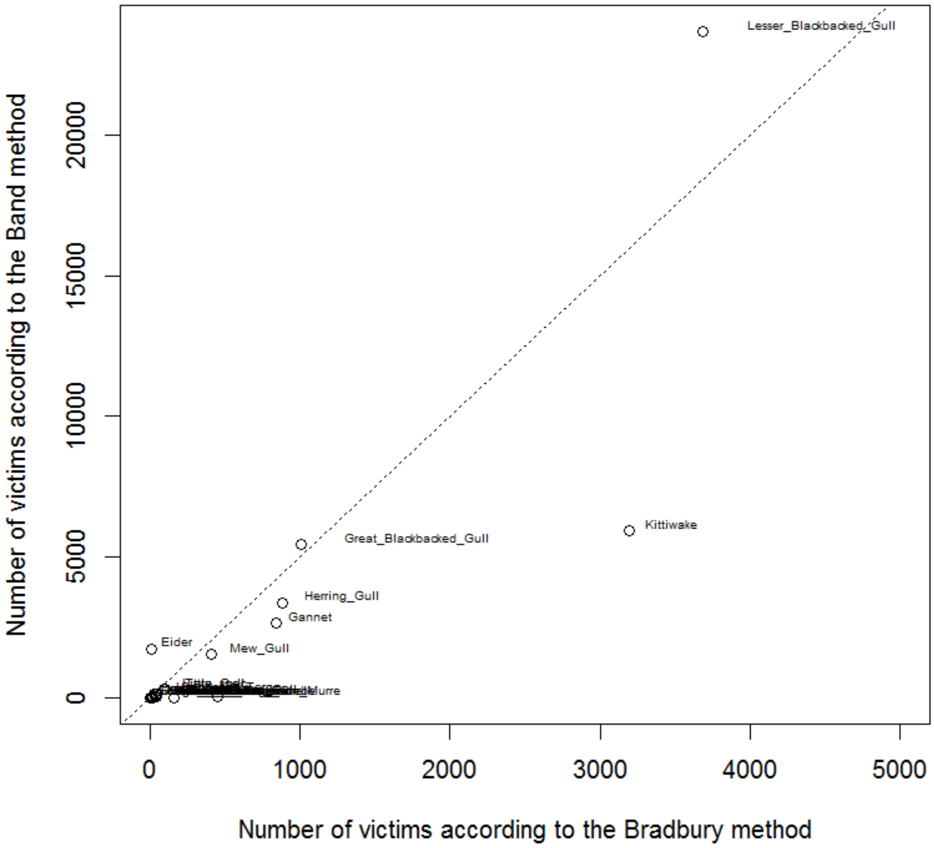


*Figure 6.2 Total number of collision victims per seabird species summed for 100 individual wind farms at the southern North Sea, calculated with the extended Band model compared with the number calculated according the extended-Bradbury method. The lined drawn depicts the proportional relationship according the factor 5. .*

## 6.3 Mitigation and compensation

Even though the number of expected casualties is expected to remain below PBR in most, if not all species, casualties will occur and habitat loss caused by increasing numbers of offshore wind farms will increase steadily. Mitigation, if this involves stopping the turning of the rotors when e.g. migration of birds is very strong, requires good estimation of peak bird occurences, on site. Peak moments of migration will need to be predicted accurately, thus preventing unnecessary loss of revenues. This will need a good system of bird tracking across the North Sea, by radar and a good predictive model.

Migrating bats, probably invisible on long-distance radar, will require special attention. Most bat migration seems to occur during nights with very low wind speeds, and even then only in considerable numbers during late summer and autimn, when turbine revenues are low anyway, so there may be possibilities to stop production temporarily, should bats be at risk in significant numbers (needs to be established). This could, with proper technology installed, possibly be organised in a ‘hand on tap’ modus, using real time, on-site monitoring to determine critical peaks in bat migration.

Other ways to mitigate effects of offshore wind farms may be found in choosing location wisely, that is, avoiding areas of high collission risk (see Figure 5.2), use configurations of wind farms that are bird- and bat-friendly, choose turbine types that cause relatively little disturbance and collisions, or optimising the visibility of the rotor blades to minimise collisions. Note that the latter approach might inadvertedly increase bird displacement, i.e. increase habitat loss. Knowledge of the effects on wind farm configuration, turbine types or turbine visibility on collision and displacement rates is still in its infancy, making assessements of effects at this stage highly speculative.

There has been speculation about the value of corridors through (larger) offshore wind farms, or between neighbouring wind farms. The validitiy of these, for aiding bird migration and reducing collision risk, needs to be established as well. An important observation at the current stage is that, if corridors are planned (for whichever reason), the design of neighbouring wind farms should be taken into account. There is a risk of a corridor of one park terminating at the fringe of another wind farm, designed by another developer. This would lead birds flying through the corridor "into a brick wall", i.e., they would be led right into the next wind farm. Such a risk might exist within the Borssele/Belgian wind farms complex and needs to be considered at the drawing stage.

Probably most important is the notion, that offshore wind development is going to be a major new issue in the southern North Sea, potentially with significant impacts on birds and bats. Our estimated cumulative mortalities remain under (using the extended-Bradbury method) or may in some instances surpass (using the Band model) PBR, or are simply unknown (bats). Considerable uncertainty on the cumulative effects of offshore wind farms on birds and bats therefore remains. However, not all planned wind farms will be built over night and a good monitoring should be part of this developement. This will help to better understand the magnitude of the problem. Each new wind farm will add to that stress, increasing the cumulative effect. Therefore, as outlined in this report, it should be possible to follow the build-up of this new stress factor, as more and more wind farms start production. It should thus also be possible to predict, with increasing certainty, when PBR levels for any species will reach, or surpass, PBR. If worse comes to worse, that is, if the cumulative effects reach PBR for a given species, it might still be possible to stop further developments in prime habitats for species A, and only develop further wind farms in other parts of the North Sea. Here, other species, still under PBR, will be impacted rather than the species near, at, or already over PBR.

Mankind has been very successful at feeding birds at sea, thus increasing population sizes, by ample supplies of fishery waste. Now these discards will be phased out, adding stress to populations also impacted by offshore wind development. Discards-eating birds will thus be hit from two sides, and populations most likely will drop. This may not be seen as a problem for species such as gulls, that are often seen as pests, and that have population levels boosted by artificial food supllies. However, most of these gulls breed in protected colonies, and are protected species under the EU Birds Directive (Natura 2000). Other, less impopular birds, such as the Northern Gannet and several terns, face the same problem. Mitigation, and even compensation, might be possible for some of these –colony breeding- birds, by better colony protection, making room for new colonies or moving colonies out of harms way. In the case of the Bass Rock gannets, this would seem impossible, but for gulls and particularly for terns, that easily shift from one breeding location to another, new colony sites may be developed and birds moved to places where collision risks are less severe (for an example, see: Leopold & Engels 2014).

Birds and bats may be 'taught' to avoid flying into wind farms, by developing warning or deterrring systems, using light (posssibly also outside the spectrum visible to humans), or sound (likewise). This would be new technology, in need of development for at-sea situations and the effectiveness, both short-term and long-term, are as yet unknown. Moreover, it should be noticed that deterrence could also increase habitat loss for seabirds, something that should be avoided.

## 6.4 Optimal wind farm configuration

With few studies available on the effects of different wind farms, and many different species possibly impacted, one should be very careful with statements about optimal configurations, both within wind farms (turbine configurations) or between wind farms. For instance, we do not know what would be best (and for which species): few large wind farms, or many smaller wind farms, more spread out. One idea for offshore wind farms and migrant birds would be to have "corridors" that can be used for safe passage. Such corridors should be lined up with dominant migration directions and will only be valuable if wind farms are large (but we do not know how large). Corridors between wind farms could have a similar benificial effect, but as yet this is speculative.

The interplay between turbine size, turbine spacing and thus the size of the footprint of offshore wind farms on one hand, and collision risks and displacement on the other, need more attention. Larger turbines need to be more widely spaced and this would seem beneficial for birds (e.g. Appendix D). If that the outer limits of future wind farms are set, the footprint of wind farms will not vary with turbine type, unless only part of a given lot will be built up. If, on the other hand, very large areas are designated for development, and developpers have some freedom to claim space within such areas, larger turbines may mean larger footprints.

## 6.5 Adaptive management

Given that projected cumulative mortalities of all future wind farms in the southern North Sea will remain below PBR, with the possible exception of some *Larus* species, according to the outcomes of our first modelling work, there appears to be no need to be careful about the sequence of building the wind farms. However, the estimates in this report are speculative to a considerable degree and should be tested as more and more wind farms will become operational. Should the mortality be greatly underestimated, PBR will be reached for a growing number of species in the course of further developments. We cannot emphasize enough that continuous monitoring of the effects of offshore wind farms is highly important, given the current uncentainties of mortality estimates for birds and bats.

## 6.6 Knowledge gaps

### *6.6.1 Birds*

Not all existing data could be incorporated in this study. This report should thus be seen as a first attempt to address this complex cumulation issue. Even if all survey efforts would have been incorporated, for some seasons unsurveyed areas remain and would require new survey effort; also some of the data used are already more than ten years old. Changing baselines, such as effects of climate change on population sizes and distribution patterns, have not yet been addressed and neither have several other factors that might impact at-sea birds numbers, simultaneously with offshore wind farm development. For migrant birds, the main uncertainties may lay in insufficient knowledge of the relevant catchment areas and the threats facing them elsewhere.

Even after about one decade of offshore wind farm impact studies, collisions of birds with turbine blades still have to be modelled without any existing data on actual collisions at sea. Model results are only as good as the model input and collision models could greatly benefit from actual at-sea determined collision rates. This would need technology to be developed, tested and installed in future wind farms that measure actual collisions for the different species of birds and bats (see e.g. Verhoef *et al*. 2002). From such data, collision models would greatly benefit and their outcomes would gain credibility.

Similarly, there are no studies that have measured the effects of displacement of seabirds from wind farms, on seabird fitness. For want of something better, Bradbury *et al*. (2014), and this study, have used a factor of 0.1 that translates the numbers of seabirds assumed to be displaced, to die. With strongly increasing proportions of marine surface area to become occupied by wind farms, proportion of displaced birds that actually die, might in fact not be a fixed figure, but is likely to gradually increase, unless habituation will occur. For the time being, we can only keep monitoring in order to find out what, if anything, is happening as little by little more wind farms are becoming operational.

Interestingly, displacement has been measured in several offshore wind farm inpact studies and the first results have become available. Results vary but may be related to a common factor, such as turbine density (this study). However, the carry-on effect of displacement on seabird fitness (or survival) is basically unknown and will need a lot of new reseach on seabird movement, habitat use and survival to explore. In contrast, the effects of collision on seabird (and landbird) survival are well-known (death), but the actual numbers of collisions have not been measured anywhere at sea, and have to be modelled. Both mortality factors thus suffer from considerable uncertainty levels and need much new 'hard data' to be assessed properly.

### *6.6.2 Bats*

The main knowledge gap is a reliable estimate of the number of fatalities at sea. Current model settings (parameters) are probably not applicable on marine areas, as behavioural characteristics at sea might differ from those on land. Additionally, model predictions cannot be validated because searches of fatalities at the open sea are not feasible. Therefore, there seems to be no other option than to investigate the actual number of fatalities at sea with thermal-imaging cameras. With this technique bat behaviour can be monitored and fatalities can be registered.

It will be necessary to establish several behavioural characteristics if models have to be developed to predict the number of fatalities based on the measured acoustic bat activity, for example:

- Flight and foraging heights
- Species-specific detection range
- To what extent bats use echolocation during migration over sea
- Echolocation characteristics in relation to the number of fatalities

At this moment it is not known whether there are specific migration routes across the North Sea. To assess the potential impact of offshore wind farms in areas with important flyways – if present - it is necessary to know the spatial pattern of bat occurrence at sea. At the same time, the temporal occurrence can be investigated and linked to weather conditions. The information obtained can be used in prediction models of bat presence at sea.

Another mayor knowledge gap is a reliable estimate of the sizes of the catchment populations. It is likely to assume that a large proportion of bats originates from countries such as Russia and Belarus. Population estimates from these countries are not available. In addition, there is insufficient kwowledge available on the life history characteristics of the Parti-coloured Bat.

# 7 Conclusions and recommendations

## 7.1 Birds

From the list of bird species considered in this study, the vast majority will suffer offshore wind farm mortalities within PBR, meaning that populations are sufficiently resilient to compensate for the losses. However, our calculations also indicated levels of mortality exceeding PBR in a few seabirds. These outcomes may be related to the clumped occurrence of these birds in the seabirds at sea counts used as input for this study, and this problem requires further scrutiny. Given these uncertainties, we cannot, at present, firmly conclude that any species of bird will be at serious risk of extinction by offshore wind farms in the southern North Sea. Additional mortality is likely to occur, but feed-back mechanisms in population ecology will largely compensate for the losses through collisions.

The effects of habitat loss through displacement are less certain and can be estimated only very roughly. With increasing numbers of wind farms put into place, however, this pressure will become larger each year, and the amount of remaining availabe space will progressively decrease. Moreover, shipping, when expelled from offshore wind farms, will become progressively more concentrated in wind farms-free space. In the most busy parts of the southern North Sea, such as the Belgian and Dutch parts, space may become a scarce commodity and the combined effects of offshore wind farm development and heavy shipping will probably increase at an accellerating rate. However, offshore wind farms are not planned in much of the area used mostly by some of the most vulnerable birds, such as loons, grebes and seaduck, while the most severely impacted areas have relatively low seabird densities. Auks, for instance, are much more numerous in the British sector, although some Dutch parts are also of international importance for these birds. These parts, however, have not yet been designated for future wind farm sites.

If habitat loss is the more structural component that might impact bird survival, this will only impact seabirds. Migrating landbirds and waterbirds are only likely to suffer from collisions. Seabirds are generally long-lived, and even when increased mortality would surpass PBR, this will only do so for a limited number of decades, i.e. the life span of the wind farms. Seabirds can run the wind farm gauntlet for many years, before above-PBR mortality will make them go extinct. Even so, the concept of increased mortality is add odds with prevailing aims and targets (under the Birds Directive), that demand from EU member states an explicit effort to conserve the actual conservation states of all migratory bird species and all species named in the relevant Annexes. This basically means that all seabird species should be conserved. Also for ethical reasons, all mortality from man-made structures is undesirable, particularly if bird numbers will be structurally reduced, for several decades.

Given that wind farms will be built progressively over time, it will be very important to put monitoring into place, both of at-sea mortality and of the carry-on effects of habitat loss on bird populations. Such monitoring will be a powerful instrument to either learn that effects of offshore wind farms truly remain within safe biological limits, or to discredit this notion. Should the latter be the case, it should be possible to adapt our strategy of renewable energy development.

## 7.2 Bats

The species that seem to migrate most regularly over the southern North Sea, Nathusius' Pipistrelle, Noctule and Parti-coloured Bat, also appear to be the most sensitive species in relation to the development of offshore wind energy. Future monitoring should therefore target these species in particular, both at sea and on land (proper asssessments of catchment population regions and sizes).

Without good data on both collision rates and catchment population sizes, we cannot, at this stage, reliably answer the question of whether increased mortality caused by offshore wind turbines can or will significantly affect bat populations. Based on the current knowledge, it is not possible to rule out that in a worst case scenario cumulative numbers of fatalities might seriously impact some bat populations.

Based on the assumptions made, the least impact is to be expected for Noctule. The knowledge gaps in Nathusius' Pipistrelle regarding the size of the catchment populations prevent any reliable evaluation of the number of an estimated worst case scenario. The same applies to Parti-coloured Bat of which even less data are available regarding the size of the catchment populations and the life history characteristics.

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# 9 Quality Assurance

IMARES utilises an ISO 9001:2008 certified quality management system (certificate number: 124296-2012-AQ-NLD-RvA). This certificate is valid until 15 December 2015. The organisation has been certified since 27 February 2001. The certification was issued by DNV Certification B.V. Furthermore, the chemical laboratory of the Fish Division has NEN-EN-ISO/IEC 17025:2005 accreditation for test laboratories with number L097. This accreditation is valid until 1th of April 2017 and was first issued on 27 March 1997. Accreditation was granted by the Council for Accreditation.

# Justification

Report : C166/14
Project number : 43061253.01

The scientific quality of this report has been peer reviewed by the a colleague scientist and the head of the department of IMARES.

Approved: Dr. R.G. Jak
Senior Researcher

Signature:

Date: 15 January 2015

Approved: Drs. J. Asjes
Head of the Department of Ecosystems

Signature:

Date: 15 January 2015

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| Annex D | Annex D1-D6, being Appendix 1-6 of Bureau Waarden document “Cumulative effects of collisions in birds of offshore wind farms North Sea” of Bureau Waardenburg”. |
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