

Final Report

Divers (*Gavia* spp.) in the German North Sea: Changes in Abundance and Effects of Offshore Wind Farms

A study into diver abundance and distribution
based on aerial survey data in the German North Sea



Authors:

Raul Vilela, Claudia Burger ¹, Ansgar Diederichs ¹, Georg Nehls ¹

Fabian Bachl ^{1, 2},

Lesley Szostek ³, Anika Freund ³, Alexander Braasch ³,

Jochen Bellebaum ⁴, Brian Beckers ⁴, Werner Piper ⁴,

¹ BioConsult SH GmbH & Co. KG (mainly responsible for the report)

² School of Mathematics, University of Edinburgh, Edinburgh, Scotland

³ IBL Umweltplanung GmbH

⁴ Institut für angewandte Ökosystemforschung GmbH

BioConsult SH GmbH & Co KG

Schobüller Str. 36
25813 Husum

Tel.: 04841 77937 -10
www.bioconsult-sh.de

IBL Umweltplanung GmbH

Bahnhofstraße 14a
26122 Oldenburg

Tel.: 0441 505017-10
www.ibl-umweltplanung.de

**Institut für Angewandte
Ökosystemforschung GmbH**

Schulterblatt 120
20537 Hamburg

Tel.: 040 432139000
www.ifaoe.de

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Divers in the German North Sea: Changes in Abundance and Effects of Offshore Wind Farms on Divers (*Gavia spp.*)

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1 SUMMARY

With the increasing expansion of offshore wind farms in the German North Sea concerns have been raised about the protection and preservation of the winter and spring resting populations of the two diver species, red-throated (*Gavia stellata*) and black-throated divers (*Gavia arctica*). The German North Sea is an important habitat for both species. As various studies suggest a stronger response of divers to offshore wind farms than previously expected and due to the high conservation status of these two species, there is a strong interest in assessing whether the development of diver populations is influenced by the expansion of wind power.

The present study is based on a high-quality data set on the distribution and occurrence of divers in the German North Sea over the last 18 years and represents the most extensive data set currently available. For spring, 16 years of data were available (no data for 2006 and 2007), for winter, 17 years were available (2006 missing). This data set considers only high-quality aerial survey data and not data from ship surveys, which is considered less suitable for the detection of divers. Through the application of a Bayesian spatial and spatio-temporal model approach, which is well suited for the complex problem of quantifying diver displacement, a reliable analysis of the development of diver populations between 2001 and 2018 is possible. Furthermore, for the first time, data on diver abundance during the winter season were also analysed in addition to spring data.

The main aim was to analyse trends in diver populations over the last 18 years, before and after the expansion of offshore wind farms (OWF) in the German North Sea. In addition, we wanted to find out up to which distance divers are displaced by offshore wind farms and what theoretical habitat loss might result from this, as well as quantifying possible variation between sub-areas and seasons.

The four key aspects were:

1. Reliable calculation of diver population size over the 18-year study period for the entire German North Sea as well as for a northern and a southern sub-area and examination of whether changes in the population might be related to the expansion of offshore wind farms.
2. Investigation of the spatial distribution of divers in relation to the location of offshore wind farms, taking into account seasonal (spring/winter) and local (north/south) factors.
3. Reliable analysis of the displacement distance of divers from offshore wind farms, taking into account seasonal (spring/winter) and local (north/south) factors.
4. Calculation of a theoretical habitat loss based on the current dataset.

The main result of this study is that over the entire study period, the spring abundance of divers fluctuated between individual years without any clear trend, with overall stable population numbers between 2001 and 2018. No connection with the expansion of wind power in the German North Sea and the inter-annual variability in diver abundance was found. In spring, divers reached the highest numbers, and on average 16,500 divers were estimated in the German North Sea. The northern part of the German North Sea and the main concentration area therein, as defined by the

BMU (2009), are of the greatest importance here, as it accounts for approx. 60 % of the German North Sea population over all the years examined. The abundance also fluctuated in the North and South sub-areas without a decline being observed with the expansion of offshore wind power. In winter, there was an increase in the population over the years, especially for the southern sub-area. However, compared to spring, there were significantly fewer divers in the German North Sea in winter ($n = 4.833$).

After the expansion of offshore wind farms in the northern part, a less variable distribution of divers was observed. The birds concentrated relatively consistently in a central area of the main concentration area, which has the longest possible distance to all surrounding wind farms. However, this area was also used by divers before construction of the first wind farms, with high densities in some years (e.g. 2003, 2010). Despite this partial redistribution, the number of divers in the main concentration area is not declining and the area is still very important for resting divers in spring. In winter, the spatial distribution is more variable and the birds are more likely to be much nearer to the coast and in significantly lower densities overall.

The results showed that divers keep different distances from the wind farms, depending on season (spring/winter) and area (north/south). The most reliable calculation of the avoidance distance is based on spring data, as this is the period with by far the highest density of divers. In spring, a displacement distance (gradient) of 10.2 km was calculated for the entire study area and for all available data which therefore forms the most robust result (see Box 1 for details on calculation). In the two sub-areas North and South, slightly shorter distances were calculated. Due to a significantly flatter displacement curve in the southern area, where less than 20 % of the spring population of divers were present, a theoretical habitat loss of 2 km (radius around a model OWF) was calculated here, while a theoretical habitat loss of 5 km was calculated for the northern area (comprising >75 % of the diver population in spring) and for the entire data set.

In winter, large differences in the displacement distance to offshore wind farms were observed between the northern and southern sub-area, potentially due to the considerably lower diver densities and the resulting greater uncertainties in the analyses. Nevertheless, these differences show that seasonal and spatial factors may play a role in the specific response of divers to offshore wind farms and results found here are therefore not directly transferable to areas other than those considered in this study.

The shape of the wind farm footprint has only a minor effect on the theoretical habitat loss. The total number of divers displaced can be reduced by combining individual OWF projects into clusters so that displacement radii overlap.

In addition to the Bayesian distance model, a before-after approach was also applied for the northern sub-area to calculate avoidance-distances. The results for two different before periods (2001-2005 and 2008-2011) show larger displacement distances (11 and 13 km, respectively), but also reveal uncertainties regarding the reference period to be chosen. A comparison of the diver distributions before and after the wind farm extension shows, depending on the selected reference period, areas of different sizes with significant diver decrease and increase, which are not circular around the respective wind farms. There is limited knowledge about the mechanisms behind the displacement effect and so possible physiological consequences and subsequent possible effects

on long-term population development cannot be assessed. But it is apparent, however, that the local population within the German North Sea is stable during the time period analysed.

Definitions:

Displacement distance:

The displacement distance is defined as the distance from the edge of an offshore wind farm up to which diver density is significantly lower than a reference density defined as the overall mean of the dataset used in the specific model (years 2009 – 2018). The displacement effect is a gradient, with lower bird densities close to the OWF and increasing densities until the reference density is reached.

Theoretical habitat loss:

The theoretical habitat loss is defined as the area corresponding to the habitat of birds that is theoretically no longer available for use and is given as a radius around an OWF. Since the displacement effect is a gradient while the theoretical habitat loss assumes a total loss of the area for divers, the radius of the theoretical habitat loss around an OWF is far lower than the displacement distance.

Main findings:

1. Over the study period (2001 - 2018), the spring abundance of divers was stable but showed inter-annual fluctuations without any clear trend. No connection was found between diver abundance and the expansion of wind power in the German North Sea. In spring, divers reached the highest numbers and an average abundance of 16,500 divers was estimated for the German North Sea.
2. After the expansion of the offshore wind farms in the northern part of the German North Sea, a less variable distribution of divers was observed. The birds concentrate relatively constantly in a central area of the main concentration area. The number of divers in the main concentration area is not declining and the area is still very important for resting divers in spring. No indication was found that the carrying capacity limit within the main concentration area has been reached. In winter, the spatial distribution is more variable in the German North Sea and the birds are more likely to be much nearer to the coast and occur in significantly lower densities overall.
3. The results showed that divers keep different distances from the wind farms, depending on different seasons (spring/winter) and areas (north/south). The most reliable result for avoidance distance is based on spring data, when a displacement distance (gradient) up to 10.2 km was calculated for the entire study area and for all available data. Slightly smaller distances were found in the two sub-areas North and South. A before-after approach was also applied for the northern sub-area and spring data to calculate displacement distances.

The results showed larger displacement distances (between 11 km and 13 km), but also revealed uncertainties regarding the reference period to be chosen.

4. A theoretical habitat loss of 5 km (radius around a model OWF) was calculated for the entire study area and for all available spring data. For the southern sub-area, in spring a theoretical habitat loss was calculated at only 2 km as a result of the considerably flatter displacement curve, while a theoretical habitat loss of 5 km was calculated for the northern part. Although part of this difference is due to a lower density of divers in the southern part, this large difference indicates that there are regional differences in the response of divers to offshore wind farms. Therefore, the available results can only be transferred to other areas outside the study area to a very limited extent and need to be tested on a case by case basis.

2 ZUSAMMENFASSUNG

Durch den fortschreitenden Ausbau von Offshore-Windparks in der deutschen Nordsee gerät der Schutz und Erhalt der Winter- und Frühjahrsrastbestände der beiden Seetaucherarten Stern- und Prachtaucher zunehmend in den Fokus. Für beide Arten stellt die deutsche Nordsee einen wichtigen Lebensraum dar. Verschiedene Studien zeigen, dass Seetaucher mit deutlicher Meidung auf Offshore-Windparks reagieren und es besteht aufgrund des hohen Schutzstatus beider Arten ein großes Interesse daran, zu prüfen, ob die Entwicklung der Seetaucherp Populationen durch den Ausbau der Windkraft beeinflusst ist.

Die vorliegende Studie baut auf einen qualitativ hochwertigen Datensatz über die Verbreitung und das Vorkommen von Stern- und Prachtauchern in der deutschen Nordsee in den letzten 18 Jahren auf, der den derzeit umfangreichsten verfügbaren Datensatz darstellt. Für den Frühling waren Daten für 16 Jahre verfügbar (keine Daten für 2006 und 2007), für den Winter waren 17 Jahre verfügbar (keine Daten für 2006). Es wurden ausschließlich Daten aus Flugzeugerfassungen berücksichtigt und die für die Erfassung von Seetauchern aufgrund ihrer deutlichen Reaktion auf herannahende Schiffe weniger geeigneten Daten aus Schiffserfassungen ausgeschlossen. Die Anwendung eines bayesianischen räumlichen und raum-zeitlichen Modells, welches sehr gut geeignet ist, die komplexen Zusammenhänge von Seetaucher-Meideverhalten zu berechnen, ermöglichte eine zuverlässige Analyse der Populationsentwicklung von Seetauchern zwischen 2001 und 2018. Darüber hinaus werden in dieser Studie neben dem Frühjahr auch erstmals Daten zur Seetaucherdichte während der Wintersaison vorgestellt.

Das Hauptziel war, die Entwicklung der Seetaucherp Population auf der Basis des umfangreichsten verfügbaren Datensatzes in der deutschen Nordsee in den letzten 18 Jahren vor dem Hintergrund des zunehmenden Ausbaus der Offshore Windkraft zu analysieren. Darüber hinaus sollte herausgefunden werden, bis zu welcher Entfernung Seetaucher Offshore-Windparks meiden, welcher theoretische Lebensraumverlust daraus berechnet wird und ob es Unterschiede zwischen Teilgebieten und Jahreszeiten gibt.

Die vier Hauptaspekte dieser Untersuchung waren:

1. Eine verlässliche Berechnung der Seetaucherp Populationsgröße über den 18-jährigen Untersuchungszeitraum für die gesamte deutsche Nordsee sowie für ein nördliches und ein südliches Teilgebiet und Prüfung eines möglichen Zusammenhangs zwischen Veränderungen in der Population und dem Ausbau von Offshore-Windparks.
2. Die Untersuchung der räumlichen Verteilung von Seetauchern unter Berücksichtigung der Standorte von Offshore-Windparks und unter Berücksichtigung saisonaler (Winter/Frühling) und räumlicher (Nord/Süd) Faktoren.
3. Eine zuverlässige Analyse zur Verbreitungsreichweite von Seetauchern durch Offshore-Windparks unter Berücksichtigung saisonaler (Frühjahr/Winter) und räumlicher (Nord/Süd) Faktoren.
4. Die Berechnung eines theoretischen Habitatverlusts auf der Basis des aktuellen Datensatzes.

Das Hauptergebnis der vorliegenden Studie ist, dass die Frühjahrsbestände der Seetaucher über den gesamten Studienzeitraum zwischen den Jahren ohne erkennbaren Trend fluktuieren und die Populationsgröße über den gesamten Zeitraum hinweg stabil ist. Die jährlichen Schwankungen können nicht mit dem Ausbau der Windkraft in der deutschen Nordsee in Zusammenhang gebracht werden. Seetaucher erreichen im Frühjahr höchste Bestandszahlen und es wurde eine mittlere Anzahl von 16.500 Seetauchern im Frühjahr in der deutschen Nordsee berechnet. Dabei nimmt das nördliche Teilgebiet und hierin das vom BMU (2009) definierte Hauptkonzentrationsgebiet die größte Bedeutung ein, da sich in diesem Hauptkonzentrationsgebiet konstant über alle untersuchten Jahre ca. 60 % des deutschen Nordseebestands aufhalten. Auch in den beiden Teilgebieten Nord und Süd schwankten die Bestände, ohne dass ein Zusammenhang mit dem Ausbau der Offshore Windkraft festgestellt werden kann. Im Winter ergab sich vor allem für das südliche Teilgebiet ein Anstieg in den Bestandszahlen über die Jahre. Jedoch halten sich im Vergleich zum Frühjahr im Winter insgesamt deutlich weniger Seetaucher in der deutschen Nordsee auf ($n = 4.833$).

Die Verteilung der Seetaucher stellt sich im nördlichen Teilbereich nach dem Ausbau der Offshore-Windparks seit ca. 2013 weniger variabel dar. Die Vögel konzentrieren sich relativ konstant in einem zentralen Bereich des Hauptkonzentrationsgebietes, der die größtmögliche Entfernung zu allen umliegenden Windparks aufweist. Dieser Bereich wurde auch bereits vor dem Bau der ersten Windparks in z. T. hohen Dichten von Seetauchern genutzt (z. B. 2003, 2010). Der Seetaucherbestand im Hauptkonzentrationsgebiet ist trotz dieser geringeren Variabilität in der Verteilung nicht rückläufig und das Gebiet ist nach wie vor im Frühjahr für rastende Seetaucher von sehr hoher Bedeutung. Im Winter ist die räumliche Verteilung variabler und die Tiere halten sich häufiger deutlich küstennäher und insgesamt in deutlich niedrigeren Dichten in der deutschen Nordsee auf.

Die Ergebnisse zeigen, dass Seetaucher unterschiedliche Meideabstände zu den Windparks einhalten, abhängig von Jahreszeit (Frühjahr/Winter) und Gebiet (Nord/Süd). Die zuverlässigste Berechnung von Meideabständen basiert auf den Frühjahrsdaten, da dieser Zeitraum die bei weitem höchste Dichte an Seetauchern aufweist. Die Analyse basiert auf allen verfügbaren Daten aus dem Frühjahr für das gesamte Untersuchungsgebiet und kommt zu einem Meideabstand (Gradient) von 10,2 km. In den beiden Teilgebieten Nord und Süd wurden geringfügig kleinere Meidedistanzen ermittelt.

Für den Gesamtdatensatz errechnet sich ein theoretischer Habitatverlust im Frühjahr von 5 km (Radius um einen modellhaften OWP). Aufgrund eines deutlich flacheren Kurvenverlaufs berechnet sich der theoretische Habitatverlust für den südlichen Teilbereich, in dem <20 % des Frühjahrsbestand vorkommen, auf 2 km, während er im nördlichen Teilbereich, in dem >75 % des Seetaucher-Frühjahrsbestands vorkommen auf 5 km berechnet wird. Im Winter wurden große Unterschiede im Meideabstand zu Offshore-Windparks zwischen dem nördlichen und südlichen Teilgebiet festgestellt, die teilweise auf die deutlich geringeren Seetaucherdichten und die dadurch größeren Unsicherheiten in der Berechnung zurückzuführen sind. Dennoch zeigen diese Unterschiede, dass saisonale und räumliche Faktoren bei der spezifischen Reaktion von Seetauchern auf Offshore-Windparks eine Rolle spielen können. Die hier gefundenen Ergebnisse sind daher nur eingeschränkt auf andere Gebiete außerhalb des Untersuchungsgebietes dieser Studie übertragbar und müssen von Fall zu Fall überprüft werden.

Die Form der Windpark-Grundfläche hat dabei nur geringe Auswirkungen auf den theoretischen Habitatverlust. Die Gesamtzahl der betroffenen Seetaucher kann dadurch reduziert werden, dass einzelne OWP-Vorhaben zu Clustern zusammengefasst werden, so dass sich die Meideabstände überlagern.

Zusätzlich zum bayesianischen Distanzmodell wurde auch ein Vorher-Nachher-Ansatz für das nördliche Teilgebiet zur Berechnung von Meideabständen angewendet. Das Ergebnis bezüglich zweier unterschiedlicher Vorher-Zeiträume (2001-2005 und 2008-2011) zeigt größere Meideabstände (11 km bzw. 13 km), jedoch deckt es auch Unsicherheiten bezüglich des zu wählenden Referenz-Zeitraums auf. Ein Abgleich der Seetaucherverteilungen vor und nach dem Windparkausbau zeigt abhängig vom gewählten Referenzzeitraum unterschiedlich große Gebiete mit signifikanter Seetaucherabnahme und Zunahme, die nicht kreisrund um die jeweiligen Windparks liegen. Das weist darauf hin, dass die Mechanismen, die den Meideeffekt bei den Vögeln auslösen, noch weitgehend unbekannt sind und daher keine Aussagen zu physiologischen Konsequenzen dieser Reaktion und eventuell daraus resultierenden Langzeit-Effekten möglich sind. Die vorliegende Studie kann aber nachweisen, dass der lokale Seetaucherbestand im Bereich der deutschen Nordsee über den betrachteten Zeitraum trotz des Ausbaus der Offshore Windindustrie stabil bleibt.

Begriffsdefinitionen:

Meideabstand:

Der Meideabstand ist definiert als diejenige Distanz vom Rand des OWPs bis zu der die Dichte der Seetaucher signifikant geringer ist als ein Referenzwert, der als die mittlere Dichte des gesamten Datensatzes in diesem Modell berechnet wird (Jahre 2009-2018). Der Meideeffekt ist ein Gradient, mit geringsten Dichten am Rande des OWP und ansteigenden Dichten bis zur Referenzdichte.

Theoretischer Habitatverlust:

Der theoretische Habitatverlust ist definiert als die Fläche, auf der die durch den Meideabstand (s. o.) berechnete Anzahl vertriebener Seetaucher vorkommen würde, wenn die Referenzdichte als Basis genommen wird. Diese theoretisch als Habitat nicht mehr nutzbare Fläche wird als Radius um den OWP angegeben. Da die Meidung graduell bis zum maximalen Meideabstand abnimmt, der theoretische Habitatverlust aber einen vollständigen Flächenverlust annimmt, ist der Radius des theoretischen Habitatverlustes um den OWP deutlich geringer als der Meideabstand.

Hauptergebnisse:

1. Über den Studienzeitraum (2001 – 2018) waren die Frühjahrsbestände der Seetaucher in der deutschen Nordsee insgesamt stabil bei Schwankungen zwischen einzelnen Jahren ohne erkennbaren Trend. Es wurde kein Zusammenhang zwischen dem Seetaucher-Bestand und dem Ausbau der Windkraft festgestellt. Seetaucher erreichen im Frühjahr

höchste Bestandszahlen und es wurde eine mittlere Anzahl von 16.500 Seetauchern im Frühjahr in der deutschen Nordsee berechnet.

2. Nach dem Ausbau der Offshore-Windparks im nördlichen Teilbereich wurde eine weniger variable Verteilung der Seetaucher festgestellt. Die Vögel konzentrieren sich relativ konstant in einem zentralen Bereich des Hauptkonzentrationsgebietes. Der Seetaucherbestand im Hauptkonzentrationsgebiet ist trotz dieser geringeren Variabilität in seiner Verteilung nicht rückläufig und das Gebiet ist weiterhin im Frühjahr von sehr hoher Bedeutung für rastende Seetaucher. Die Daten geben keine Hinweise darauf, dass die Tragfähigkeitsgrenze für Seetaucher im Hauptkonzentrationsgebiet erreicht ist. Im Winter ist die räumliche Verteilung variabler und die Tiere halten sich deutlich küstennäher und insgesamt in deutlich niedrigeren Dichten in der deutschen Nordsee auf.

3. Seetaucher halten unterschiedliche Meideabstände zu den Windparks ein, abhängig von verschiedenen Jahreszeiten (Frühjahr/Winter) und Gebieten (Nord/Süd). Das zuverlässigste Ergebnis für einen Meideabstand (Gradient) basiert auf allen verfügbaren Daten aus dem Frühjahr und für das gesamte Untersuchungsgebiet und kommt auf 10,2 km. In den beiden Teilgebieten Nord und Süd wurden geringfügig geringere Distanzen festgestellt. Ein Vorher-Nachher-Ansatz für das nördliche Teilgebiet im Frühjahr zeigte größere Meideabstände (zwischen 11 km und 13 km), der Ansatz zeigt jedoch Unsicherheiten abhängig vom jeweils verwendeten Vorher-Zeitraum.

4. Für das gesamte Untersuchungsgebiet und für alle verfügbaren Daten aus dem Frühjahr wurde ein theoretischer Habitatverlust von 5 km (Radius um einen Modell-OWP) berechnet. Aufgrund eines deutlich flacheren Verlaufs der Vertreibungskurve ergibt sich für den südlichen Teil ein theoretischer Habitatverlust von 2 km (Frühjahr), während für den nördlichen Teil ein theoretischer Habitatverlust von 5 km (Frühjahr) berechnet wurde. Obwohl ein Teil dieses Unterschieds auf eine geringere Dichte von Seetauchern im südlichen Teil zurückzuführen ist, zeigt dieser Unterschied, dass es regionale Unterschiede in der Reaktion von Seetauchern auf Offshore-Windparks gibt. Die vorliegenden Ergebnisse sind daher nur sehr begrenzt auf andere Gebiete außerhalb des Untersuchungsgebietes übertragbar und müssen im Einzelfall geprüft werden.

3 INTRODUCTION

This report investigates the effects of large-scale offshore wind farm developments in the German North Sea on the diver species red-throated diver (*Gavia stellata*) and black-throated diver (*Gavia arctica*), two arctic breeding, protected seabird species which winter in European coastal waters. The analysis is based on a large, unique dataset from aerial surveys which have been accomplished as part of the monitoring program which has been made mandatory by German regulator BSH for all offshore wind farms in order to gain insight into the effects on marine wildlife to inform further planning of the rapidly developing industry.

As part of a concerted effort to reduce the reliance on fossil fuels and nuclear energy (EEG 2017, 2014), the installation and operation of offshore wind farms has been expanding in the German North Sea since the first German offshore wind farm alpha ventus was installed in 2009, consisting of 12 turbines with a capacity of 5 MW. By early 2019, the number of turbines within completed wind farms amounted to 1,052 with a further 152 turbines currently under construction, many of which will be far larger than the early models, with capacities now exceeding 8 MW¹.

With the growing numbers and sizes of wind turbines within the German North Sea, concerns have been raised about possible impacts on those bird populations that rely on the North Sea as their permanent or migratory habitat. Factors that might affect birds include visual and acoustic disturbance due to unfamiliar land-like structures, rotor movements, increased ship traffic as well as mortality due to collision with turbines.

Members of the diver family are among the seabird species most susceptible to disturbance in the North Sea (e.g. Dierschke et al. 2012, 2016). The two diver species that most commonly occur in this area, red-throated diver and black-throated diver, are subject to special conservation measures under German and EU law. Both diver species are listed in Annex I of the EU Birds Directive (Europäisches Parlament und Rat der Europäischen Union 2013) as species of special conservation concern, as well as in the Agreement on the Conservation of African-Eurasian Migratory Waterbirds (UNEP / AEWA Secretariat 2016). A Special Protection Area and bird reserve (SPA Östliche Deutsche Bucht) was established within the European Natura 2000 network in the Eastern part of the German Bight where divers show their highest density (BMUB 2017). However, and despite a globally decreasing population trend, the species are not threatened on a global scale due to a wide distribution and large populations in some areas (BirdLife International 2017).

Both species of divers occur in the German North Sea from autumn to spring. The highest numbers of individuals are observed during spring migration (Mendel et al. 2008). Divers that winter in or migrate through the German Bight belong to the North-west European wintering population. Reliable estimates as to population size are hard to find (see Dierschke et al. 2012 for overview), but it is estimated that around 90,000 red-throated divers and around 31,250 black-throated divers (BirdLife International 2004) constitute the European winter population, of which an estimated 18 % and 6 % respectively migrate through the German North Sea during spring (Dierschke et al. 2012). Estimates of diver population size for the German North Sea itself range from a population

¹ BSH: https://www.bsh.de/DE/THEMEN/Offshore/Offshore-Vorhaben/Windparks/windparks_node.html (accessed: 31.07.2019)

size of ca. 20,000 individuals for the period of 2002-2013 (Garthe et al. 2015) to a more recent estimate of ca. 35,000 individuals for the period before and ca. 25,000 individuals for the period after offshore wind farm development commenced (Garthe et al. 2018). However, the recent estimate was based on a model mainly investigating the displacement effect and not covering the whole Exclusive Economic Zone (EEZ). A trend analysis over the same timeframe and area as the previous studies estimated population sizes for red-throated diver for the German North Sea of between 3,200 and 31,000 individuals from 2002 to 2017 with a decreasing trend since 2013 (Schwemmer et al. 2019).

Divers show avoidance behaviour towards vessels at distances of clearly more than one kilometre (Bellebaum et al. 2006, Fliessbach et al. 2019) and show reduced densities in areas of high ship traffic (Schwemmer et al. 2011, Burger et al. 2019). Many studies have been conducted into reactions of divers to offshore wind farms and consistently reported avoidance behaviour towards the wind farm itself and lower sighting rates within a certain buffer zone around the wind farm (e.g. Dierschke et al. 2012, 2016). However, there are strongly varying estimates of how far from the wind farm a displacement effect is noticeable. Early studies found comparatively short displacement distances. At the Danish North Sea wind farms Nysted and Horns Rev I, a significantly lower encounter rate of divers was found within 2 km of the wind farm (Petersen et al. 2006), or 4 km in a different study (Petersen & Fox 2007). Some British wind farms also showed short distances of displacement: at the wind farm Kentish Flats a maximum of 1 km was estimated, however, this might be because the observed area around the wind farm was smaller than in other studies (Percival 2014). A study at the first German offshore wind farm alpha ventus described similarly moderate displacement distances of at least 1.5 km (Welcker & Nehls 2016).

More recent studies reported wider-reaching effects. At Horns Rev II a displacement distance of 5 - 6 km was estimated (Petersen et al. 2014), while a similar effect of 2 - 6 km was found at the British wind farm Lincs (Webb et al. 2015). Since 2015 based on digital aerial surveys, the monitoring programs during offshore wind farm developments have revealed a significantly greater avoidance distance than previously assumed. Based on digital aerial high definition video (HiDef) surveys around several wind farms in the area of highest diver density in the German North Sea, Heinänen et al. (2016) estimated displacement effects ranging over more than 10 km. Similar estimates of 9 - 12 km were made by Garthe et al. (2018) when using baseline data as reference value, while Mendel et al. (2019) estimated an effect range of up to 16 km as evaluated from the post-construction period of the dataset. Due to this strong avoidance reaction and their preferred low flight height, however, divers are not considered at risk of mortality due to collision with wind turbines (Cook et al. 2012, Furness et al. 2013).

Although individual divers are regularly but very rarely observed inside wind farms, so far, no clear signs of habituation have been observed. At the British wind farm Kentish Flats the magnitude of the displacement seemed to lessen in the fourth and fifth year of monitoring (Percival 2010), but similar observations have not been made at other wind farms.

Based on early displacement estimates, the German Federal Maritime and Hydrographic Authority assumed a 2 km radius of theoretical habitat loss around a new offshore wind farm (BSH 2008), which was calculated by summing up the (estimated) number of birds displaced due to the wind farm and calculating the area of habitat they would have occupied based on baseline data ('theoretical habitat loss'). This approach has first been adopted for the permitting of the Butendiek

offshore wind farm in 2002 (BSH 2002) and used as a basis for all following projects since then. However, based on newer estimates of displacement radii, a more recent study suggests a higher theoretical habitat loss of around 5.5 km (Garthe et al. 2018).

It is a concern that the expansion of wind farms, especially in the area of highest springtime density, will affect population numbers through habitat loss or displacement into suboptimal habitats or knock-on effects on breeding success. In a long-lived seabird family like the divers, effects on breeding success can take many years to manifest at population level. Yet, no mitigating measures exist so far to reduce the potential impact of wind farms on divers. Due to possibly strong impacts and as a consequence of the more recent estimates of displacement and habitat loss, the German Federal Maritime and Hydrographic Authority (BSH) has adjusted the offshore energy development strategy: It has issued a stop to new offshore wind farm developments around the SPA Östliche Deutsche Bucht and the area of highest diver density (BMU 2009) and is reviewing (and in one case has excluded) the possibility of repowering for several wind farms in the area, while pursuing the expansion of wind energy generation in areas less relevant to divers (BSH 2019).

As biological systems are always a complex interplay of many factors, population trends are difficult to measure, calculate and predict. The density and distribution of divers is likely dependent on multiple factors besides the effects of offshore wind farms, which makes it difficult to isolate their effects. A new statistical framework, based on Bayesian spatial modelling, was developed specifically to deal with spatial distribution and variation in ecological data under the inclusion of prior knowledge. This modelling framework, called Latent Gaussian Models (LGMs), uses the Integrated Nested Laplace Approximation (INLA) to realistically describe complex spatial relationships, while accounting for spatio-temporal interdependence and autocorrelation in the data. It is a very flexible modelling approach which can integrate environmental variables. It is therefore an ideally suited method for the complex problem of quantifying diver displacement due to offshore wind farms.

During the last years, Bayesian methods have developed greatly and are now widely established in many research areas. The basic idea behind the Bayesian approach is that effectively only one form of uncertainty exists, which is described by suitable probability distributions. Thus, there is no fundamental distinction between observable data or unobservable parameters, which are also considered as random quantities (Blangiardo & Cameletti 2015). The inferential process combines the prior and observed data to derive the posterior distribution (Bernardo et al. 2000, Lindley 2006).

The Integrated Nested Laplace Approximation (INLA; Rue et al. 2009) approach was developed as a computationally efficient alternative to Markov Chain Monte Carlo (MCMC) methods (Robert & Casella 2010, Brooks 2011). INLA is designed for latent Gaussian models, a very wide and flexible class of models, and this approach has been successfully used in a great variety of applications thanks to the availability of the R-INLA package for R software (Martino & Rue 2008) and more recently the R package inlabru (Bachl et al. 2019) specifically developed to model spatial distribution and change from ecological survey data.

Furthermore, INLA can be combined with the Stochastic Partial Differential Equation (SPDE) approach proposed by Lindgren et al. (2011) in order to implement spatial and spatio-temporal models for point reference data. The SPDE approach consists in representing a continuous spatial process, e.g. a latent stationary Gaussian Field (GF) with the Matérn covariance function as a

discretely indexed spatial random process (e.g. a Gaussian Markov Random Field (GMRF); Rue & Held 2005). This approach is computationally efficient while accounting for spatio-temporal interdependence and autocorrelation in the data. The INLA-SPDE is therefore a very flexible modelling approach which can integrate environmental variables, and it is an ideally suited method for the complex problem of quantifying diver displacement due to offshore wind farms.

As a condition for receiving consent for offshore wind farms in the EEZ of the German North Sea, an extensive monitoring program pre-, during and post-construction is to be conducted, which include regular aerial surveys for seabirds. As a result, there is an extensive dataset of diver distribution from different areas in the German North Sea which by now spans over a period of 18 years (2001-2018) with data available for 16 years (spring) and 17 years (winter), respectively. Two types of aerial surveys were conducted: visual aerial surveys with two or three observers on board the airplane until 2013 and digital aerial surveys since 2014, with some overlap between these types. The analysis includes data from all major offshore wind farm projects within the German North Sea, as well as some additional monitoring programs. Using this extensive dataset will give a better understanding of the effects of offshore wind farms (OWF) on diver displacement in the German North Sea.

In this study, the following four key aspects will be addressed:

1. Reliable calculation of diver population size over the 18-year study period for the entire German North Sea as well as for a northern and a southern sub-area and examination of whether changes in the population might be related to the expansion of offshore wind farms.
2. Investigation of the spatial distribution of divers in relation to the location of offshore wind farms, taking into account seasonal (spring/winter) and local (north/south) factors.
3. Reliable analysis of the displacement distance of divers from offshore wind farms, taking into account seasonal (spring/winter) and local (north/south) factors.
4. Calculation of a theoretical habitat loss based on the current dataset.

4 METHODS

4.1 Data

For the time period between 2001 and 2018, aerial survey data from the German North Sea were analysed. All surveys included in the present study took place between 1.11 and 15.5 of each year and were assigned to the species-specific seasons "Winter" (1.11 - 28.02) and "Spring" (1.3 - 15.5) as defined by Garthe et al. (2007). The flights were carried out on different dates depending on the weather, so there is a varying amount of effort behind the seasons of each year. Ship survey data was not included for several reasons: The main issue with ship survey data is that divers respond to the approaching survey vessel very early by flying up or diving, at distances of up to a few kilometer in front (Bellebaum et al. 2006, Burger et al. 2019). This makes numbers estimated by ship surveys less certain and hardly comparable with aerial survey data. Furthermore, due to the slow survey speed and the high dependence of the detection probability on the sea state, a high error correction is necessary (Garthe et al. 2015), so that a reliable density calculation is connected with very large uncertainties for these species. For these reasons, ship survey data has been omitted altogether for this study.

In total, 287 days of digital surveys were available. Out of these, 88 days of surveys were conducted during spring, 53 during winter. The remaining 146 days, conducted during the rest of the year, were not included. Regarding conventional (visual) surveys, 69 surveys were conducted during winter and 56 during spring. In total, 34,000 divers were observed during all surveys. Data sources comprised data from wind farm monitoring (~ 80 %), Natura2000 Monitoring (FTZ, 15 %), research projects (~ 5%) and other sources (< 5 %). Details about specific projects can be found in the appendix (Table -10 and Table -11). For spring, 16 years of data were available (no data for 2006 and 2007), for winter, 17 years were available (2006 missing).

For divers, identification on species levels is rather difficult using aerial surveys. For both methods (digital and visual) a significant part of all individuals was only identified as diver spp. Analyses were therefore conducted including all individuals observed. However, from previous studies (e.g. Mendel et al. 2008, Garthe et al. 2015) and from the wind farm monitoring projects it is known that the majority of divers (~ 90 %) in this area are red-throated divers (*Gavia stellata*).

4.2 Study Area Wind Farm Projects

The study area covered by the surveys, is the German North Sea, including coastal waters as well as the Exclusive Economic Zone (EEZ). For divers, the Special Protected Area (SPA; DE 1011-401) "Eastern German Bight" covering 3,100 km², is of high importance. Furthermore, a main concentration area of divers has been defined covering 7,000 km² (BMU 2009), which largely overlaps with the SPA. An overview of the study area as well as protected areas is given in Figure 4-1.

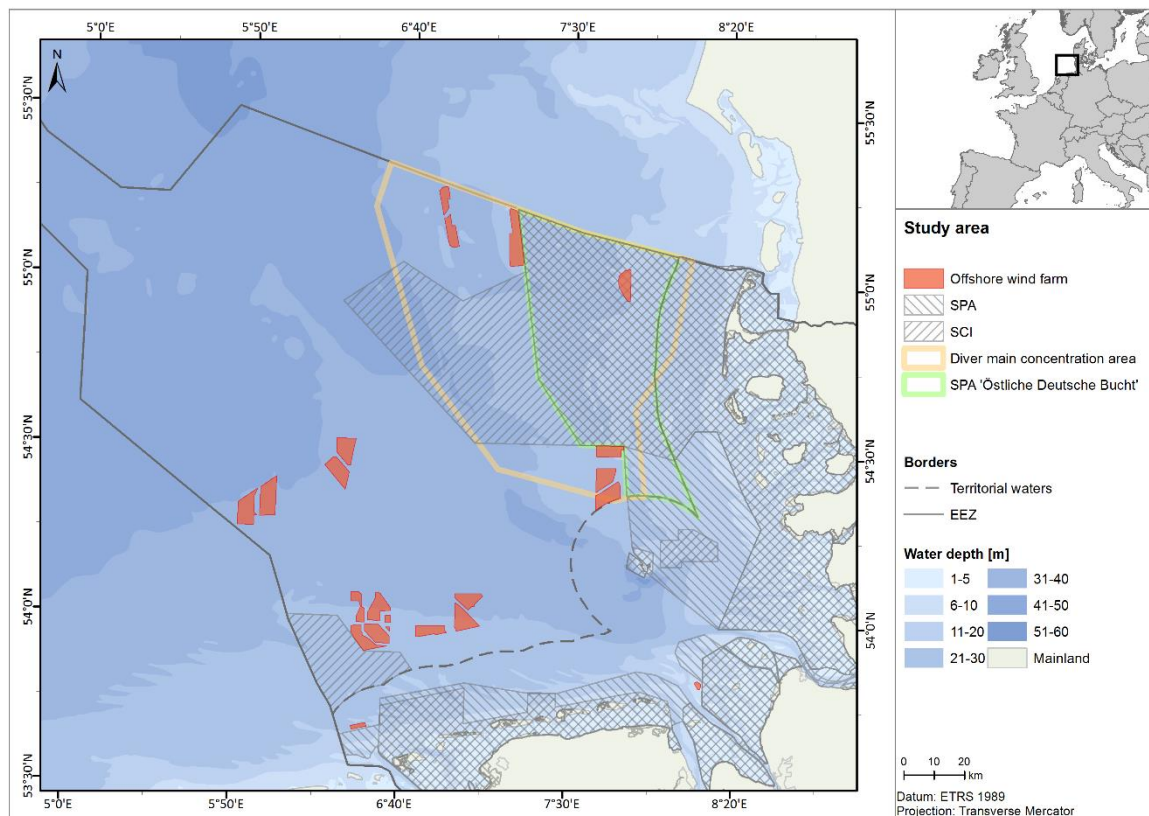


Figure 4-1 Overview of the study area, with EEZ and protected areas, including diver main concentration area and SPA „Eastern German Bight“.

Table 4-1 Data coverage in all years (spring only).

Year	Area covered in spring
2001	16,6%
2002	19,5%
2003	35,2%
2004	52,4%
2005	16,9%
2008	90,3%
2009	50,7%
2010	99,4%
2011	88,3%
2012	92,3%
2013	72,2%
2014	93,7%
2015	92,8%
2016	97,7%
2017	84,0%
2018	87,4%

Data coverage for the study area varied between time periods (Table 4-1, Figure 4-2): for the years 2001 to 2005, coverage was low to medium, for years 2006 and 2007, no data was available in spring (for winter, data was available for 2007), and for years 2008 to 2018, coverage was medium to very high (up to nearly 100 % in 2010).

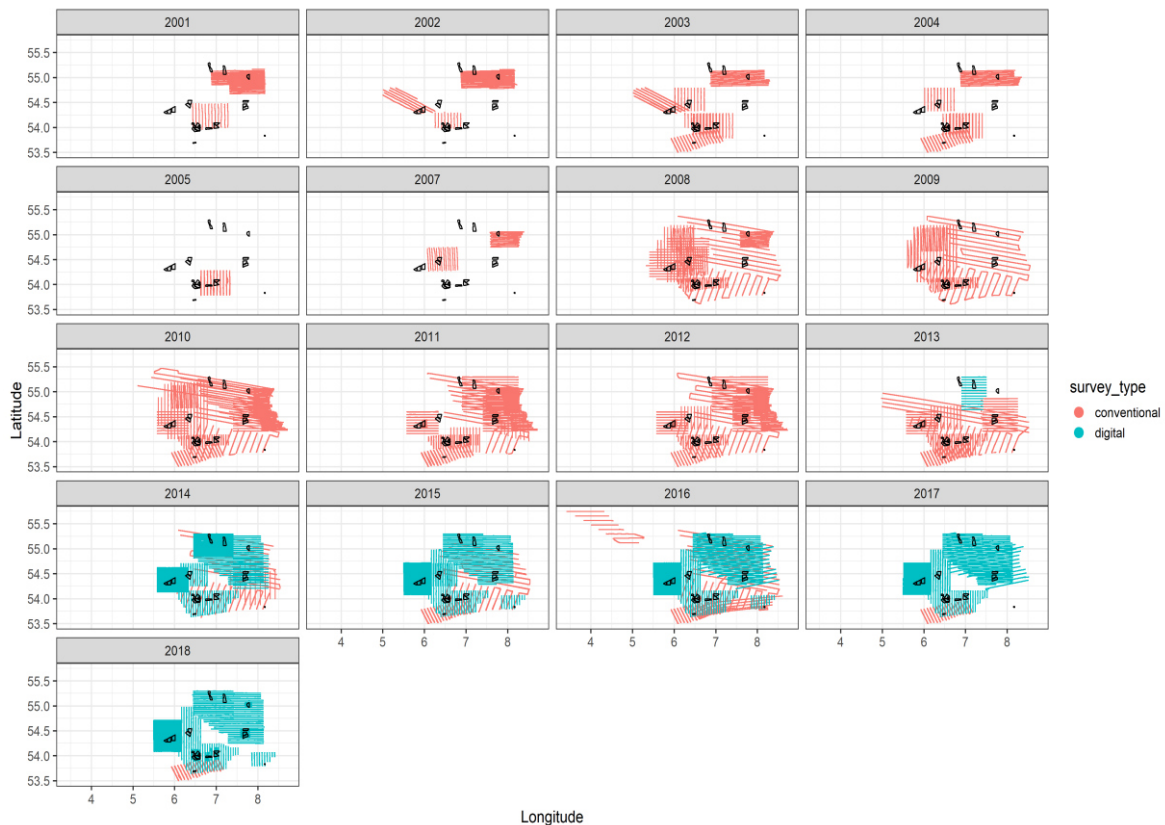


Figure 4-2 Survey effort for analyses between 2001 and 2018 (spring and winter combined).

To examine potential regional differences in the disturbance effect of wind farms on divers as well as regional differences in abundance, the dataset was divided into two separate areas, a northern and a southern area (Figure 4-3). The northern area includes the main concentration area for divers in spring (BMU 2009) with the highest densities, while the southern area mainly includes areas of low or medium densities. The two areas did not overlap but both also included data from coastal areas (not just EEZ). Also, data reaching beyond the EEZ borders into Denmark and the Netherlands were included in the models. For all figures and the calculation of stock size the area was cut at the EEZ border. The spatio-temporal model contained all data from both areas together, and only predictions were made separately for the two areas and for the main distribution area. The total prediction area was 28,625 km², prediction area for the northern area was 12,782 km² including the main concentration area defined by BMU (2009) of 7.000 km² and prediction area for the southern area was 13,375 km².

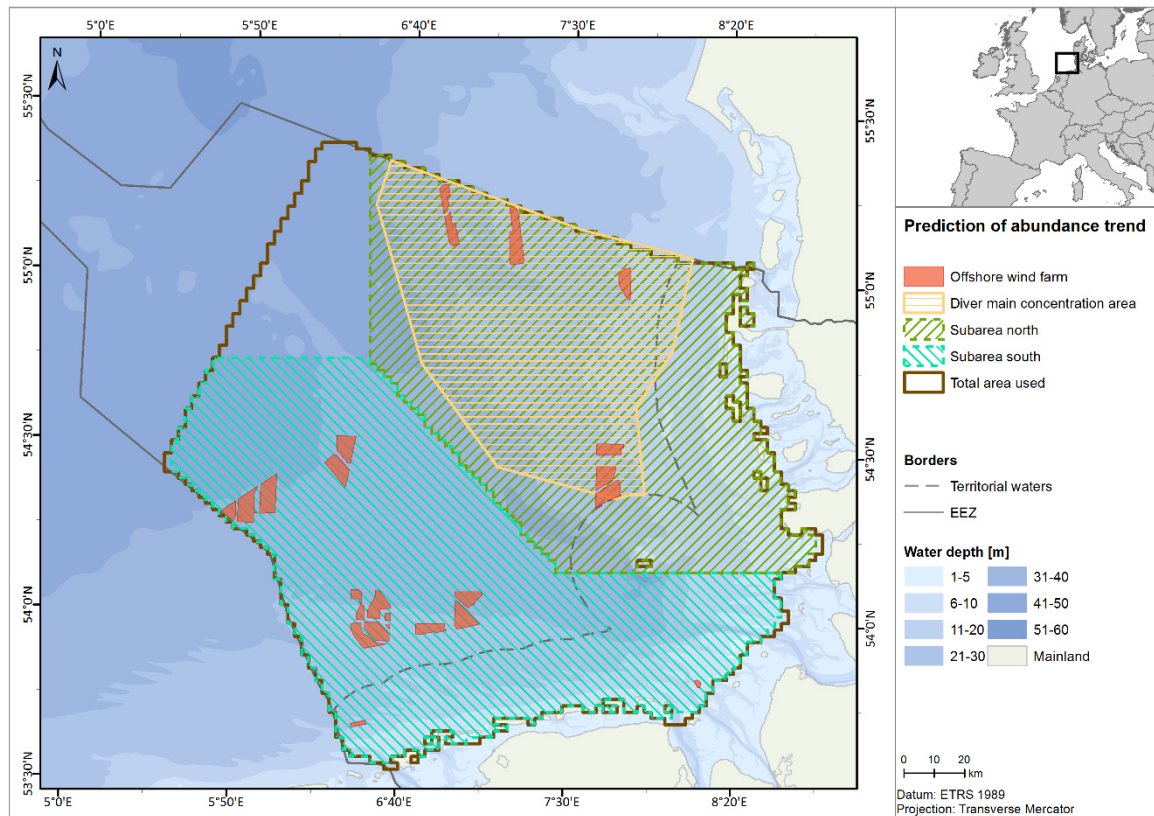


Figure 4-3 Total prediction area with northern and southern sub-areas and diver main concentration area (BMU 2009), as well as operational wind farms as of 2018. Blank area in the northwestern corner was not included for predictions of north and south, but was part of the total area.

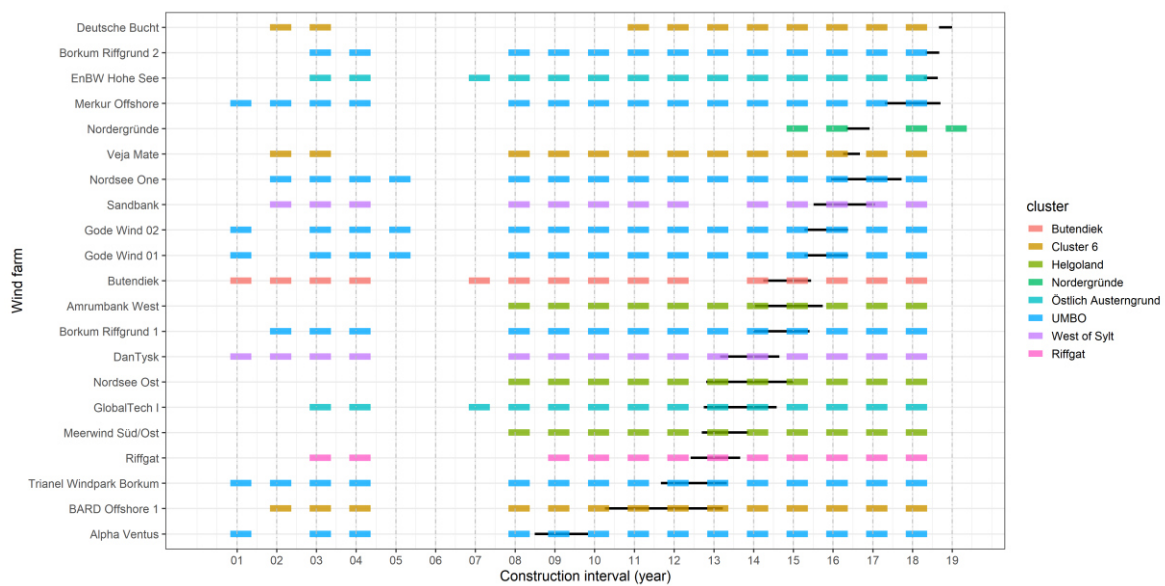


Figure 4-4 Years with survey data for wind farm projects within the German North Sea and indication of construction periods (black line). Spring and winter combined (see Appendix A.1.2 for separate figures).

Data from baseline monitoring, construction or operation of 20 different wind farm projects were included in the analyses (Figure 4-4). For each wind farm, periods before construction, during construction and operation were defined. For each wind farm, the complete spring or winter period (all surveys) was assigned to one phase (before or after construction). In some cases, for OWF starting construction in spring, surveys taking place before the exact start of construction were still assigned to the construction phase. During spring, this applied to 16 % (digital) and 3 % (visual) of surveys in the data set.

4.2.1 Aerial Monitoring: visual and digital aerial surveys

Aerial surveys are an established method for monitoring distribution and abundance of seabirds relatively quickly and over large areas (e.g. Diederichs et al. 2002, Camphuysen et al. 2004, Garthe & Schwemmer 2005, Zydels et al. 2019). Over the years, the methodology advanced and so there are two main ways of aerial surveys: visual observer flight surveys and digital aerial surveys. All surveys were restricted to favourable weather conditions with sea state less than 4 (visual surveys) and 5 (digital surveys) and clouds at least above flight altitude.

4.2.1.1 Visual Aerial Surveys

Observer-based visual survey flights were conducted primarily as part of baseline monitoring from 2001 to 2013, based on the standards set by the German Federal Maritime and Hydrographic Agency (BSH 2003, 2007). In this method, qualified observers would detect and record birds in real time from a small aircraft along transect lines (ca. 3-5 km apart). Surveys were carried out from twin-engine, top-winged airplanes equipped with bubble windows to allow observations directly underneath the plane at a flight height of ca. 250 ft (76 m) and a speed of around 180 km/h (Noer et al. 2000, Diederichs et al. 2002, Camphuysen et al. 2004). Two main observers and an additional control observer would record their observations on a digital tape recorder. The following information was recorded for each sighting: UTC time, number of animals, age (adult or juvenile), behaviour (flying, swimming, diving, etc.) and flight direction. By using an on-board GPS, it was possible to geo-reference every single observation. Observations were divided into distance bands by their distance to the transect line using an inclinometer (see also 4.2.1.1.1). The original method used three bands A-C with a width of 119 m, 268 m, and 656 m, respectively, resulting in a total transect width of 1,043 m on each side of the transect line (Figure 4-5). Over the course of various projects, distance bands were adjusted by either further dividing the inner band A or including the 44 m wide innermost band D underneath the aeroplane. Prior to each survey, weather conditions (sea state on the Beaufort scale, turbidity, maximal visibility range in km, cloud cover, cloud reflexion and glare) were recorded. Whenever conditions changed, the observers recorded that information. These parameters were used to estimate the valid observation effort.

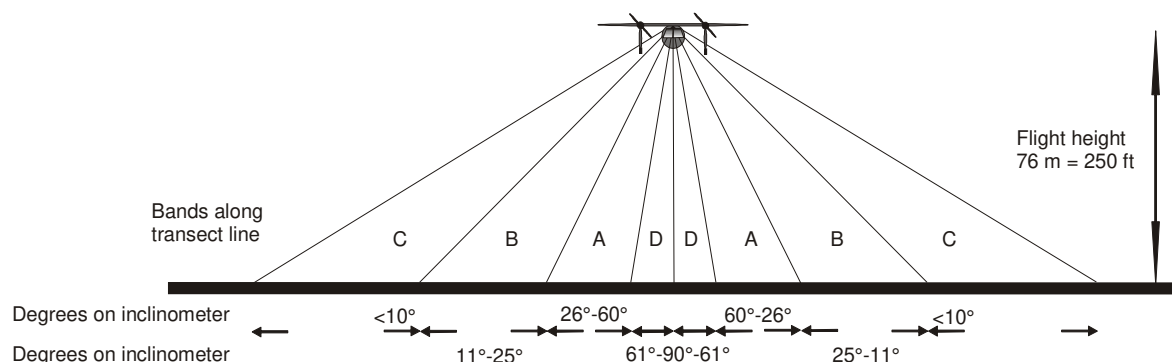


Figure 4-5 Schematic representation of distance bands for visual aerial surveys.

4.2.1.1.1 Distance sampling for visual aerial survey data

The detectability of seabirds decreases with increasing distance from the survey platform. In order not to underestimate the density of birds in farther transect bands with lower detection probability, an effective strip width (ESW) is calculated, which is smaller than the total transect width (Buckland *et al.* 2001). We applied distance sampling to all visual aerial data using the package *mrds* 2.1.14 (LAAKE *et al.* 2015) in R 3.2.3 (R CORE TEAM 2015). We tested models with different predictors and selected the model with the lowest value for Akaike's Information Criterion (AIC) to estimate the total number of divers for each transect segment (see Appendix).

4.2.1.2 Digital Aerial Surveys

The digital aerial surveys were conducted according to the standards set by the German Federal Maritime and Hydrographic Agency (BSH 2013) and subdivided into three different techniques: "APEM", "DAISI" and "HiDef". Generally, the survey method stayed consistent per survey area, however in some areas, i.e. DanTysk/Sandbank and Cluster Östlich Austergrund, more than one method was used. This kind of monitoring was based on digital image recordings (pictures or film) collected in the survey area, which were examined later. Unlike in the visual surveys, species identification was done based on recorded images, not in the field. The recorded footage was evaluated by professionals qualified in species identification, with a separate step for random sample quality control. Flight height in digital surveys was greater than in visual survey flights, so survey aircraft could fly over the wind turbines and disturbance to birds was minimised. In all digital surveys, a twin-engined airplane was used. Precise geographical positions of each observation were recorded using GPS technology. While survey flights were generally only conducted during favourable weather conditions, parameters such as seastate, glare, cloud cover, air and water turbidity were recorded and pictures of insufficient quality were excluded from analysis.

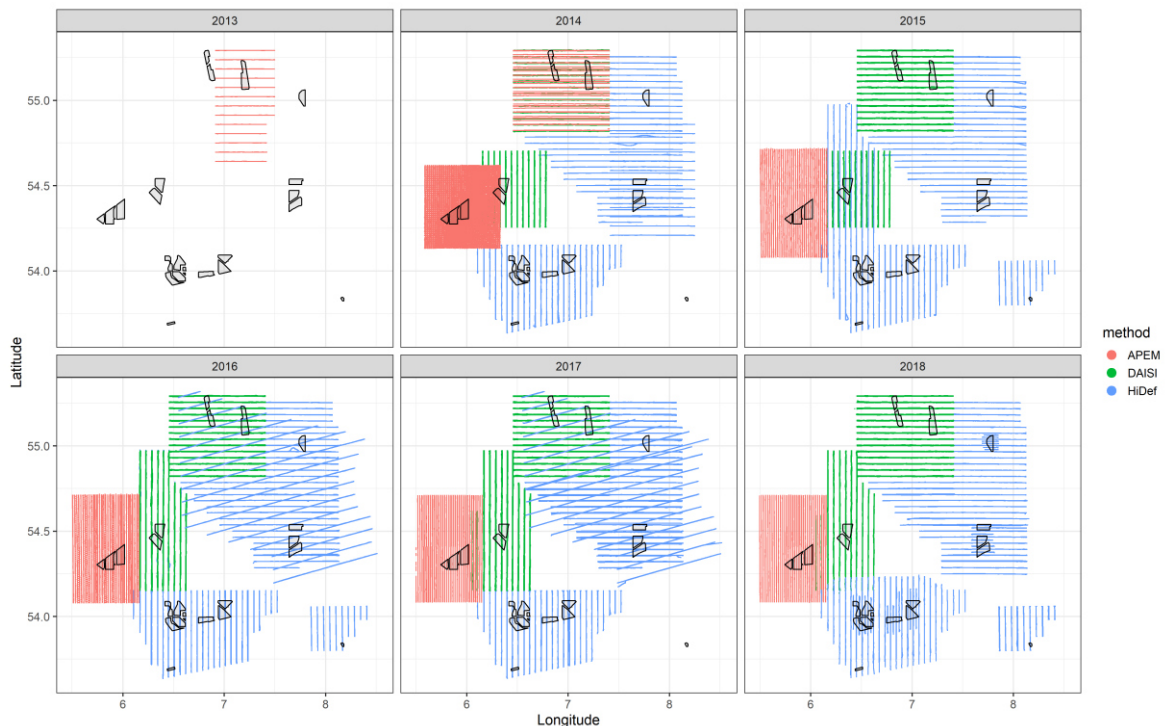


Figure 4-6 Coverage of digital aerial surveys for the three different techniques.

APEM

The APEM² technique (APEM Ltd.; Busch 2015) is based on still image recordings along transect lines in the survey area. Four cameras took images simultaneously and constantly. The four frames were then merged into one image with a resolution of ca. 3 cm (2 cm since January 2017) on the sea surface. Flight height was approximately 400 m (1,300 ft) at a speed of 120-130 knots. This method included narrow transect lines (ca. 1.6 km spacing), which were close enough to allow the forming of a grid. This is one of the main differences to the other two survey methods described below.

Cluster 6 was surveyed exclusively by the company APEM Ltd. Species identification and quality control was done by IBL Umweltplanung GmbH up to January 2016. Thereafter, APEM Ltd continued with the image analysis. APEM also surveyed the DanTysk/Sandbank area in March and April 2014 with image analysis done by APEM Ltd. Here, the transect lines of the survey area were used rather than a grid.

DAISI

The surveying technique DAISI³ ("Digital Aerial Imagery System") was developed by and belongs to IfAÖ GmbH. Like APEM, it uses a photo technique to record objects along transect lines. DAISI consists of two medium-format cameras with a resolution of 2 cm on the sea surface. Photos were taken at a minimal interval of 1.5 s, which leads to an overlap of ca. 48 % between frames. At a

² <https://www.apemltd.co.uk/>

³ <http://www.ifaoe.de/en/services/ornithology/daisi.html> (accessed 23.07.2019)

flight height of ca. 426 m (1,400 ft) and a flight speed of 100-120 knots, the camera system covered an area of at least 407 m at sea surface level. Transect lines were 3-4 km apart.

The survey areas DanTysk/Sandbank and Cluster Östlich Austergrund were monitored using DAISI. The transect lines in the area DanTysk/Sandbank were oriented East-West, whereas transect lines in Cluster Östlich Austergrund were oriented North-South. Dan-Tysk/Sandbank was surveyed by APEM for two months in spring 2014, and Cluster Östlich Austergrund was surveyed by HiDef in selected months. Species identification and quality control was done by IfAÖ.

HiDef

The HiDef⁴ technique uses a high-resolution video camera system consisting of four independent cameras with a resolution of 2 cm on the sea surface. The position of the cameras can be adjusted to avoid glare on the sea surface. On each side, the cameras covered an area of 143 m and 129 m with a distance of ca. 20 m in between. Thus, a total coverage of 544 m along a 604 m strip at sea surface level was achieved. Flight height was approximately 549 m (1,800 ft) and flight speed around 220 km/h (120 knots) on transect lines that were ca. 3-4 km apart. Depending on the survey area, species identification and quality control was done by BioConsult SH, IfAÖ, or IBL Umweltplanung.

The survey areas Butendiek, Cluster Helgoland, Nordergründe, Cluster Nördlich Borkum were exclusively covered using HiDef video systems. Cluster Östlich Austergrund was partly surveyed by DAISI and HiDef. The survey areas included either North-South or East-West transect lines that were around 3-4 km apart. Digital survey data made available to the project via the Federal Agency for Nature Conservation (BfN) is also based on the HiDef method.

4.3 Model description

We applied an INLA-SPDE approach for spatio-temporal geostatistical data by integrating the observed intensities and effort on the mesh nodes. The data was fitted by means of a negative-binomial family distribution, where the intensity of the observed process is the main driver of the posterior probability. As an advantage, for this model the use of environmental predictors is generally not required but if desired it is possible to include them. As bathymetry was an important environmental explanatory variable in other studies (Petersen et al. 2014, Heinänen 2016), a model validation process was performed to assess whether or not there was a need to add this covariate to the model.

4.3.1 Creating mesh structure

A constrained refined Delaunay triangulation spatial mesh (Figure 4-7) was constructed for the entire surveyed area and digital and visual flight data was integrated on the mesh nodes for

⁴ <https://hidef.bioconsult-sh.de/> (accessed 23.07.2019)

computational convenience. Information regarding data collection method, number of sightings, effort, season and year was preserved for modelling purposes.

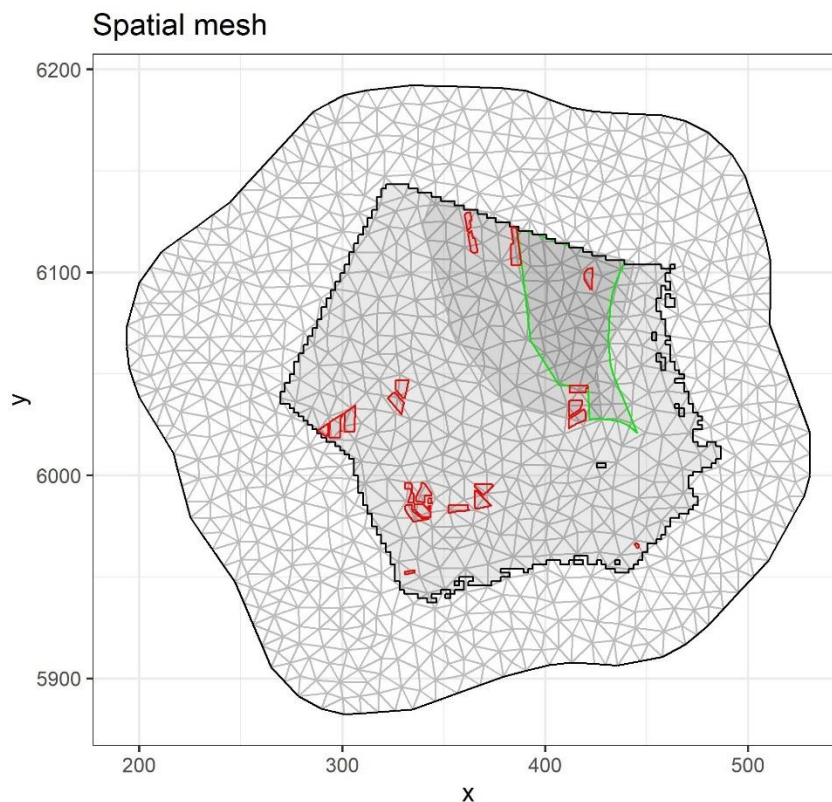


Figure 4-7 Spatial mesh used for the spatio-temporal model. Main diver concentration area (grey line) and SPA "Eastern German Bight" (green line) are depicted.

4.3.2 Data structuring

Seasonal variation was assessed by splitting the dataset in two, one for the wintering season (1st November to 28th February) and the second for the spring season (1st March to 15th May). Although diver stocks and wind farm displacement were assessed for the overall area during both periods, in addition, and as a result of the different distribution pattern found in the north-east and south-west areas of the German North Sea, results were also presented for the northern and southern area separately, as well as for the main diver concentration area (BMU 2009). See Table 4-2.

4.3.3 Model development

To answer the questions formulated, two different models were developed for each season: a) An only-distance model to assess the displacement of divers due to the construction of new wind farms; and b) an explicit spatio-temporal model to assess changes in the spatial distribution and estimate trends in abundance.

The model assumptions were: 1. Perfect detection for HiDef method (see also section 4.3.3.3) 2. Independence of spatial and temporal processes 3. No persistence (individuals move around) 4. Time is discrete (e.g. years).

To incorporate an accurate assessment of the model error and properly account for the property that an observation is more correlated with an observation collected at a neighbouring location than with another observation that is collected farther away, we used a spatially-structured random effects model which incorporates such spatial dependency.

- a) For the only-distance model we constructed a spatio-temporal multivariate GMRF with a one-dimensional Matérn covariance for the spatial domain (distance to the nearest operating or in-construction wind farm at each season) and an autoregressive process of order 1 (AR(1)) to describe the temporal dependence. The data collection techniques (HiDef, DAISI, APEM or Visual) were also included as categorical covariates. Distance to the wind farm was calculated for each year based on the wind farms already built or under construction during the surveyed season. New wind farms were taken into consideration based in their start date of construction.
- b) Regarding the spatio-temporal model, data were not only spatially but also temporally indexed. As such, the interest is not only the species spatial distribution, but also in assessing how the spatial distribution changes over time. To incorporate all such dependencies into one modelling framework, we construct a spatio-temporal multivariate Gaussian Random Field (GRF) with a Matérn covariance for the spatial domain, and an autoregressive process of order 1 (AR(1)) to describe the temporal dependence. We take an integrated nested Laplace approximation (INLA) approach (Rue et al. 2009) for Bayesian inference, coupled with a Stochastic Partial Differential Equation (SPDE) model (Lindgren et al. 2011) to account for the spatial autocorrelation. Data collection methods were included into the model as categorical covariates.

Calculations were performed in the R statistical software using the inlabru package (R Core Team 2019, Bachl et al. 2019).

Table 4-2 Summary of the datasets, models and prediction areas used

Dataset	Model	Prediction area
Spring	Spatio-temporal	Total area, north, south, main concentration area (BMU 2009)
	Only distance	Total area, north, south
Winter	Spatio-temporal	Total area, north, south
	Only distance	Total area, north, south

4.3.3.1 Diver displacement due to wind farms

The displacement distance is defined as the distance from the edge of an offshore wind farm up to which diver density is significantly lower than a reference density defined as the overall mean of the dataset used in the specific model (years 2009 – 2018). The one-dimensional mesh for the only-distance model was constructed at 1 km steps up to 40 km distance to get a detailed picture at short ranges, and then at 10 km steps up to 100 km to catch the general density at long distances. Prediction was performed at 500 m intervals to get a smooth curve of average effect of the distance to wind farm on the predicted diver densities. Values above 0 represent a positive effect and values below 0 showed a negative effect.

The average total population size estimated by the model for the whole range was used as the reference point to estimate the actual diver density in the region and was later used to calculate the overall habitat loss in the area due to the effect of the wind farms.

The displacement distance was calculated as the intersection between the average value from the distance model (zero-line) and the model curve. The intersections between the zero-line and the confidence intervals of the model curve are taken as confidence interval for the displacement distance.

4.3.3.2 Calculation of theoretical habitat loss

The theoretical habitat loss is defined as the area corresponding to the habitat of birds that theoretically no longer is available for use (see Figure 5-25) and is given as radius around an OWF.

It is a term that has become established in the context of the Environmental Impact Assessment (EIA) for offshore wind farms over the past years. However, in connection with the displacement of fish-eating sea birds from a wind farm area and its surroundings, it must be noted that this does not mean the complete loss of habitat including the resources available to the species, such as in the case of the felling of a tree for a tree-dwelling species. As food resources (e.g. fish) in these offshore areas are not fixed in their location and are constantly moving between the wind farm area and the area where the divers are, one cannot really speak of a "loss of habitat" but rather of a deterioration of habitat. However, in order to keep the terminology consistent with common praxis, we stick to it within this report.

The expected number N_{Ref} and expected density D_{Ref} are taken from the distance model outcome. The theoretical habitat loss can then be calculated with the following formula:

$$A = \frac{N_{Ref} - N_{Loss}}{D_{Ref}}$$

N = number (or proportion)

D = number (or proportion) per unit area

A = calculated theoretical habitat loss (in km²)

By far, the largest number of divers found in the study area occur in the spring, when the calculation of theoretical habitat loss was done for the total study area, as well as for the northern and southern sub-area.

The displacement distance was defined as given in chapter 4.3.3.1 and displacement strength was determined by the shape of the distance model curve up to the defined displacement distance. For each 0.5 km band around the wind farm footprint, the area was calculated (in km²) to account for an increase in area with increasing distance from the OWF. Using then the predicted densities for these 0.5 km bands around the footprint, the number of individuals that should be there in an undisturbed situation could be calculated.

From the number of displaced divers it can be back-calculated how much area these birds would occupy with an assumed reference density. This area is defined as theoretical habitat loss (in km²). In order to determine the radius of this area, which is the theoretical habitat loss around a wind farm, the respective 0.5 km bands around the wind farm are summed up until the area corresponds to (or exceeds) the previously calculated habitat loss area. The maximum value of that range is given in the results. The calculation was applied to three different polygons in order to investigate the influence of different designs of the wind farm areas on the size of the theoretical habitat loss: First, a square with 8 x 8 km edge length, second a rectangle with 3 x 20 km edge length and third a circle with a radius of 5 km.

For the calculation of the number of individuals displaced, the average density from the spatio-temporal model for the respective (sub-) area was used as reference.

4.3.3.3 Estimation of diver population size per year and inter-annual population changes.

In order to capture the general population trends and for computational convenience, the spatial mesh for the spatio-temporal model was constructed using a maximum distance between nodes of 5 km. Enough space was incorporated around the surveyed area to avoid undesired boundary effects (Lindgren et al. 2011). Prediction points were masked according to the desired prediction area (total, north or south).

In order to avoid the inclusion of an excessive number of factors, all visual surveys were assumed to have a similar detection rate. By using distance-sampling we have already accounted for a large part of the variation in detection rates between groups of observers. As there were surveys from all three different digital and the visual survey technique performed simultaneously in the same area, it was possible for the model to correct for differences in the detection rate between all these four techniques. To take into account differences between the different recording techniques within the model approach, we assumed a 100 % detection rate in the sampled area for HiDef surveys (Mendel et al. 2018). The error associated with the sampling technique was included in the results together with the natural temporal and spatial random variability among surveyed seasons. Surveys collected by the same sampling technique during the same season were averaged and the model was only fed by one value per season per technique per mesh cell without taking standard deviation for these values into account. Thus, the intra-seasonal variability was lost during the integration process and the resulting variance has not been further considered as a necessary implication to obtain a general inter-annual pattern.

Total population and associated error for each single year and season were summed up from the posterior predicted densities at each location of the spatial domain for the corresponding year.

4.3.4 Model validation

4.3.4.1 Model validation using cross-validation

To assess the model's predictive performance, the spring dataset was randomly split into two subsets: a training dataset including 80 % of the total observations, and a validation dataset containing the remaining 20 % of the data. The model was performed using the training dataset and its predictive accuracy for each year was assessed using the validation dataset. We repeated this calibration-validation procedure 20 times and model performance was assessed using the correlation index between the observed and predicted values at the testing dataset locations.

4.3.4.2 Model validation (bathymetry)

Using the spatio-temporal model, it was explored whether adding environmental covariates improved the model. As one of the most important environmental covariates (Petersen et al. 2014, Heinänen 2016), bathymetry was added as linear effect or alternatively as non-linear effect. Therefore, three different models were tested (Table 4-3). Models were compared using different estimators of model quality such as the Conditional Predictive Ordinate (CPO; Pettit 1990), a Bayesian diagnostic which detects surprising observations; the Deviance Information Criterion (DIC; Spiegelhalter et al. 2002) which can be seen as a Bayesian version of AIC (Gelman et al. 2014); and the WAIC (Watanabe 2010), a fully Bayesian approach for estimating the out-of-sample expectation scores. In all cases, a decrease in the scores means a model improvement.

Table 4-3 Simplified spatio-temporal model formulas of three different models with regard to bathymetry

Models tested	Model formula
Model 1	bird density ~ structured spatial random effect + temporal random effect + sampling method + intercept
Model 2	bird density ~ structured spatial random effect + temporal random effect + sampling method + bathymetry + intercept
Model 3	bird density ~ structured spatial random effect + temporal random effect + sampling method + bathymetry structured spatial random effect + intercept

4.3.4.3 Before-after approach to estimate changes in distribution and displacement

We also explored an alternative method of using a before-after approach on the annual densities from the spatio-temporal model. For this approach, only the northern sub-area could be used as only here could be found a purely operational phase of at least two years (2017/2018) without any

other wind farm under construction close by. In the southern part there was no year in which at least one wind farm was not in the effect range of another wind farm during construction phase.

Two different reference periods were defined:

- a) data from 2008 to 2012 (before the construction of the first OWF in the northern sub-area, but after construction of several wind farms in the southern sub-area. It was assumed that wind farms already built in the south did not have any impact on the northern area.
- b) data before 2008 (2001-2005) before any OWFs were constructed in the whole study area.

Bird densities from these two time periods were predicted and areas of net gain and net loss for each time period were calculated against the after period (year 2017 and 2018).

To estimate the displacement distance, distances to the already built or in construction wind farms were added to the average spatial predictions for both reference periods (2001-2005 and 2008-2012) and for the after period (2017-2018). Later, densities were scaled and aggregated by kilometre to make them comparable. Scaling was performed for each time period by subtracting the average density divided by the maximum density for the whole distance range from the predicted density at each distance. A Kolmogorov-Smirnov-test was used to assess significant differences between the before and the after data for each distance class (1 km).

5 RESULTS

5.1 Model validation

5.1.1 Comparison of survey techniques (visual, DAISI, APEM and HiDef)

In order to include differences in detection rates between the different techniques of aerial surveys, the categorical variable “survey technique” was integrated in the models as a covariate. The model outcome shows that detection rates varied relative to “HiDef” technique, which was used as a basis for the comparison (1).

Table 5-1 Differences between survey techniques obtained from the model describing the relative detection rate to “HiDef” technique.

	median	sd	0.025	0.975	FACTOR
DAISI	-0.05	0.08	-0.21	0.10	0.95
APEM	-0.38	0.21	-0.80	0.03	0.68
VISUAL (after distance sampling)	-0.23	0.05	-0.32	-0.14	0.80

The comparison given in Figure 5-1 reveals that for DAISI there was no significant difference to HiDef, showing a factor of 0.95. Visual surveys (after applying distance sampling) had slightly lower detection rates, significantly lower than HiDef and the lowest rates and higher SD were calculated for APEM. For APEM, however, this high variability is presumably driven by the early surveys before 2017 where a lower resolution camera was used.

5.1.2 Results of cross-validation

The overall CV score based on 20 random runs (80% training - 20% testing) was 0.71 (Figure 5-1). However, although most years performed very well, other performed poorly (2001, 2002 and 2009). Excluding these 3 years, mean predictive accuracy reaches 0.76. Although years with lower sampling effort and unusual distribution pattern scored lower, years from 2010 onwards all performed very well.

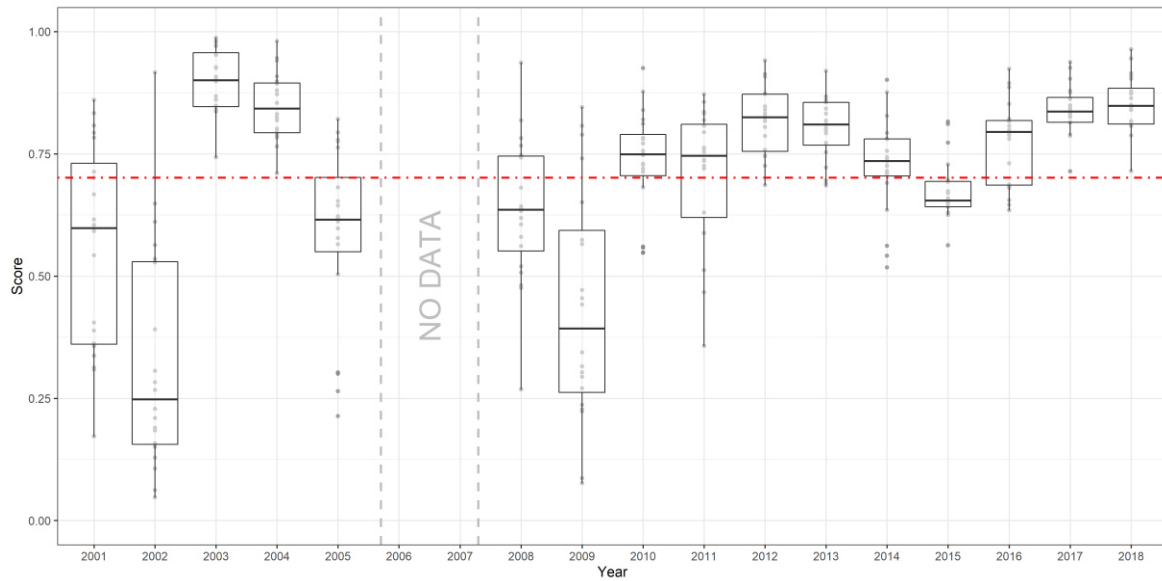


Figure 5-1 Results of the cross-validation for all data during spring. Dashed horizontal red line indicates average score among all years.

5.1.3 Results of model validation for bathymetry

Comparison of the three models using data for spring season shows that adding bathymetry to the spatio-temporal model does not improve the model, which can be shown using four different criteria for Bayesian statistical model evaluation (Table 5-2). Spatial distribution of divers including a linear effect of bathymetry was similar to the model without bathymetry (see Appendix). Since bathymetry is one of the most important environmental factors explaining distribution of seabirds (Petersen et al. 2014, Heinänen 2016) this result shows that no additional predictors are needed for analysis of this dataset.

Table 5-2 Comparison of model with and without bathymetry. Four Bayesian estimators are given to assess the quality of the models (for details see 4.3.4.2).

	CPO	CPO.2	DIC	WAIC
spatial spde model	26468.6542	52937.3084	30293.4711	30401.0672
linear bathymetry model	26502.5548	53005.1096	30298.1512	30407.1988
non-linear bathymetry model	26352.7887	52705.5774	30299.776	30406.2258

5.2 Population size pre and post-construction

5.2.1 Population size during spring

For the years 2001 to 2005 an average of 18,602 individuals were present in the study area per year (Figure 5-1, Table 5-2). By far the highest abundance during the study period was estimated for the year 2003 (29,539 individuals). Then, an exceptional high-density location within the main concentration area resulted in the high overall abundance. However, as data coverage was rather low in those first years, the estimated total numbers are less reliable. For visual representation,

abundances between 2001 – 2005 were therefore averaged. The lowest diver abundance was found for 2008, with only 8,835 individuals. Throughout the study period, strong fluctuations in numbers were found. Maximum numbers slightly larger than 20,000 individuals were estimated for 2011 and 2014, and an estimated 19,221 individuals were present in the study area in spring 2018.

For the northern sub-area, patterns were very similar to the total area, as the majority of divers occurred in this northern part of the study area (Figure 5-3). On average, 13,046 individuals were present in the northern area. After construction of wind farms in the area, the highest abundance was found in 2014 with 18,824 individuals.

For the southern sub-area, during the early years (2001 – 2005), the average abundance was 2,851 individuals. The highest abundance was found for 2012, with an estimated 6,763 individuals (Figure 5-4). Subsequently, numbers declined and the lowest abundance was found in 2017, with 1,681 individuals.

For spring, the season with highest diver densities, diver abundance was also calculated separately for the main diver concentration area, as defined by BMU (2009). Here, patterns were again very similar to the northern and total area (Figure 5-5), showing no decline in abundance but annual fluctuations. After construction of wind farms in the area, the highest abundance was found in 2016 with 13,274 individuals (Table 5-3). On average, 61 % of divers in the German North Sea were found within this main concentration area (n=10,079).

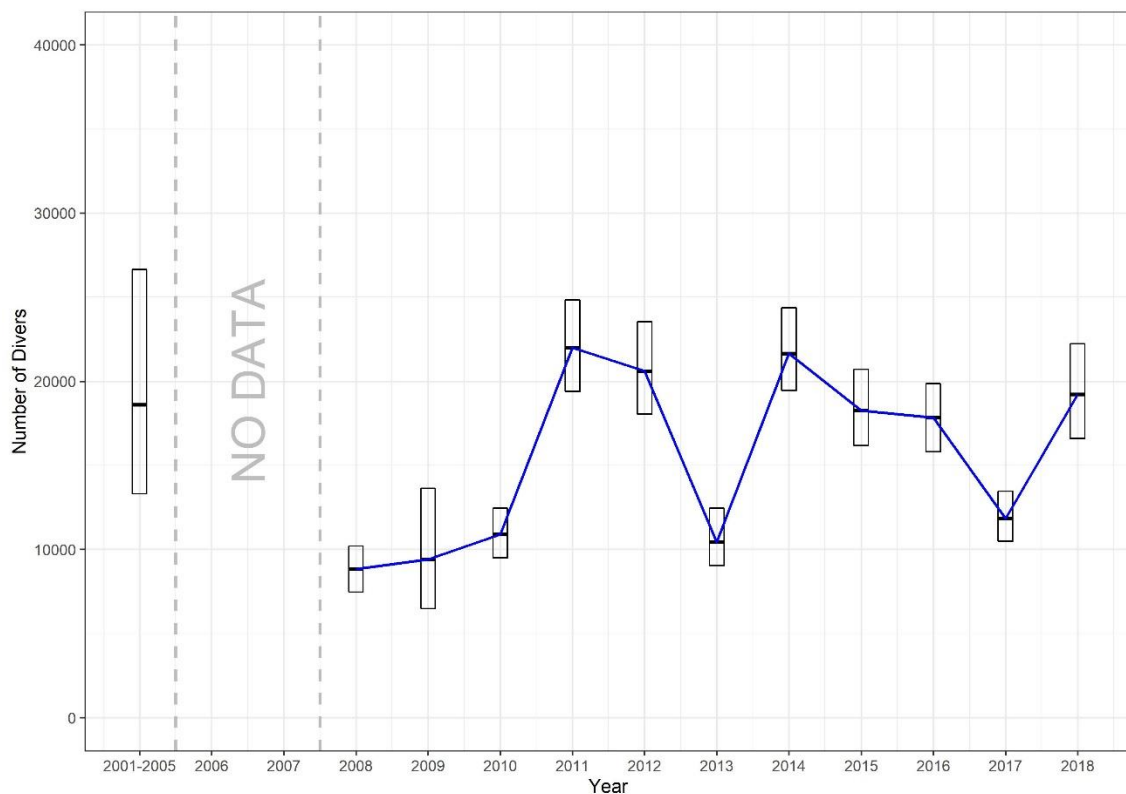


Figure 5-2 Diver abundance during spring for the total study area. Error-bars show 95 % confidence intervals given by the model.

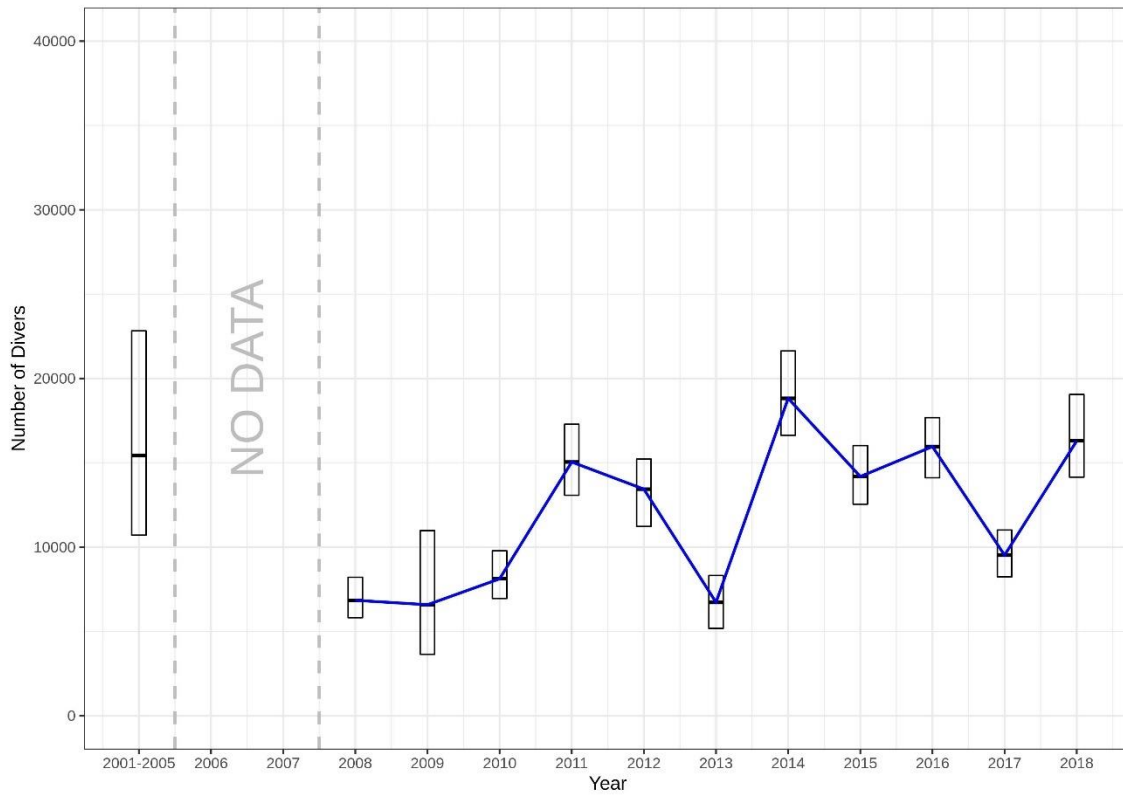


Figure 5-3 Diver abundance during spring for the northern area. Error-bars show 95 % confidence intervals given by the model.

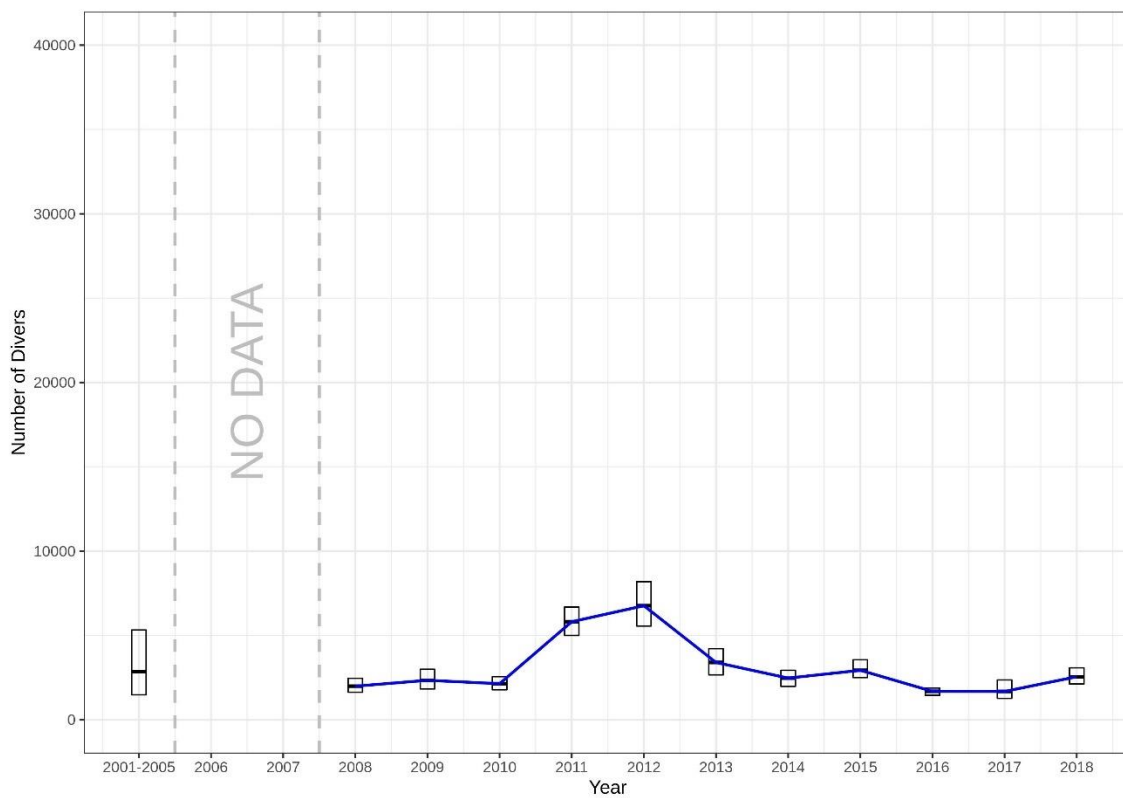


Figure 5-4 Diver abundance during spring for the southern area. Error-bars show 95 % confidence intervals given by the model.

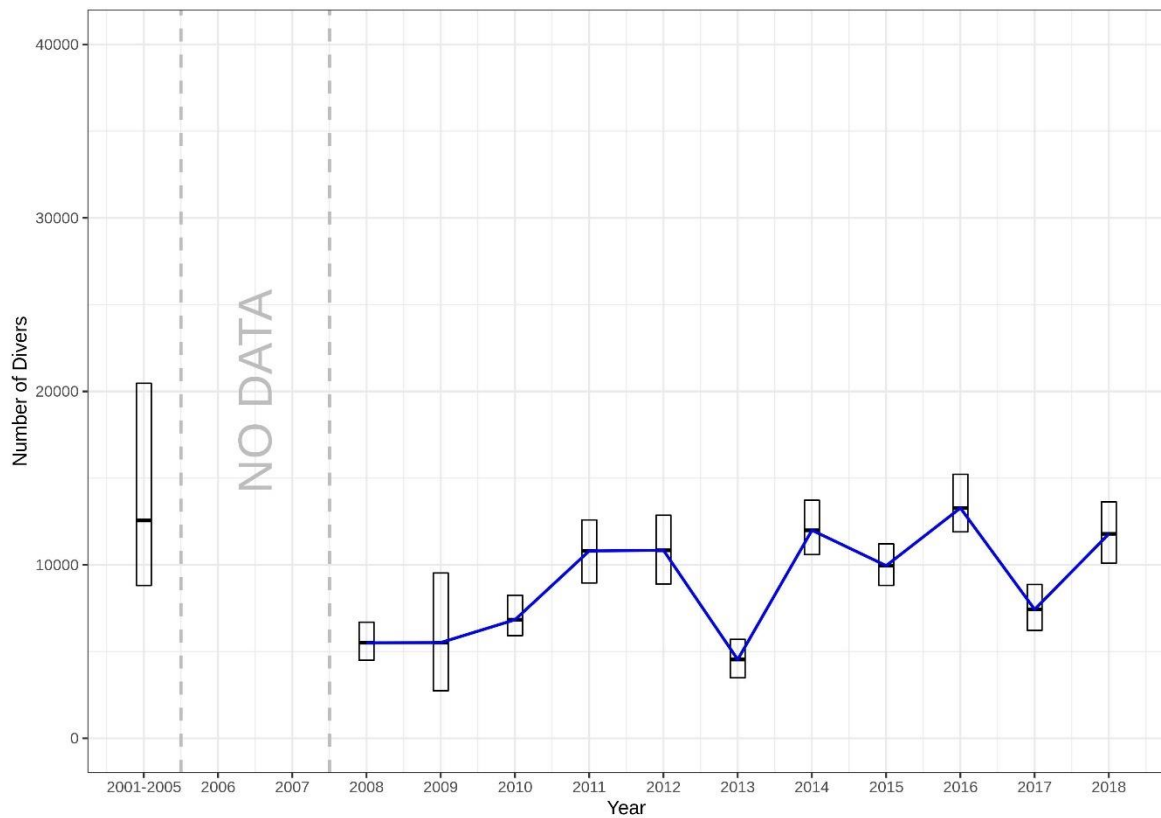


Figure 5-5 Diver abundance during spring for the diver main concentration area (BMU 2009). Error-bars show 95 % confidence intervals given by the model.

Table 5-3 *Diver abundance as predicted from the spatio-temporal model, for spring – total area, north and south. Part of the area covered by the total area is not included in the sub-areas north and south. Therefore, numbers from north and south do not sum up exactly to total numbers and might in some cases slightly exceed total numbers due to the inherent randomness of the modelling process.*

Year	Spring total	Spring north	Spring south	Spring BMU main concentration area
2001	18,822	15,553	3,727	12,280
2002	13,760	10,032	2,590	7,452
2003	29,539	27,681	1,661	24,404
2004	20,276	15,802	3,856	12,554
2005	10,611	8,110	2,422	6,109
2008	8,835	6,846	1,994	5,506
2009	9,415	6,585	2,341	5,518
2010	10,899	8,120	2,139	6,835
2011	21,994	15,047	5,815	10,803
2012	20,609	13,438	6,763	10,839
2013	10,427	6,727	3,399	4,540
2014	21,658	18,824	2,470	11,993
2015	18,266	14,187	2,936	9,950
2016	17,842	15,962	1,686	13,274
2017	11,833	9,523	1,681	7,429
2018	19,221	16,299	2,554	11,784
Average	16,500	13,046	3,002	10,079

5.2.2 Population size during winter

During winter, diver abundance was much lower during all years as compared to spring. The highest abundance was found for 2016 with an estimated 10,144 individuals (Table 5-4). Overall, an increase in diver abundance over the years was found (Figure 5-6). Especially from 2014 onwards, abundance was higher than in previous years.

Similar to spring, the change in diver abundance for the northern area was similar to the total study area (Figure 5-7). Here, the highest abundance was found in 2015 with 6,874 individuals.

For the southern area, population size steadily increased over the years (Figure 5-8). The highest abundance was found in 2018 with an estimated 4,051 individuals.

In all (sub-) areas studied, the highest abundances were found during the last four years (2015 – 2018). No estimates are given for the diver main concentration area for winter, as this area is less important during winter, with lower densities and a more variable spatial distribution.

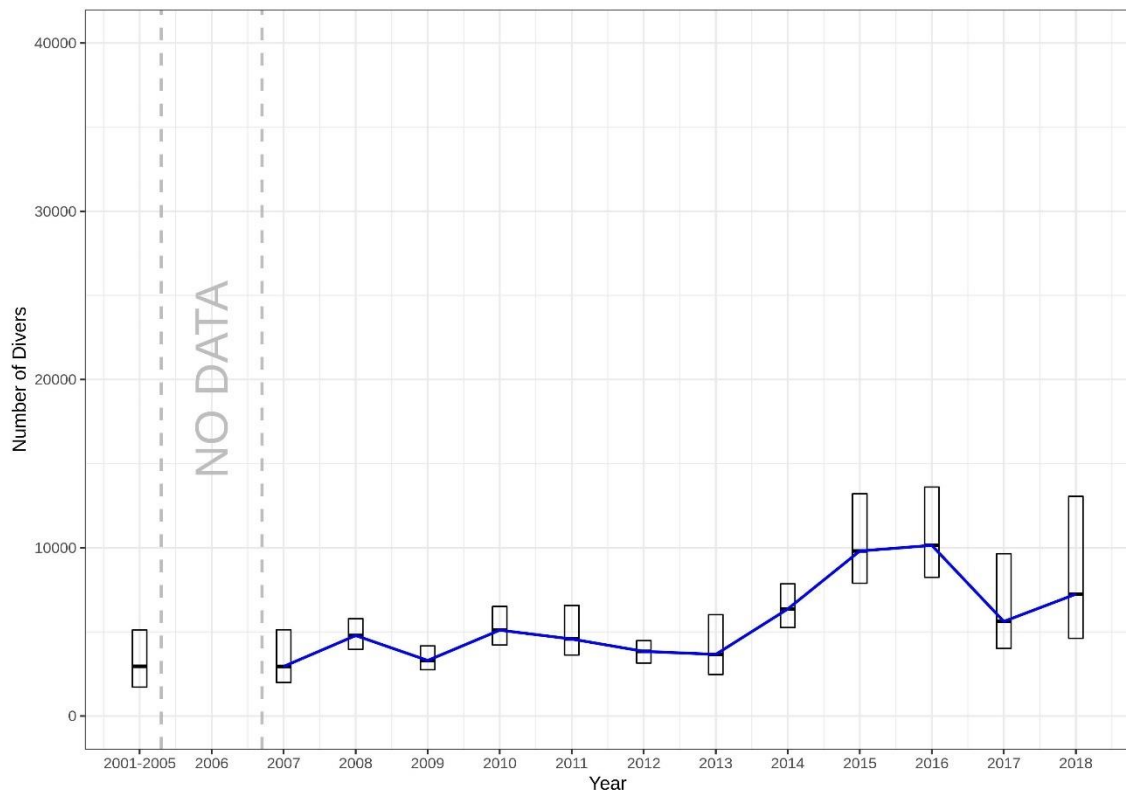


Figure 5-6 Annual population size for winter season and the total study area. Error-bars show 95 % confidence intervals given by the model.

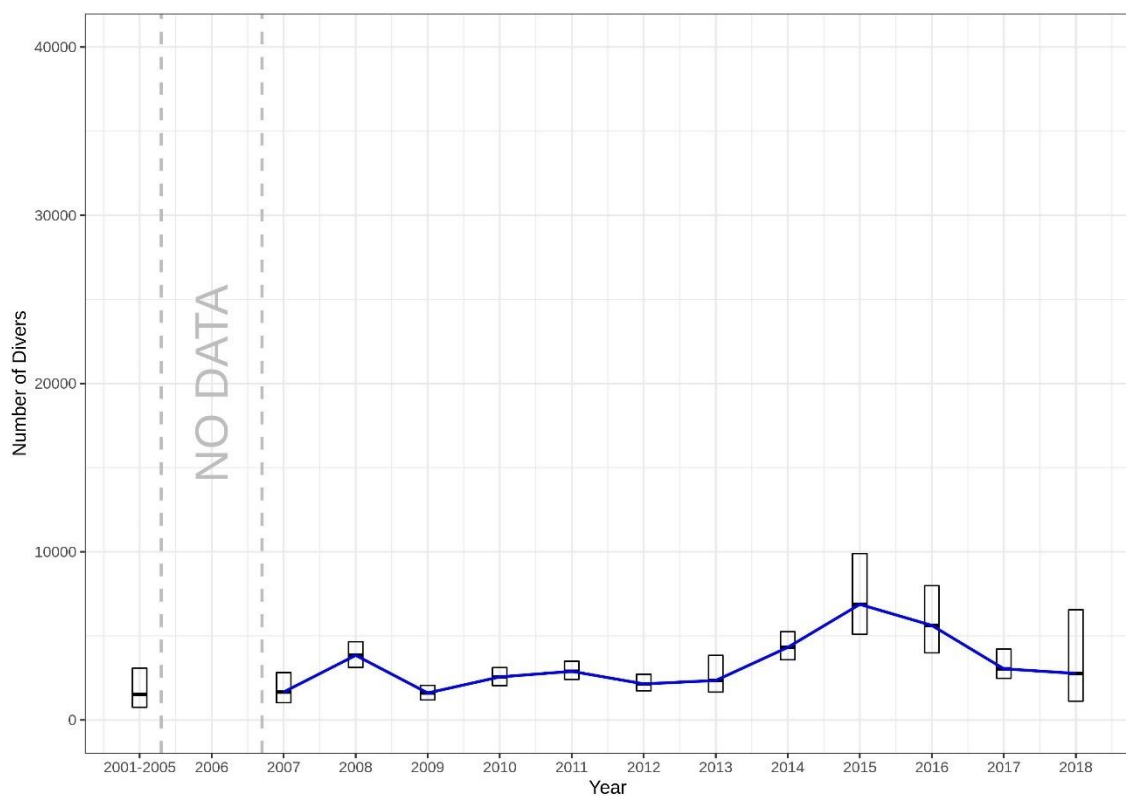


Figure 5-7 Annual population size for winter season and the northern study area. Error-bars show 95 % confidence intervals given by the model.

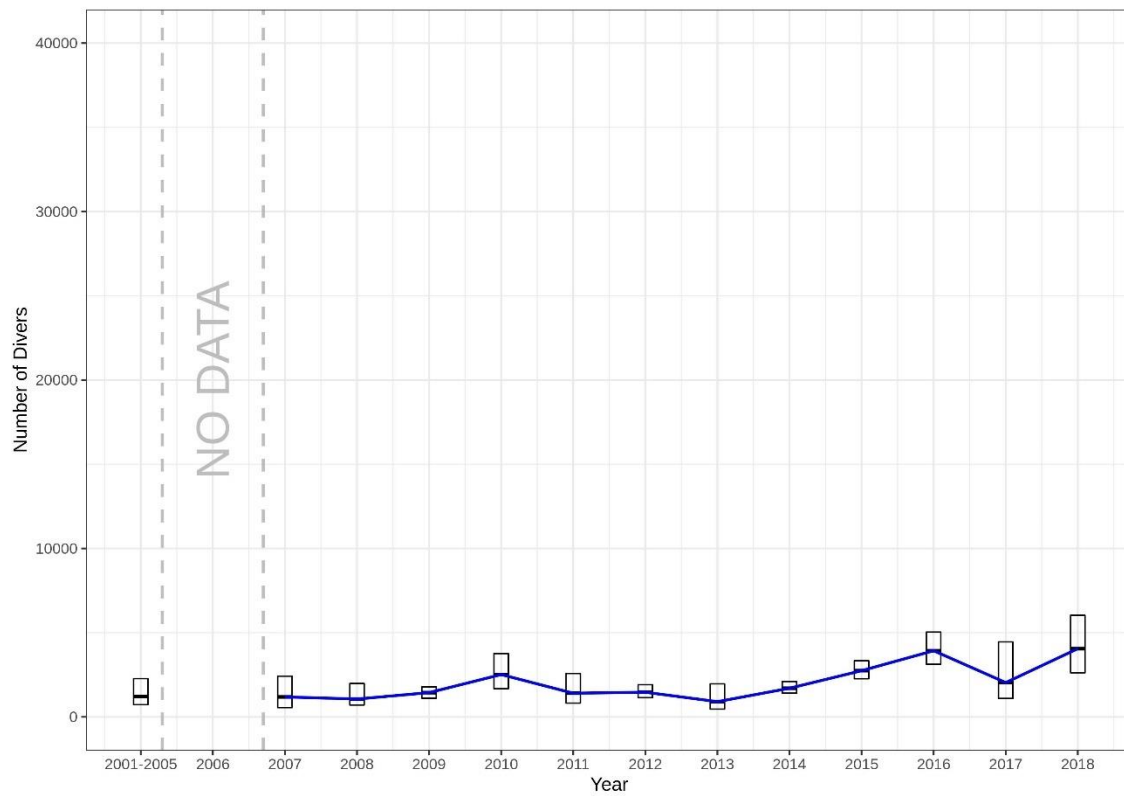


Figure 5-8 Annual population size for winter season and the southern study area. Error-bars show 95 % confidence intervals given by the model.

Table 5-4 Diver abundance as predicted from spatio-temporal model, for winter – total area, north and south. Part of the area covered by the total area is not included in the sub-areas north and south. Therefore, numbers from north and south do not sum up exactly to total numbers and might in some cases slightly exceed total numbers due to the inherent randomness of the modelling process.

Year	Winter total	Winter north	Winter south
2001	2,326	1,399	815
2002	2,131	1,268	653
2003	3,209	2,019	1,197
2004	3,089	1,224	1,377
2005	4,008	1,770	2,011
2007	2,946	1,664	1,184
2008	4,784	3,845	1,061
2009	3,301	1,606	1,442
2010	5,103	2,560	2,513
2011	4,575	2,894	1,407
2012	3,842	2,146	1,464
2013	3,663	2,350	904
2014	6,364	4,319	1,701
2015	9,808	6,874	2,743
2016	10,144	5,622	3,929
2017	5,621	3,043	2,036
2018	7,245	2,769	4,051
Average	4,833	2,787	1,793

5.3 Spatial distribution

5.3.1 Observation effort and densities during spring

A.1.4 shows the mesh used for model calculation of the spatio-temporal model and, for each year, gives the area covered by data with the respective densities per node. In some cases, data points were located outside of German waters, but these were included in the model, to make use of all available information.

As shown in Table 4-1 data coverage was rather low for the first years (2001 – 2005), but nevertheless, the north-eastern part with highest diver densities was covered in all years, except for 2005 and 2009. From 2008 onwards, coverage was high in most of the years, and in several years the study area was covered completely (> 90 %).

The predicted densities from the spatio-temporal model show that the highest densities consistently occur within the main diver concentration area (BMU 2009) in the North. Within this

area, distribution varied somewhat between 2001 and 2012, when no wind farms were built in this area yet (Figure 5-13). From 2013 onwards, divers started to concentrate in an area away from existing wind farms and consistently reached the highest densities in this location.

The average density (all years) for the total area was 0.58 Ind./km²; 1.02 Ind./km² for the northern area and 0.23 Ind./km² for the southern area.

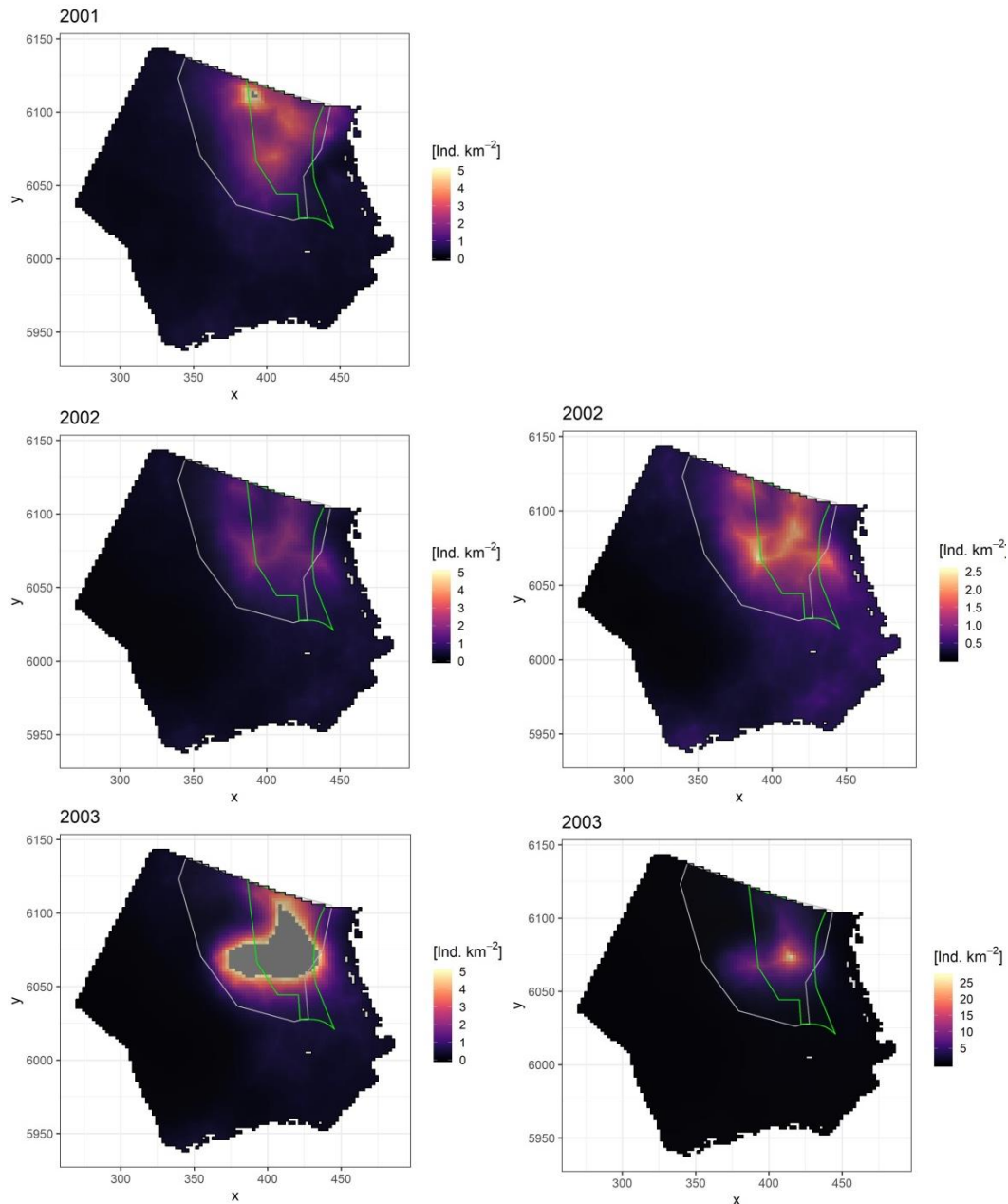


Figure 5-9 Predicted densities for spring for the total study area (years 2001 – 2003). For more details see Figure 5-13.

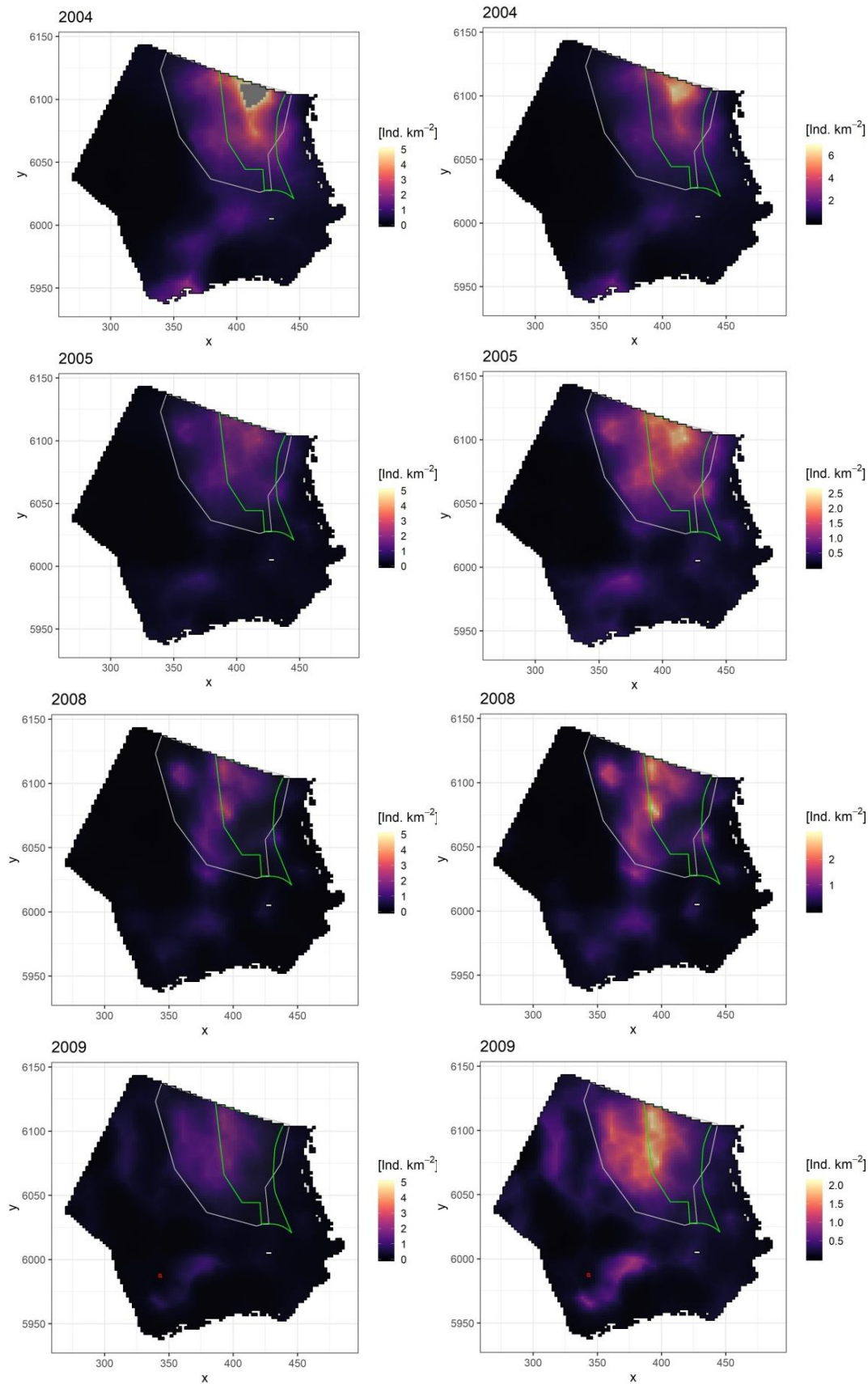


Figure 5-10 Predicted densities for spring for the total study area (years 2004 – 2009; 2006+2007 were not covered). For more details see Figure 5-13.

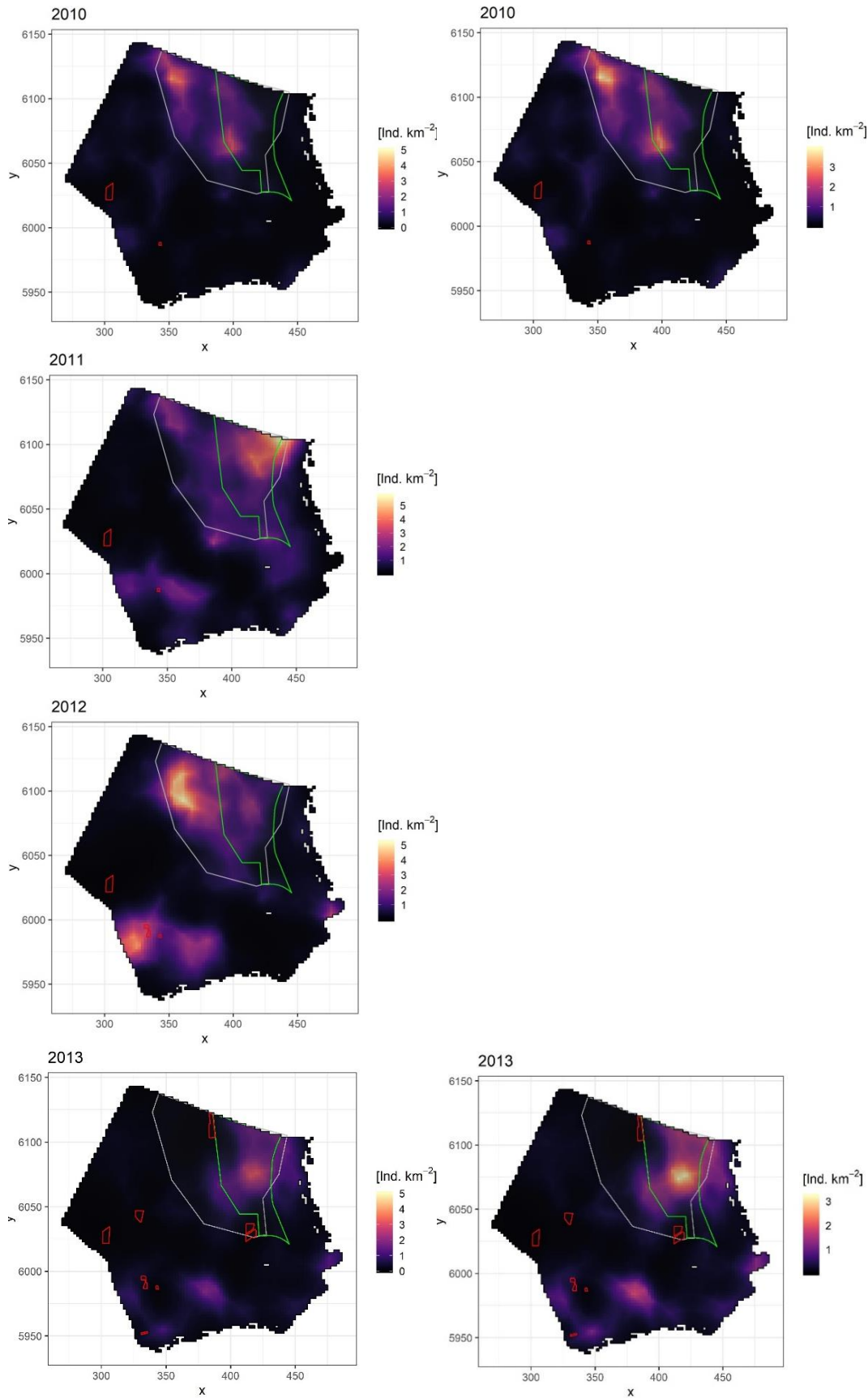


Figure 5-11 Predicted densities for spring for the total study area (years 2010 – 2013); For more details see Figure 5-13.

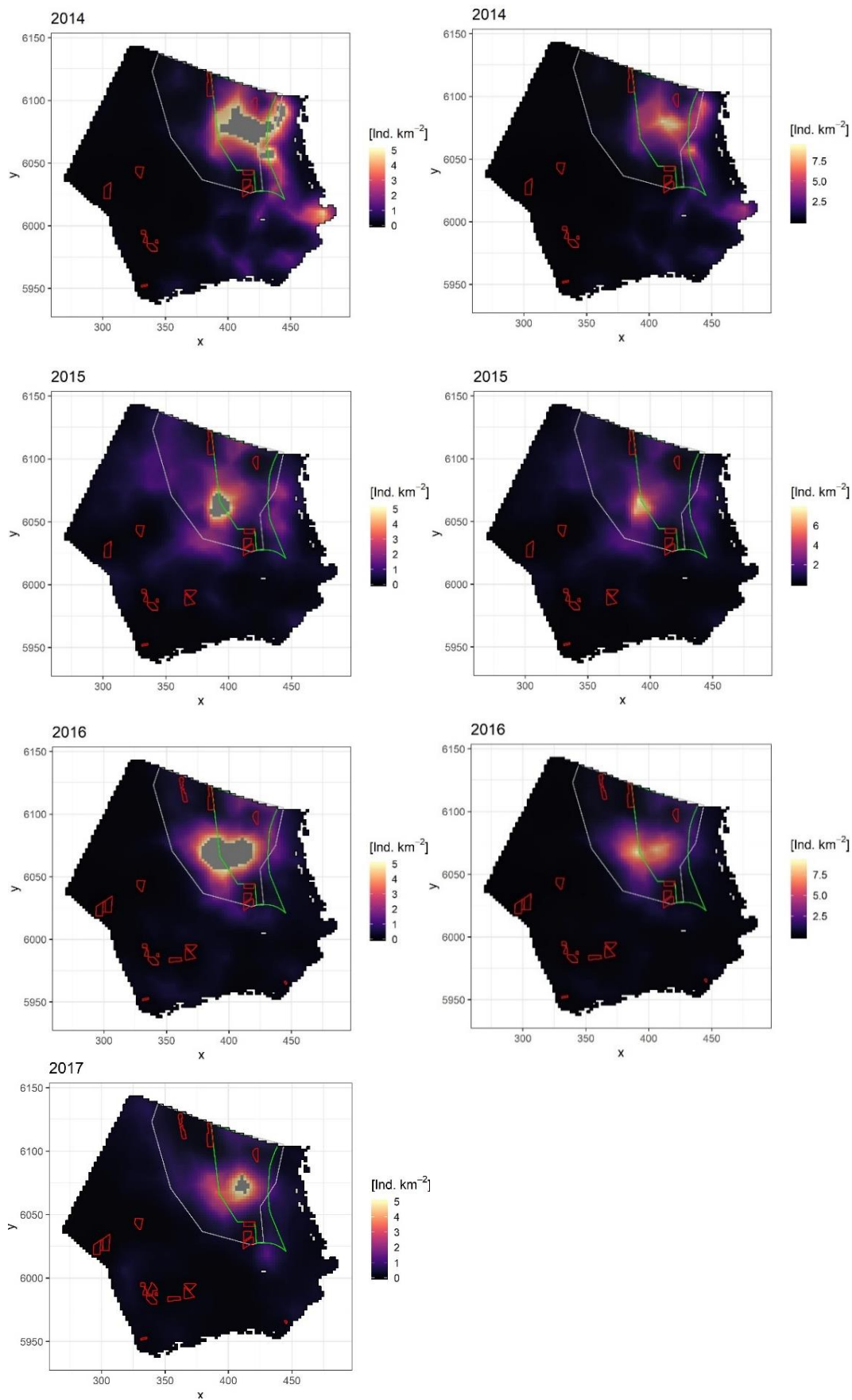


Figure 5-12 Predicted densities for spring for the total study area (years 2014 – 2017). For more details see Figure 5-13.

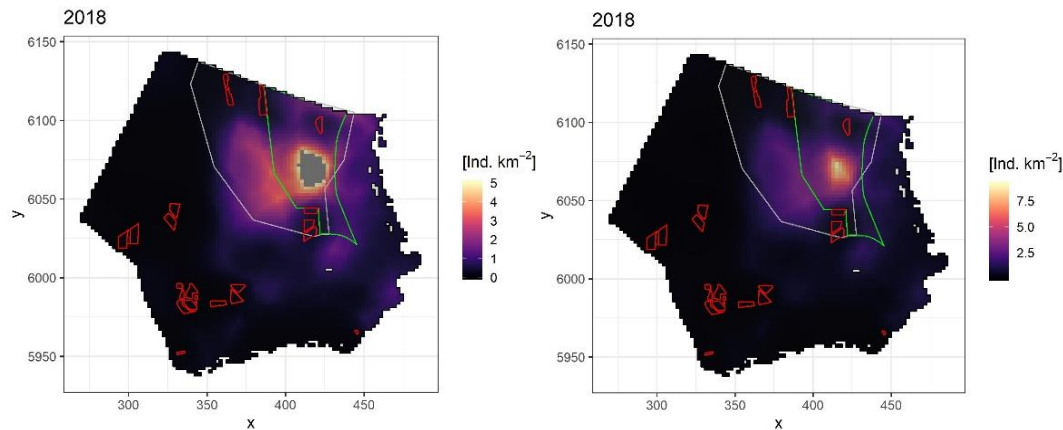


Figure 5-13 Predicted densities for spring for the total study area (2001 – 2018). Densities are given on a constant scale (left) and on a variable scale (right) for each phase/year, if scales differ considerably. Values exceeding the maximum value of the constant scale, are shown in grey. Red borders indicate wind farms under construction or in operation. Green line depicts border of SPA “Eastern German Bight”, white line depicts main concentration area for divers as defined by BMU (2009). Figures for sub-areas (spring, with adjusted scale) can be found in the appendix.

5.3.2 Observation effort and densities during winter

For winter, data coverage varied somewhat from spring season, and was slightly higher during the first years (2001 – 2005), but lower in other years (e.g. 2008). From 2009 onwards, coverage was high (>90 %) in most of the years.

During winter, spatial distribution was different from spring: Areas of high diver densities varied more and were more often located closer to the coast (Figure 5-14 to Figure 5-18). The main diver concentration area was used to a much lower extent than during spring, however, in some years, the highest densities were still reached in that area (e.g. 2002, 2012, 2014). Concentrations north of the East Frisian Islands were found in some of the early years, but these shifted to a more coastal occurrence.

Relatively high densities were also frequently found from the area of the Elbe river estuary towards the north along the North Frisian coast, within the 12-nm zone.

The average density for the total area was 0.17 Ind./km²; 0.22 Ind./km² for the northern area and 0.13 Ind./km² for the southern area.

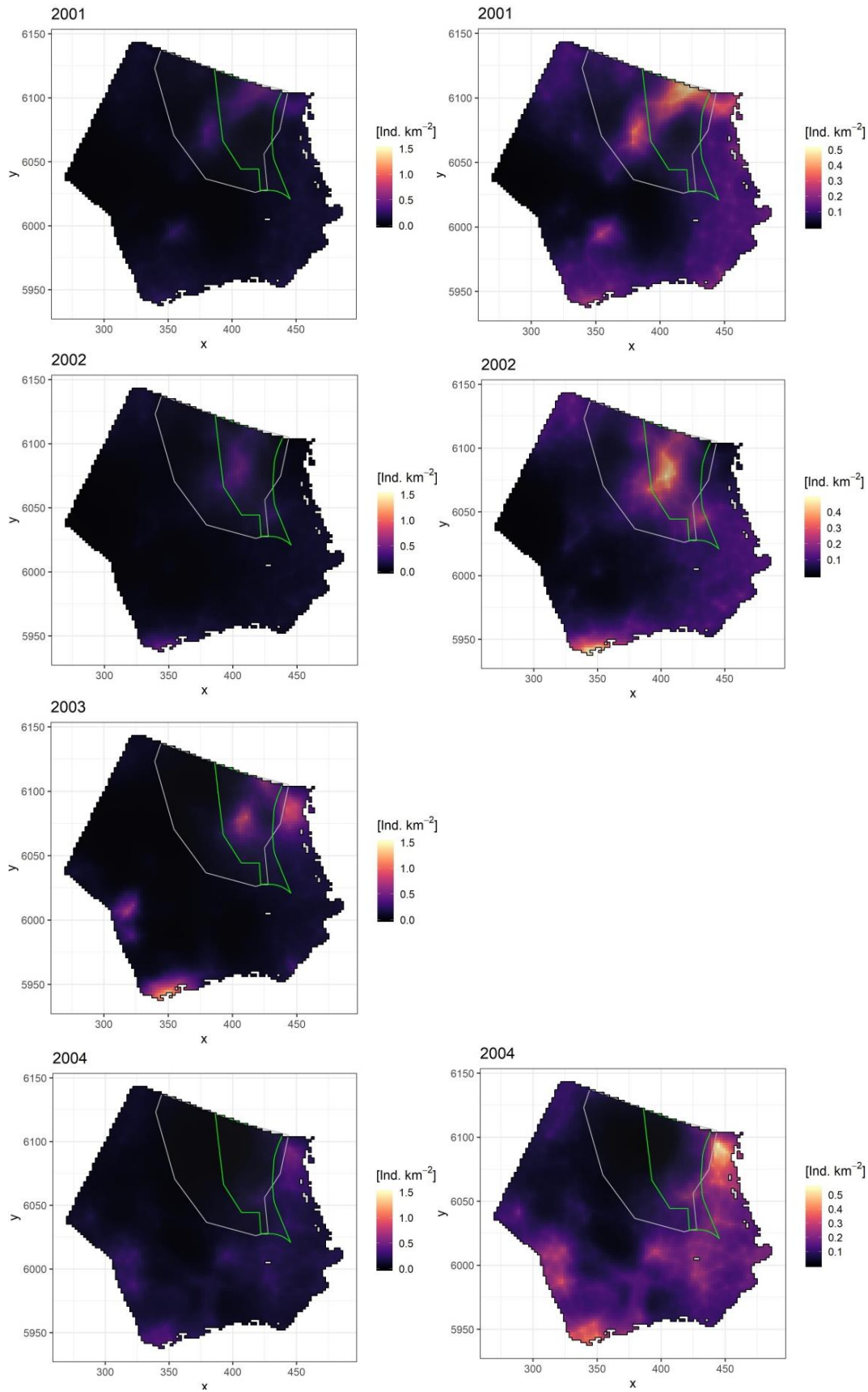


Figure 5-14 Predicted densities for winter for the total study area (years 2001 – 2004). For more details see Figure 5-18.

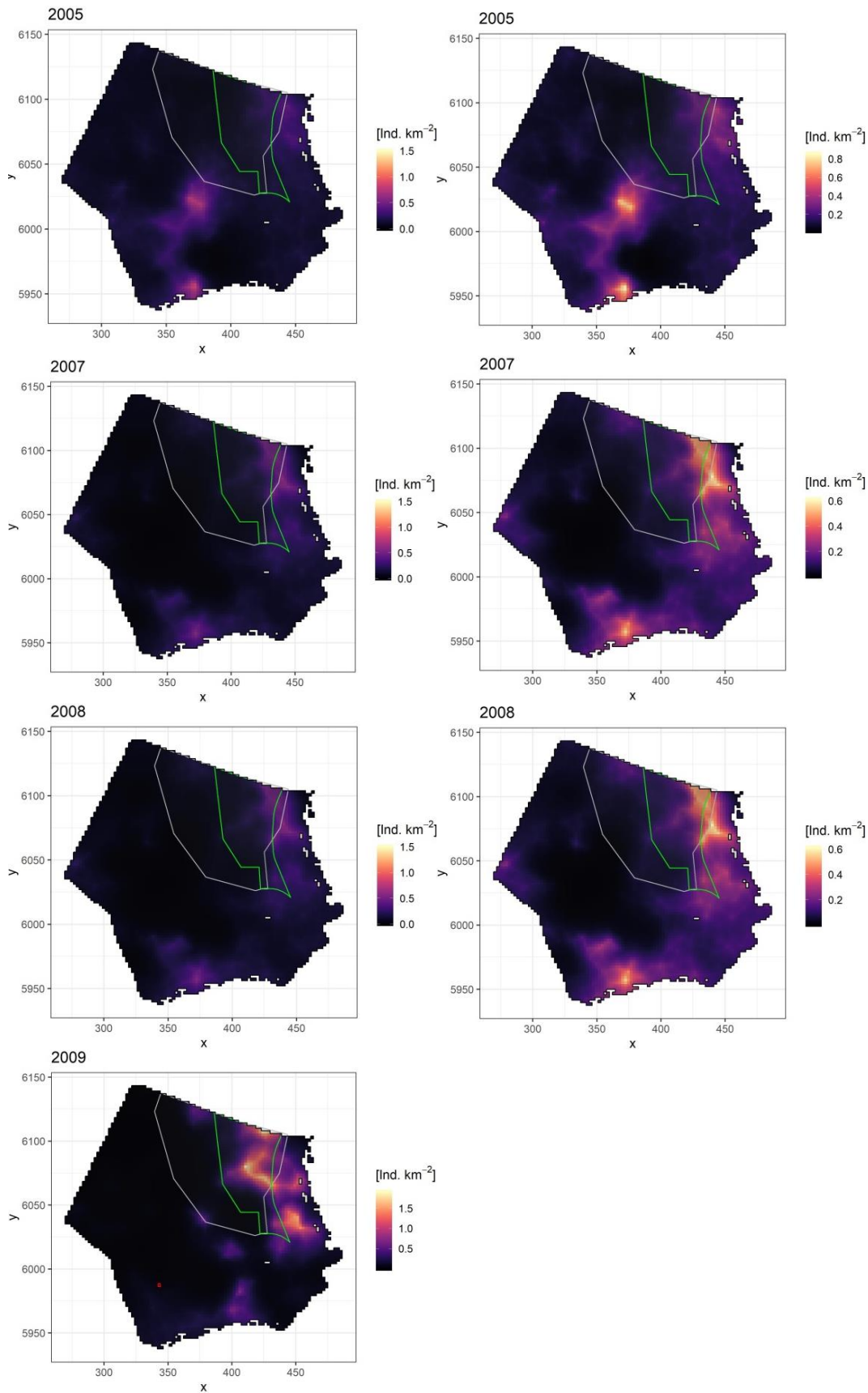


Figure 5-15 Predicted densities for winter for the total study area (years 2005 – 2009; 2006 not covered). For more details see Figure 5-18.

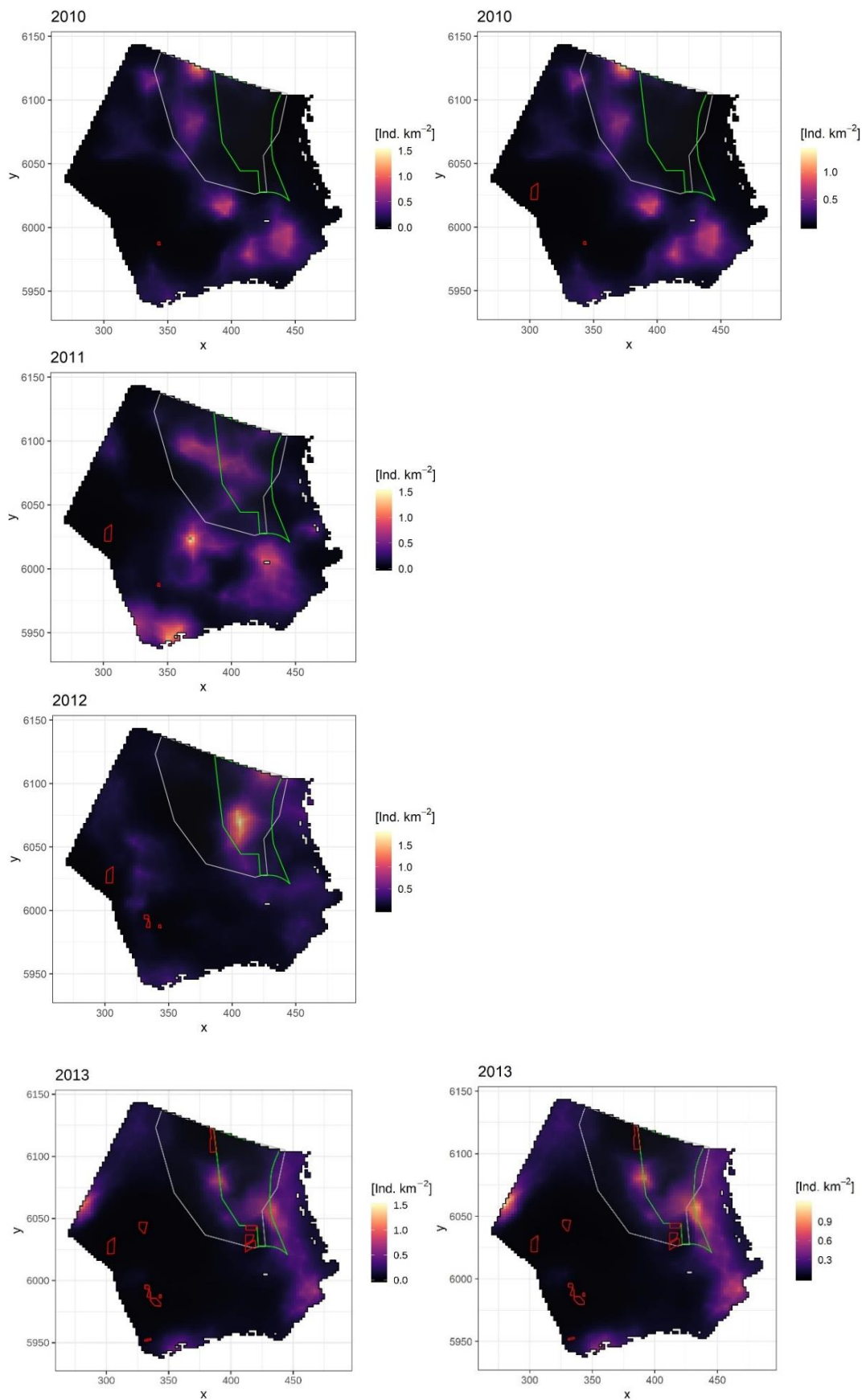


Figure 5-16 Predicted densities for winter for the total study area (years 2010 – 2013). For more details see Figure 5-18.

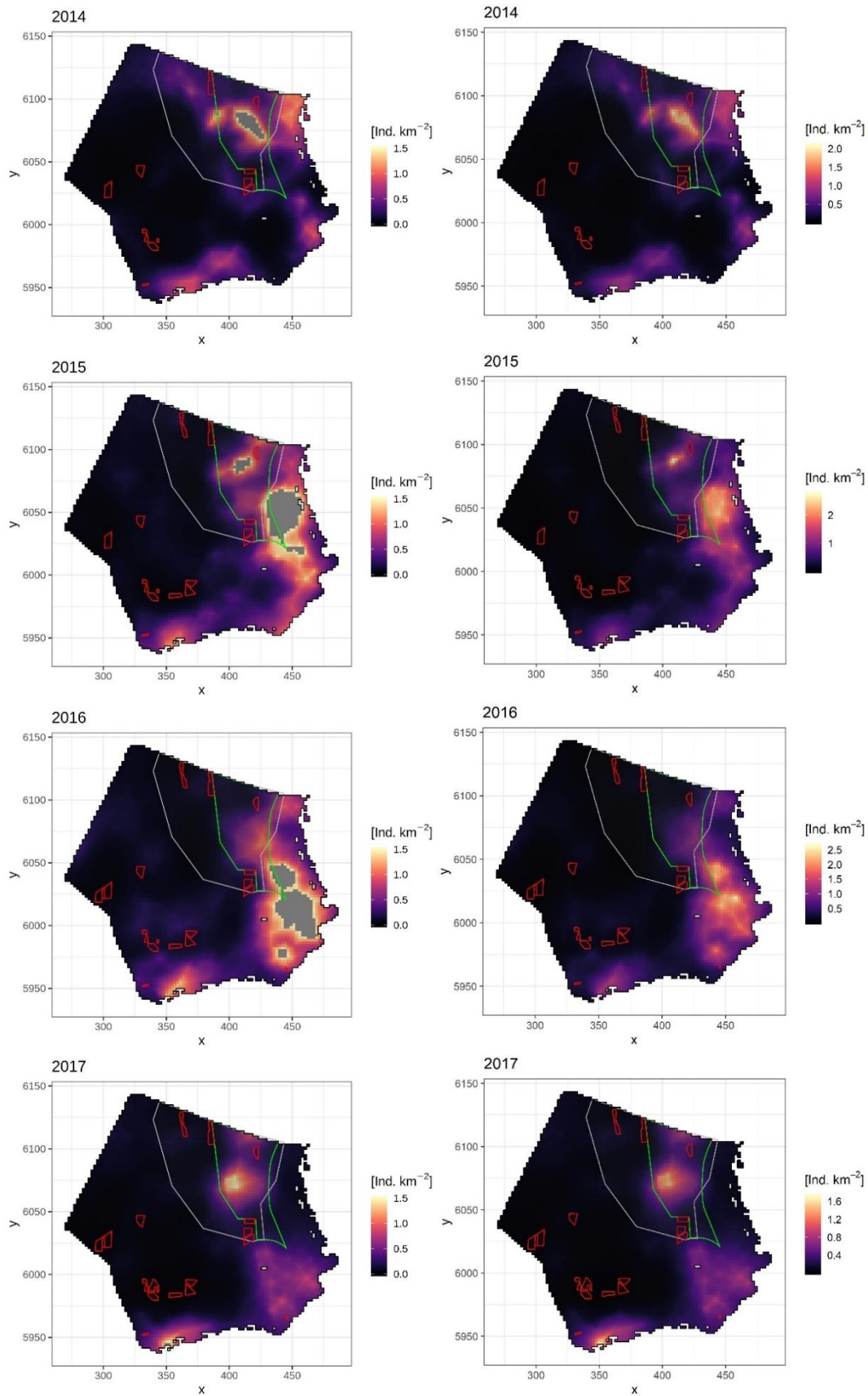


Figure 5-17 Predicted densities for winter for the total study area (years 2014 – 2017). For more details see Figure 5-18.

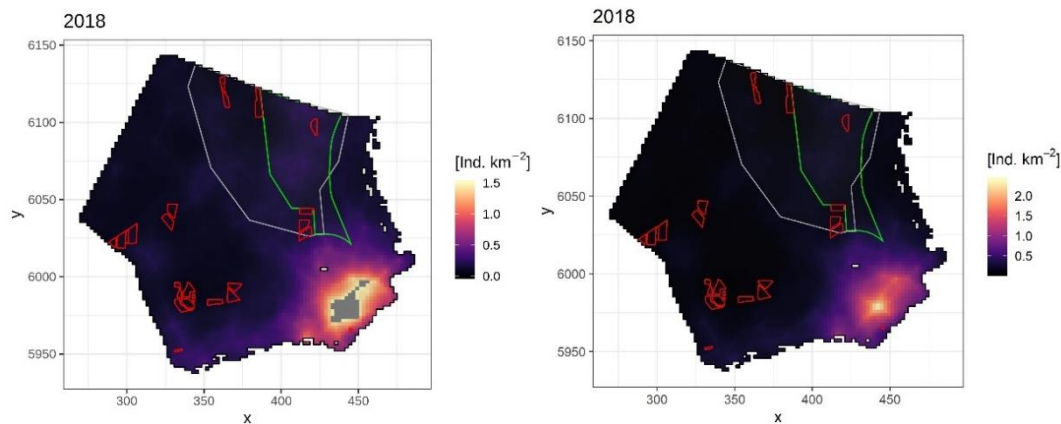


Figure 5-18 Predicted densities for winter for the total study area (year 2018). Densities are given on a constant scale (left) and on a variable scale (right) for each phase/year, if scales differ considerably. Values exceeding the maximum value of the constant scale, are shown in grey. Red borders indicate wind farms under construction or in operation. Green line depicts border of SPA “Eastern German Bight”, white line depicts main concentration area for divers as defined by BMU (2009). Figures for sub-areas (spring, with adjusted scale) can be found in the appendix.

5.4 Displacement effects

5.4.1 Seasonal and regional factors affecting displacement

We investigated the displacement effect separately for the spring and winter period. Furthermore, the effect was calculated for the total study area, as well as separately per sub-area (north and south).

During spring, the displacement effects ranged from 8.5 km in the southern sub-area to 10.2 km (total area). For the northern sub-area, the effect was intermediate (9.1 km).

For spring – total area and spring – northern sub-area, the model curve increased steeply with increasing distance from the wind farms and reached a peak at about 20 km and then decreased again (Figure 5-19, Figure 5-20).

For spring – southern sub-area, the model curve showed a rather flat increase for the first kilometres from the OWF, and then formed a plateau indicating relatively constant densities (Figure 5-21). Therefore, an estimation of a displacement effect is more difficult, as small changes to the model or dataset might shift the intersection between the zero-line and the model curve to a large extent. This can also be seen in the large confidence interval (5.2 – 17.4 km; Figure 5-21).

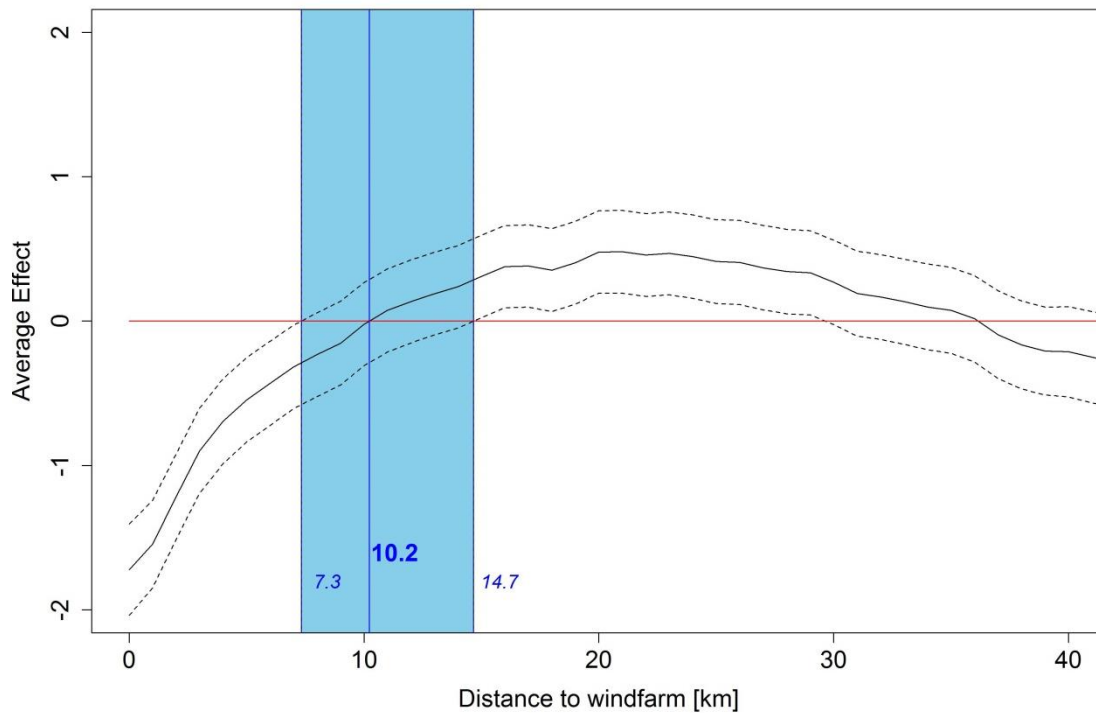


Figure 5-19 Displacement effect for spring - total area. Dotted line depicts the upper and lower 95% CI from the model curve (solid line); the blue shaded box indicates the range of the possible effect. Vertical blue line shows the intersection between the zero-line and the model curve.

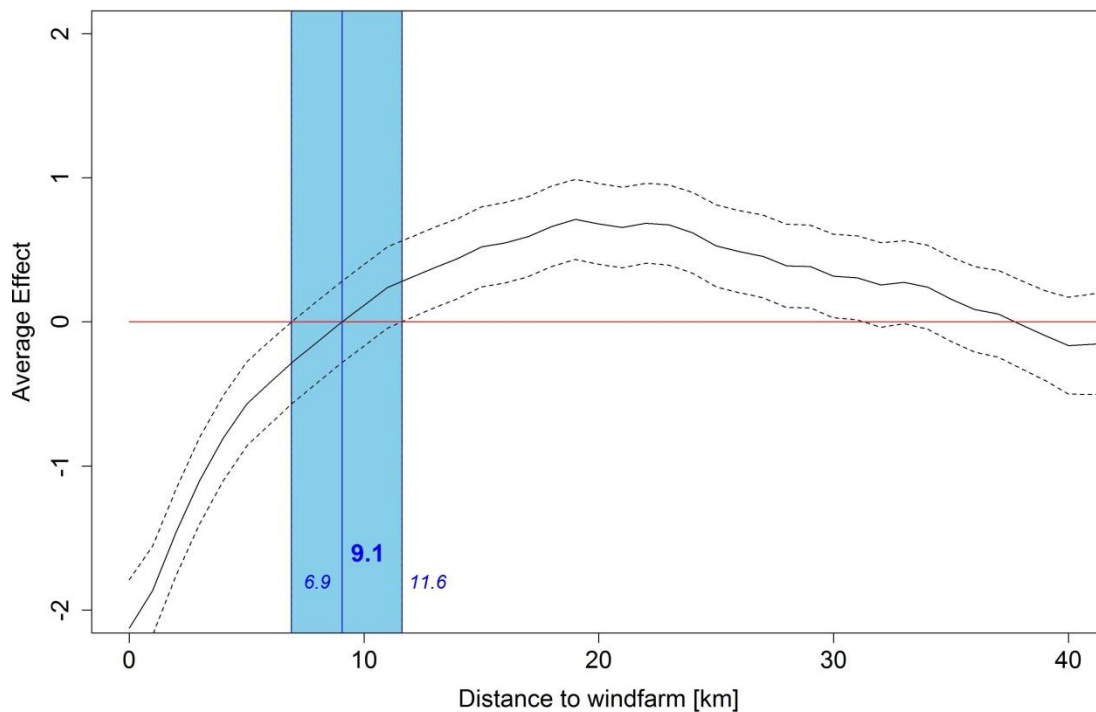


Figure 5-20 Displacement effect for spring - northern area. Dotted line depicts the upper and lower 95% CI from the model curve (solid line); the blue shaded box indicates the range of the possible effect. Vertical blue line shows the intersection between the zero-line and the model curve.

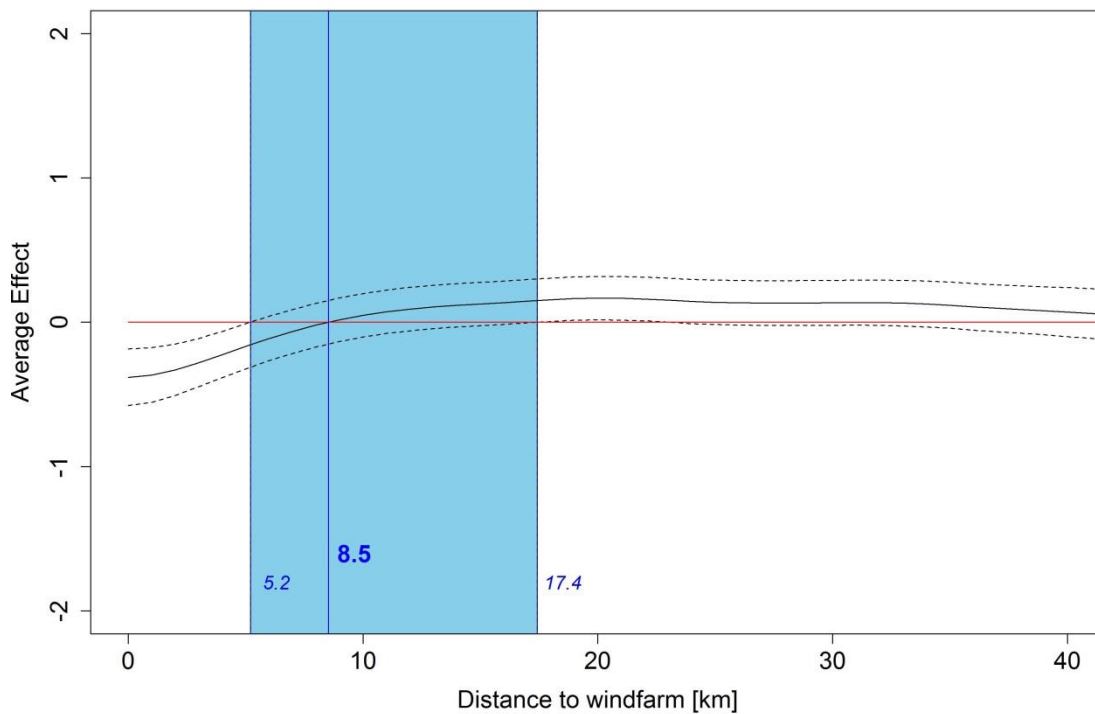


Figure 5-21 Displacement effect for spring - southern area. Dotted line depicts the upper and lower 95% CI from the model curve (solid line); the blue shaded box indicates the range of the possible effect. Vertical blue line shows the intersection between the zero-line and the model curve.

For winter, displacement effects varied strongly between areas. For winter – total area, the effect was estimated at 19.3 km, with large confidence intervals (10.2 – 22.4 km, Figure 5-22). The model curve showed a steep increase for the first few kilometres and then started to level off and show a plateau with more or less constant densities. Due to the shape of the curve, estimation of the effect (intersection of zero-line with model curve) is very susceptible to small changes in model setup, which is also shown by the large confidence intervals.

For winter – northern sub-area, the displacement effect was estimated at only 3.3 km (confidence interval from 2.1 to 4.8 km, Figure 5-23).

For winter –southern sub-area, the displacement effect was estimated at 23.1 km (confidence interval from 18.9 to 31.5 km, Figure 5-24).

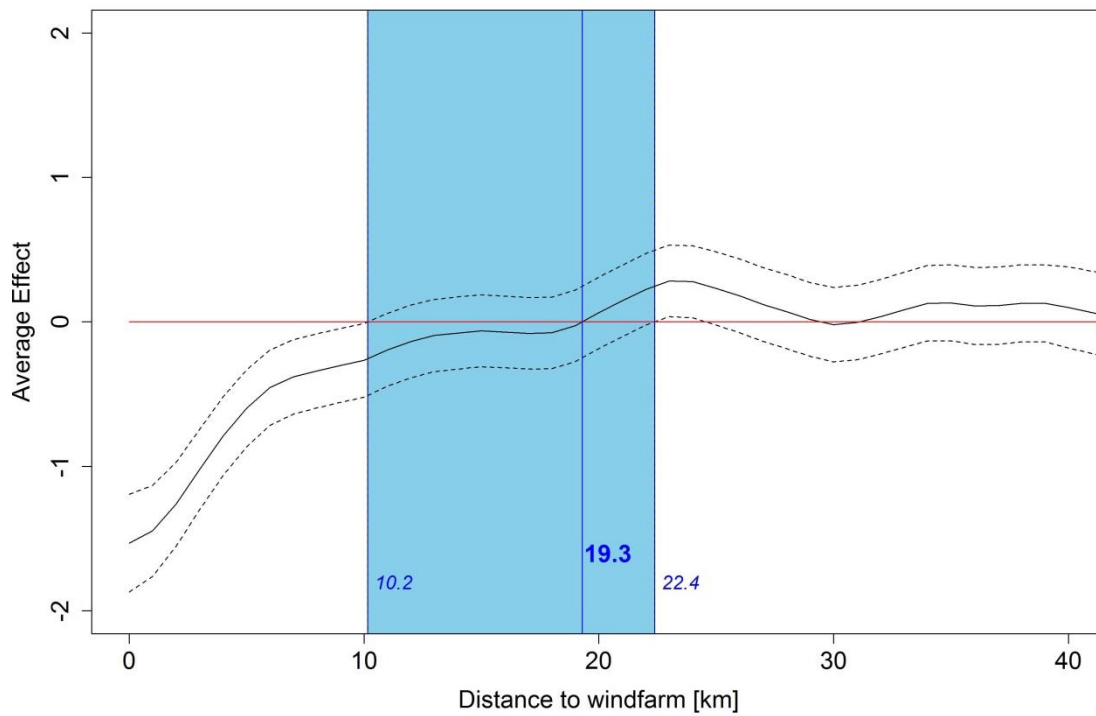


Figure 5-22 Displacement effect for winter - total area. Dotted line depicts the upper and lower 95% CI from the model curve (solid line); the blue shaded box indicates the range of the possible effect. Vertical blue line shows the intersection between the zero-line and the model curve.

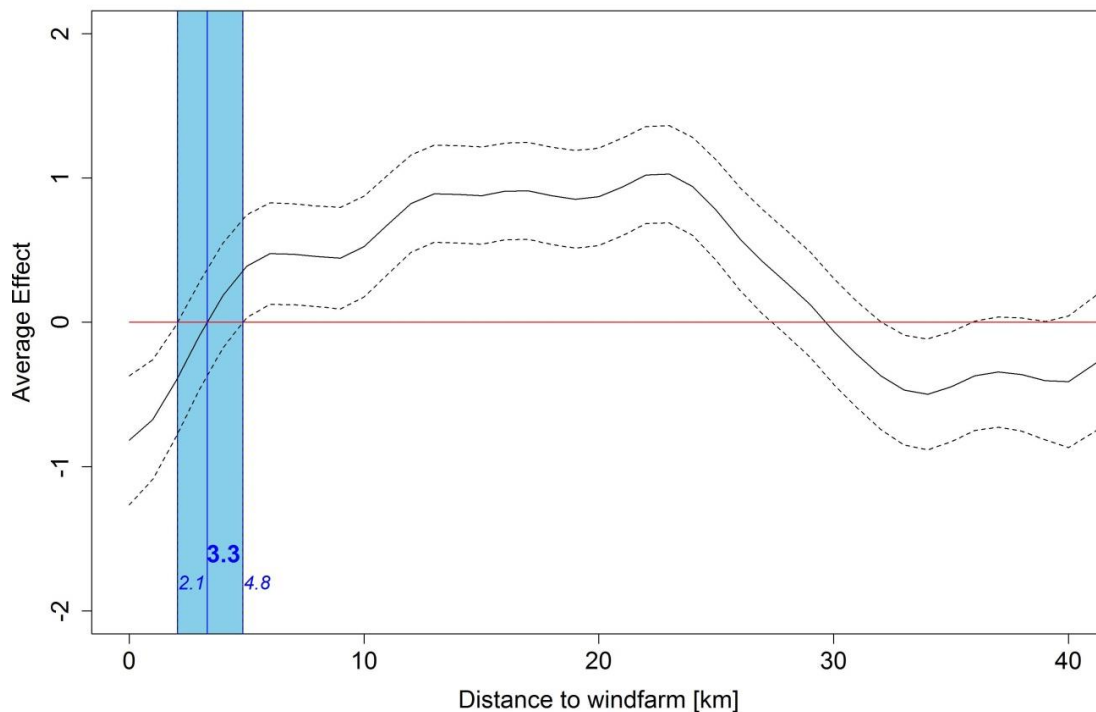


Figure 5-23 Displacement effect for winter – northern sub-area. Dotted line depicts the upper and lower 95% CI from the model curve (solid line); the blue shaded box indicates the range of the possible effect. Vertical blue line shows the intersection between the zero-line and the model curve.

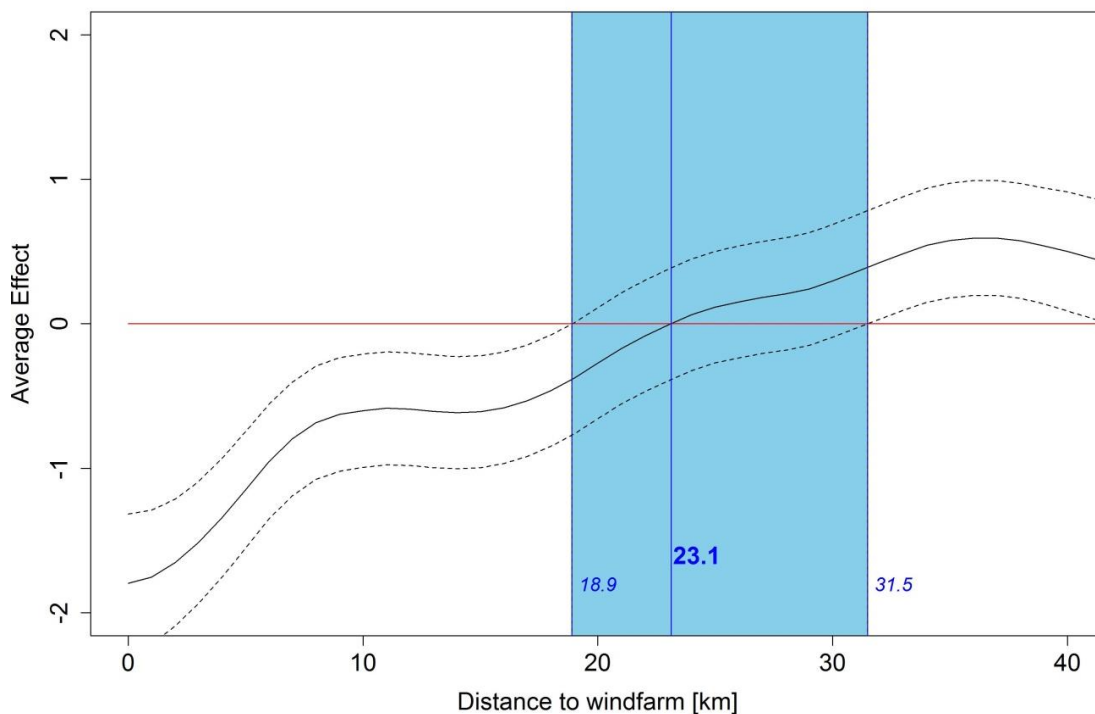


Figure 5-24 Displacement effect for winter – southern sub-area. Dotted line depicts the upper and lower 95% CI from the model curve (solid line); the blue shaded box indicates the range of the possible effect. Vertical blue line shows the intersection between the zero-line and the model curve.

5.4.2 Theoretical habitat loss

The theoretical habitat loss was calculated for the spring phase only, as well as separately for the northern and southern sub-areas.

Using a presumed wind farm footprint of 8 x 8 km (compare Garthe et al. 2018), the theoretical habitat loss was calculated at 5 km for the total study area as well as for the northern sub-area alone. For the southern sub-area, which showed a similar displacement distance but a much weaker displacement effect given by the flatter curve, the theoretical habitat loss was calculated at only 2 km.

If the calculation is applied to a rectangular wind farm layout with edge lengths of 3 x 20 km, the theoretical habitat loss does not change. Even a circular wind farm layout has only slightly greater effects and no influence within the 500 m classes.

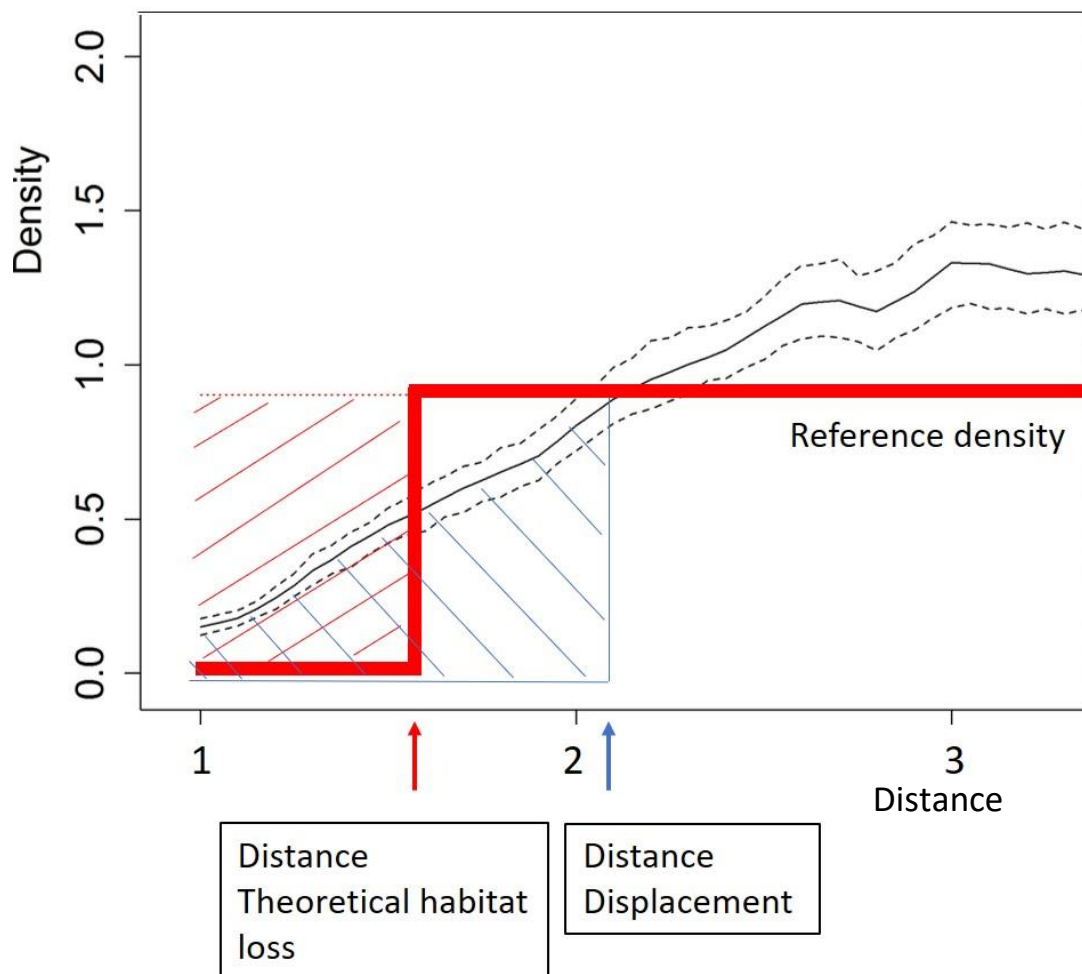


Figure 5-25 Example of calculation of displacement distance and theoretical habitat loss. The reduced bird density up to the displacement distance is used to calculate the area of theoretical habitat loss given the reference density. The result is given as radius around a model OWF.

5.5 Results of the before-after approach to estimate displacement

The before-after spatio-temporal model was used to calculate the areas of significant net gain and loss within the study area after the construction of wind farms.

Using the 'before' time period 2001 – 2005 and the 'after' period 2017 and 2018, an area of 1.231 km² was calculated where a significant reduction of diver densities was found, and an area of 715 km² where a significant increase of diver densities was found (Figure 5-26).

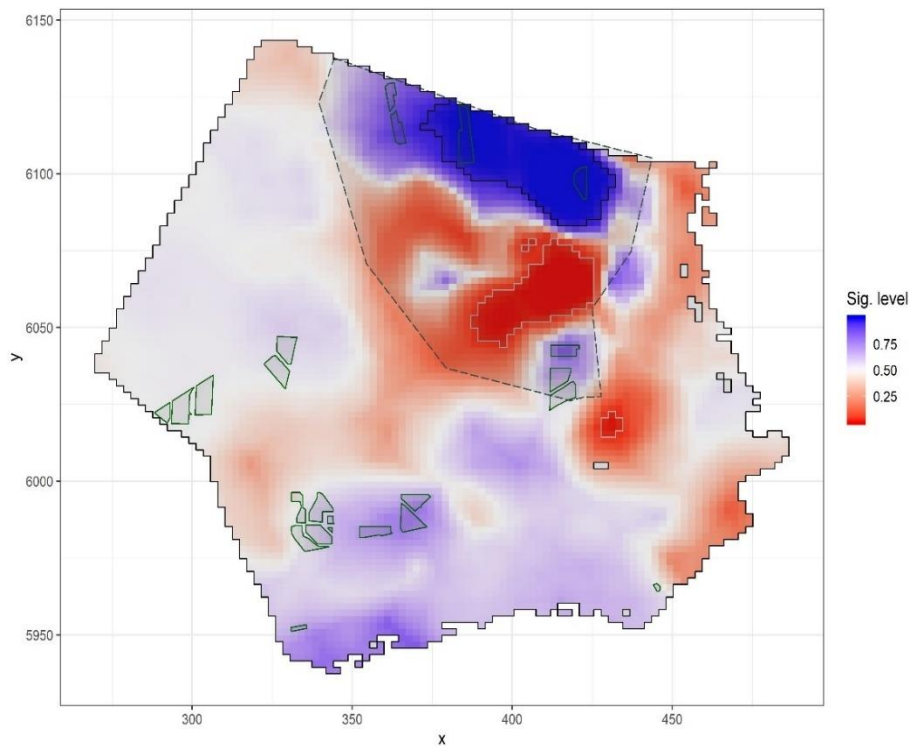


Figure 5-26 Level of significance for comparison of diver densities between before-period (2001 – 2005) and after-period (2017 & 2018). Encircled is area of significant net gain (red) or loss (blue). Dashed line depicts diver main concentration area (BMU 2009).

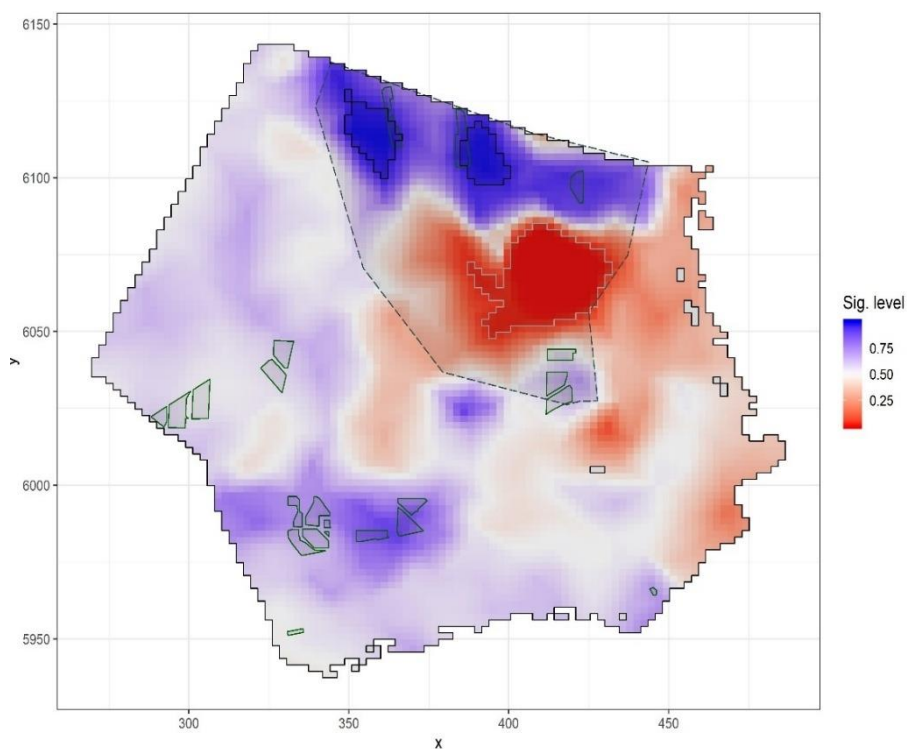


Figure 5-27 Level of significance for comparison of diver densities between before-period (2008 – 2011) and after-period (2017 & 2018). Encircled is area of significant net gain (red) or loss (blue). Dashed line depicts diver main concentration area (BMU 2009).

Using the ‘before’ time period 2008 – 2011 and the ‘after’ period 2017 and 2018, an area of 480 km² was calculated where a significant reduction of diver densities was found, and an area of 945 km² where a significant increase of diver densities was found (Figure 5-27). Here, the area of significant reduction was about a third of the area that was calculated using the time period 2001 – 2005 as reference.

The figures also show that the significant reduction was not equal in all directions around OWFs. In Figure 5-27, significant reductions were found towards the western side of OWF “Sandbank” and towards the eastern side of OWF “DanTysk”, while in Figure 5-26 significant reductions were found around OWF “DanTysk” and OWF “Butendiek” and in the area between the two OWF.

The two ‘before’ time periods 2001 – 2005 and 2008 – 2011, in comparison with the ‘after’ time period, 2017 & 2018, were also used for the estimation of displacement distances for the northern sub-area. For the northern area, displacement distances were significantly different, up to 11 km (p-value: 0.051) when using the ‘before’ time period 2001 – 2005, and up to 13 km (p-value: 0.408) when using the ‘before’ time period 2008 – 2011 (Figure 5-28, Figure 5-29).

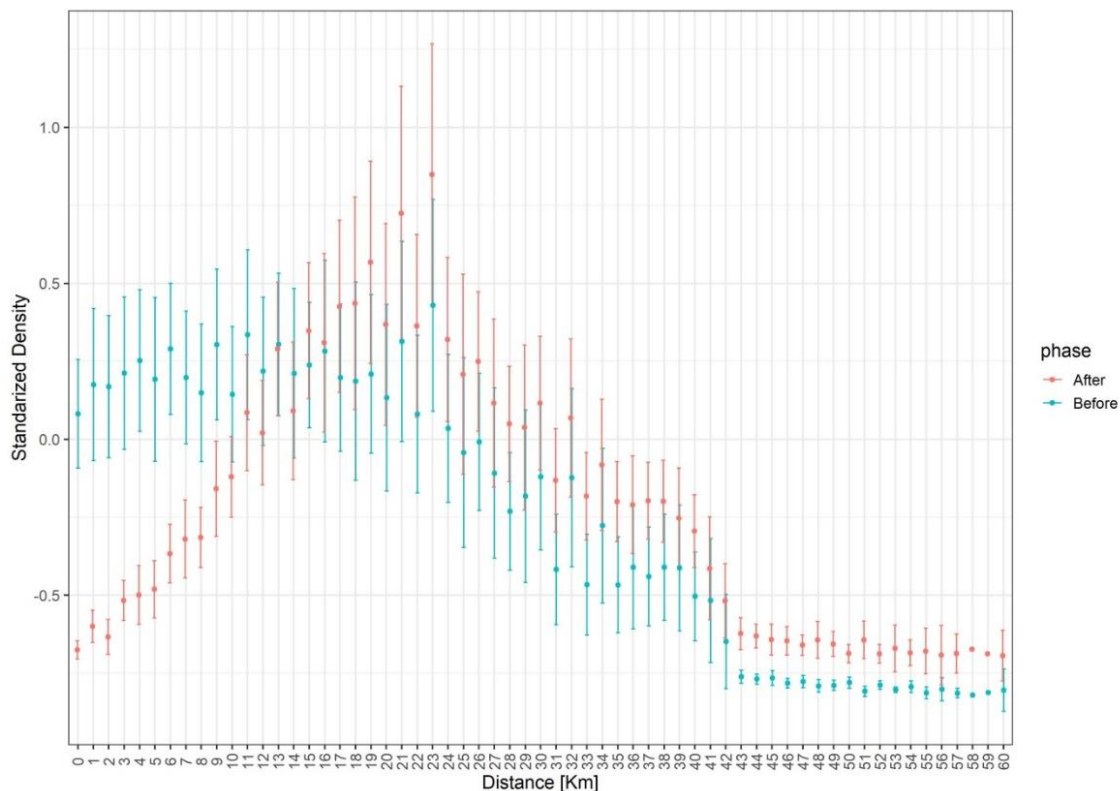


Figure 5-28 Densities (mean $\pm$ 95% CI) from spring data 2001 – 2005 (before) and 2017/18 (after) for estimation of the displacement effect.

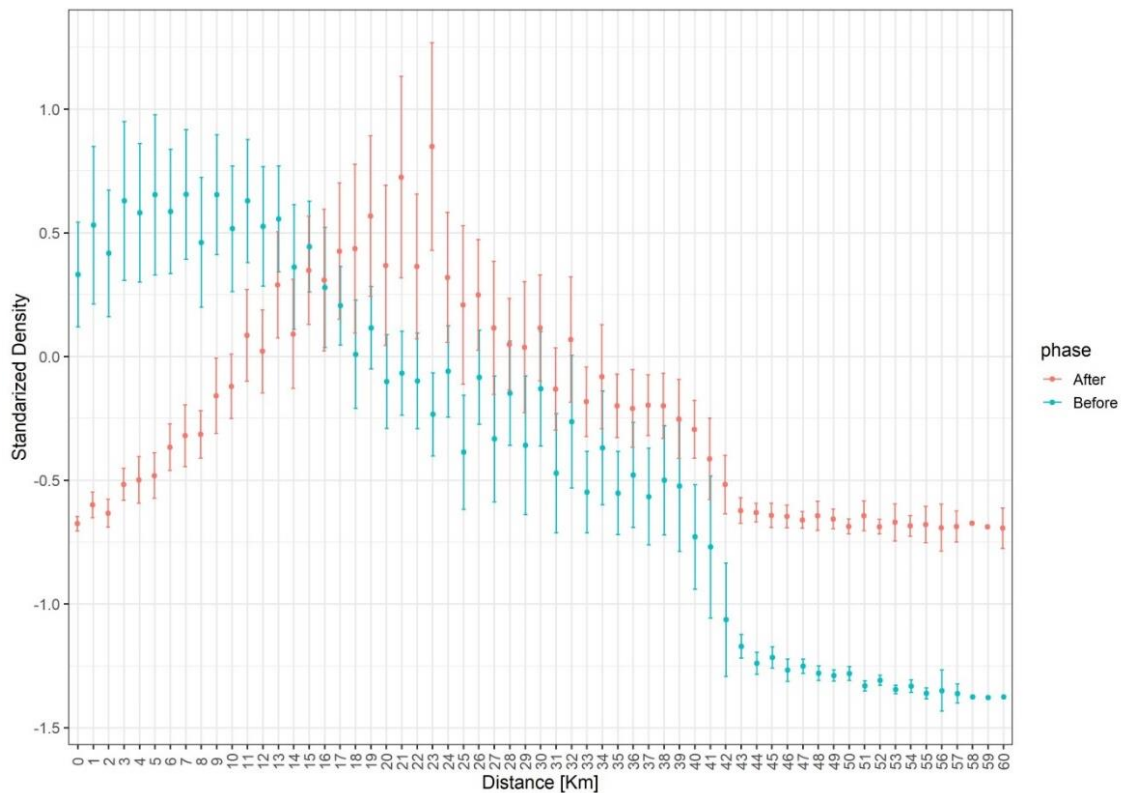


Figure 5-29 Densities (mean $\pm$ 95% CI) from spring data before (2008 – 2011) and after (2017/18) for the estimation of the displacement effect.

Densities and spatial distribution varied somewhat between the three time periods in spring (2001 – 2005; 2008 – 2011; 2017 & 2018). During all time periods, the highest estimated densities were found within the main diver concentration area, but for 2008 – 2011 densities were rather low as compared to 2001 – 2005 and 2017 & 2018 (Figure 5-30, Figure 5-31, Figure 5-32).

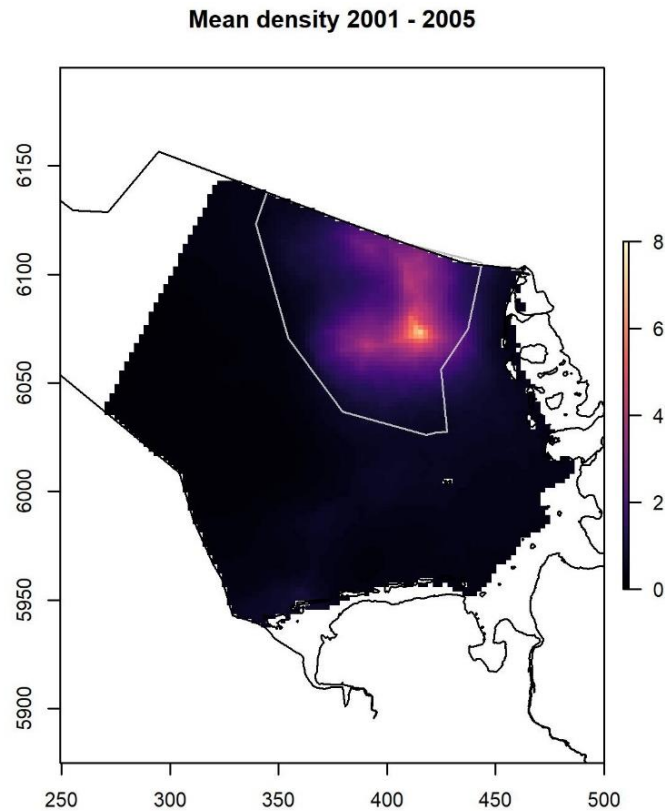


Figure 5-30 Diver densities for the time period 2001 - 2005 during spring. Grey line encircles the diver main concentration area (BMU 2009).

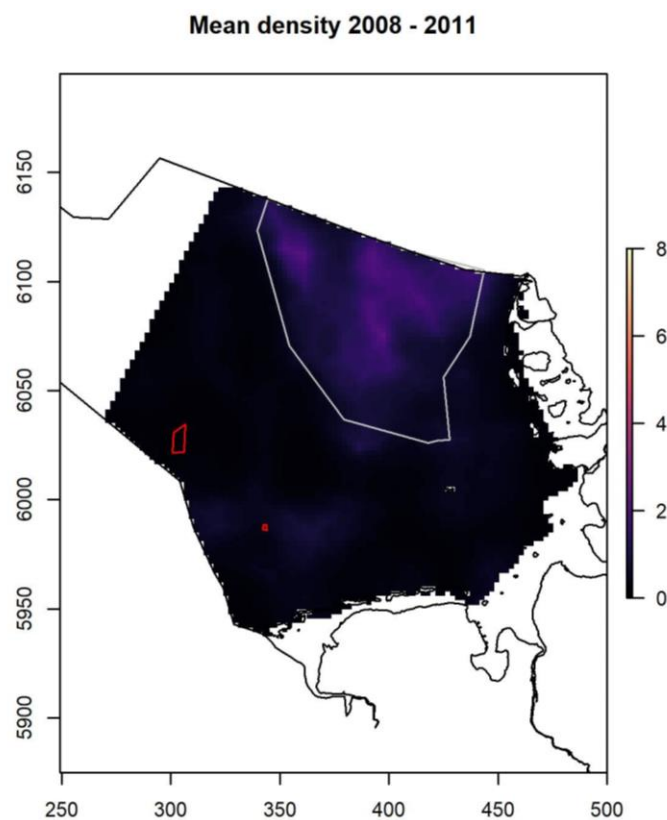


Figure 5-31 Diver densities for the time period 2008 - 2011 during spring. Grey line encircles the diver main concentration area (BMU 2009).

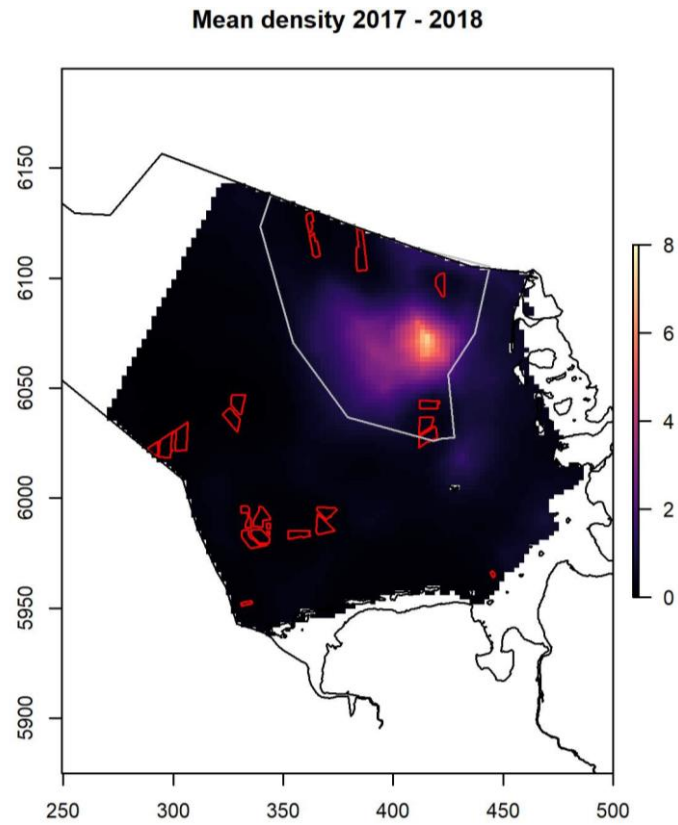


Figure 5-32 Diver densities for the time period 2017 - 2018 during spring. Grey line encircles the diver main concentration area (BMU 2009).

6 DISCUSSION

The standard concept for monitoring the environmental impacts of the expansion of offshore wind farms, conceived and coordinated by the BSH, has created an excellent and homogeneous data basis, collected by the authors of this study themselves. This allows us to create reliable distribution models for divers in the German North Sea by also adding data from other sources like research projects. The study confirms the significance of the 7,000 km² main concentration area (BMU 2009), which during spring holds around 60 % of the divers of the German North Sea throughout the study period (Figure 5-30 to Figure 5-32). This area has been affected by the construction of six wind farms distributed over four locations. This study reveals marked displacement effects on divers over 10 km. While this demonstrates a strong response of these species to anthropogenic structures and a substantial part of this area is now subject to displacement effects, diver numbers remained at the same high level after construction of the wind farms.

6.1 Population size development pre and post-construction

The causes of a population change are usually hard to investigate, especially in migratory seabirds. Impacts might occur during wintering, migration or on the breeding grounds (e.g. Newton 2004). As divers are long-lived birds, any negative impacts of OWF on their fitness might only show after several years (Sæther & Bakke 2000, Jenouvrier et al. 2005, Sim et al. 2011) and on short-term only if survival is directly affected. Nevertheless, the data set of 17 years of the present study offers a great opportunity to uncover population changes that could potentially be related to the development of offshore wind power, as offshore wind farms in the German EEZ have been part of the environment for more than ten years since the construction of alpha ventus in 2009, and for more than five years after the offshore wind farm expansion within the main concentration area of divers.

Using this large dataset of aerial surveys, no significant trend in spring numbers of divers appeared and we did not detect a decline within the German North Sea after the construction of wind farms in the area despite a clear displacement effect. Although the population development in the northern part of the study area might indicate a slightly positive trend for the period from 2008 until 2012, post-construction (2014-2018) population size is similar to the first years (2001-2004) of the study period (Figure 5-3). Overall, abundances were on an average level during this time period, except for 2017 when numbers were lower. This confirms that no negative trend is apparent and that short-term declines can probably be attributed to yearly fluctuations, as can also be found during pre-construction years. It is also remarkable that diver numbers in the main concentration area stayed relatively constant during this time (Figure 5-5). For the southern area, numbers were comparatively low overall, with the highest abundances for years 2011 and 2012, and a subsequent return to pre-construction levels.

For the winter season we found an increase in diver numbers especially during the most recent years (2015, 2016, 2018). The results indicate that especially during the last three years since 2016 the southern sub-area was used more frequently during winter as compared to the early years. Similar to several other bird species, seasonal patterns of divers in the North Sea may have shifted due to climate change (Visser & Both 2005; but see Keogan et al. 2018). Therefore, movements of divers into their main spring staging area might occur earlier nowadays than in previous years due

to shifts in prey phenology. For cod, for example, changes in distributions in the North Sea over the last decades can be attributed to both climate change and fishing pressures (Engelhard et al. 2014). The observed pattern is also supported by the finding that during spring, diver abundance in the southern sub-area tended to decrease in recent years. For our analyses, we used species-specific seasons defined by Garthe et al. (2007) based on historic data of diver phenology. Our results indicate that for future analyses an adjustment of the seasons used so far for divers could be considered. A fine-scaled analysis of seasonal patterns of diver abundance could give more insight into this topic.

6.2 Spatial distribution of divers within the German North Sea

During spring, the highest densities of divers were consistently found within the main diver concentration area, which was defined by BMU (2009) based on an analysis described by Garthe et al. (2007), including data between 2000 and 2009. On average about 60 % of all divers were found to be present within this area during spring during all years. However, in recent years, after the construction of wind farms in this area, distribution varied less within this area and was strongly localised in a central area without wind farms. It has been proposed that due to several wind farms already built within the main concentration area, divers choose a central location far away from wind farms. Densities in the remaining undisturbed area have become higher in this location compared to most previous years (except for 2003, when very high densities in this area were found). Also, the results of our cross-validation show that during the last few years the prediction of diver distribution was very reliable, probably due to the existence of wind farms, whereas in former years divers were more widely distributed and areas of higher concentrations were less predictable.

Local concentrations (up to 3 Ind./km²), found north of the East Frisian Islands in some years (e.g. 2012), were not very stable and no clear pattern between the location of wind farms and diver distribution was apparent.

Due to the fact that the number of divers has not decreased but is distributed over a smaller area, as they avoid the wind farms and their immediate surroundings, the density within the unaffected areas must have increased accordingly. This raises the question of how high the carrying capacity of the area is for divers and when a capacity limit is reached above which density dependent limitations occur, e.g. related to food availability, or intra-specific competition. Such effects could reduce the body condition of individuals and might lead to reduced survival or breeding success (e.g. Szostek & Becker 2015). Although our data show that the maximum densities in the area used by divers today are on average higher than before the wind farms were expanded, we have no indication that the carrying capacity limit has been reached. This is also supported by the fact that no decrease in the population is apparent. Whether the less variable spatial distribution of divers might have verifiable fitness consequences or whether there is enough suitable habitat with sufficient food resources (Gill et al. 2001), can only be determined on the basis of physiological studies and a longer time series covering more generations of divers.

In winter, the main concentrations of divers are found in coastal areas, especially along the coast of Schleswig-Holstein (see also Mendel et al. 2008) and the East Frisian Islands. In some years, the main concentration area (BMU 2009) is also of high importance, but this varies throughout the study period, possibly related to prey abundance. Similar to spring, concentrations north of the East Frisian Islands were found in some of the early years, but these shifted to a more coastal occurrence with the construction of wind farms further north (Cluster “Nördlich Borkum”). This phenomenon is also reflected in the apparently large displacement distance in the southern area in winter. Whether this shift is a consequence of wind farm expansion or a natural shift to the coastal area is not fully clear. However, densities during winter are usually much lower than during spring, and thus fewer birds are impacted, even though displacement distances are large. The majority of divers registered during spring therefore spend the winter period outside of the German North Sea (e.g. Great Britain, Netherlands; divertracking.com). A significant part of individuals present in the German North Sea during spring uses the area as a short-term stop-over site, while others utilize it for several weeks or months as resting site before migrating towards their breeding grounds (Dorsch et al. 2019, www.divertracking.com).

The main concentration area (BMU 2009) may be called relatively small, as telemetry studies showed that divers which have been tagged in this area, move over much larger areas and individual home ranges often greatly exceed the size of the main concentration area (Dorsch et al. 2019). Divers tagged in the main concentration area were highly mobile and commuted between Danish and German staging areas. Still, even though the main concentration area is only a part of the diver staging areas in the eastern North Sea, it maintains its function and divers have not redistributed to other areas.

6.3 Displacement effect and theoretical habitat loss

The results show a strong displacement effect on divers from offshore wind farms. Although the spatial distribution of the birds indicates that avoidance is not equal in all directions from the wind farm, the calculation of an avoidance distance aims at estimating a distance to the wind farm up to which a decrease in bird densities due to the existence of the wind farm is verifiable. Following our approach and taking all available data into account, we found displacement effects that varied between seasons (spring, winter) and areas (north, south). In spring, when the highest numbers of divers occur in the Eastern German Bight, average displacement for the total study area was estimated at 10.2 km and was somewhat lower for the two sub-areas. During winter, large differences were found for the two sub-areas with only 3.3 km in the northern area and even 23.1 km in the southern area. For winter, it has to be taken into account that bird densities were 3.5-times lower in the study area than in spring, making displacement effects more difficult to estimate precisely. Similar problems exist for the southern area in spring, which shows much lower densities and a more variable distribution than the northern area. Partly larger confidence intervals confirm greater uncertainty. The fact that during winter the curve for the southern sub-area levels off between approx. 9 and 17 km suggests that no clear wind farm effect can be derived at these larger distances. Despite these limitations, the results indicate different bird responses to the wind farms, depending both on the area and the season.

This study, as well as the previous study from Petersen et al. (2014), showed that the detected displacement effects are not equal on all sides but show considerable variation in different

directions of the wind farm (see Figure 5-18, Figure 5-26, Figure 5-27). The reasons for this are not yet understood as it cannot be assessed yet whether this is a constant effect or whether this reflects a temporal variation which is not fully picked up by infrequent snapshots of the distribution from aerial surveys. Food abundance varies locally and between years and this resulted in a variable distribution of divers before construction of any OWF, as the birds needed to follow these food sources. Diver distribution is also strongly affected by ship traffic (Burger et al. 2019, Mendel et al. 2019), and this factor was not included in the current analysis. Tracking data of red-throated divers equipped with satellite transmitters also revealed that the response to offshore wind farms is related to the visibility during daylight (Dorsch et al. 2019). Furthermore, the displacement distance has been found to be larger at night, when the turbines are illuminated, than during the day. This gives some indication that variable external clues define the displacement response. Overall, displacement distances were similar between aerial survey data and tracking data (Nehls et al. 2018), suggesting that the specific conditions during which aerial surveys took place (good visibility, calm weather) did not lead to an overestimation of displacement. On the other hand, divers might balance their response in relation to differences in habitat quality (*sensu* Frid & Dill 2002) which could lead to a spatially structured response with birds approaching the wind farms at closer distances in patches of high-quality habitats. The hydrography of the diver staging area west of Schleswig-Holstein is characterized by frontal zones of mixing water bodies from the river Elbe and the North Sea (Skov & Prins 2001). The frontal system is assumed to cause a spatially and temporarily variable distribution of fish which serve as food for red-throated diver (Kleinschmidt et al. 2019) and leads to a patchy distribution of divers and other seabirds in this area.

For an assessment of habitat loss, especially with regard to marine spatial planning, the translation of displacement distances, which represent a gradient, into a theoretical habitat loss around a standard wind farm, is necessary, and potential uncertainties in the estimation need to be discussed. It is, however, important to emphasise that the theoretical habitat loss does not equal a total loss of habitat for divers, but is rather a degradation in habitat quality, since mobile resources like fish from these areas are still available to divers (cf. 4.3.3.2). Due to the higher diver densities and lower uncertainties in the spring data subset, we focused on the spring period and the total study area for the estimation of a theoretical habitat loss. Displacement effects might also depend on the amount of available habitat that is suitable to divers. Due to shifts in food abundance between winter and spring (Kleinschmidt et al. 2019), the available habitat might vary between seasons and areas and birds will adjust their distribution accordingly. However, more data on habitat suitability and feeding behaviour of divers would be needed to better be able to interpret the observed variations in displacement effects.

The theoretical habitat loss for the spring season was estimated at 5 km radius for the total study area and also for the northern sub-area. For the southern area a lower value was estimated at 2 km. Diver densities in the southern area were considerably lower compared to the northern area and showed more variable aggregations between years as compared to the North. The estimated displacement effect for the southern area was weaker and more difficult to estimate due to the flat model curve. However, we do not yet fully understand the causes of these differences in disturbance radius and habitat loss. Possible explanations could be factors such as local food availability, seasonal usage by divers (cf. Dorsch et al. 2019) and diver density, as well as abiotic factors like the distance to the shoreline, currents, water depth, sediment and many more. Since these biological and environmental factors will differ strongly between different areas, the

transferability of these values to other (sub-) areas, such as different regions of the North Sea outside the German Bight or the Baltic Sea is therefore limited.

The theoretical habitat loss calculated in this study is thus on average considerably higher than the previously assumed 2 km radius around wind farms (BSH 2013). This results in higher numbers of divers being affected by wind farm development than previously thought. The actual number of birds being affected by habitat loss depends on the exact location of the wind farm and the size and shape of the wind farm footprint. The results suggest only very small differences in habitat loss depending on the shape of the footprint, with slightly lower habitat loss for square designs as compared to a rounded shape.

6.3.1 Limitations

An important issue in the estimation of the displacement effect is the decision on a cut-off point or reference value, indicating the range of the disturbance. Different approaches have been applied: Heinänen et al. (2016) and Mendel et al. (2019) used data from after construction of wind farms and estimated displacement based on the shape of the model curve (using the 1st derivate thereof). In the past, before-after approaches were also used (Petersen et al. 2014, Webb et al. 2015, Garthe et al. 2018).

However, several limitations exist with regard to the before-after approach: using densities from before-construction as reference assumes that no changes in overall abundance have occurred. Especially when using a relatively short period of data of only a few years, this may not be a reliable estimate of the before-situation. Long-term changes in abundance or distribution due to e.g. climate change might thus result in inappropriate reference values. Even when scaling densities for the two time periods, changes in spatial distribution due to changes in the environment (other than OWF) might still bias the results and the spatial limitation up to which an effect is considered plays an essential role.

Our main modelling approach for the calculation of displacement uses all years starting when the first wind farm (alpha ventus) was under construction and all available data covering distances up to 100 km to single wind farms were considered. For every season and year, distance to wind farm was recalculated based on the current wind farm status. From this large dataset spanning 10 years and a large study area, we considered the average density by season to be the best available proxy and this was used as the reference value for calculating displacement distances.

Another problem in the data used for the present analysis is the change in survey techniques from visual aerial surveys to digital aerial surveys. Distance correction is needed for a correct estimation of bird densities during visual surveys. In our model, we had a good overlap in the use of all different survey techniques, which allowed us to further correct for any differences between them. Therefore, in addition to distance sampling, the model was able to correct for differences in survey techniques which improved accuracy of estimation and could indicate the difference between the different survey techniques. This result is in line with findings by Mendel et al. (2018) and Zydalis et al. (2019) and proves that all techniques are capable of reliably detecting divers and that there are no major differences between them (including visual aerial surveys).

6.3.2 Comparison of displacement results with before-after approach

Displacement distance from the before-after approach was higher than from the distance model using all data. For the distance model, wind farms were assigned as having an impact on divers when construction started during the same phase (e.g. spring 2015). As this assignment was not exact in all cases, a few surveys were assigned to construction phase when no construction work had taken place yet. For red-throated divers it is known that the vertical structures are a main disturbance factor, as they may adjust their displacement distance depending on visibility (Heinänen et al. unpublished data). However, ship traffic also plays an important role and can lead to disturbance ranging over several kilometres (Burger et al. 2019, Mendel et al. 2019, Fließbach et al. 2019). Including data from construction phase in the after-period is therefore reasonable. However, the level of disturbance during construction might vary, depending of the amount of ship traffic and progress of construction work.

In contrast to the distance model, the before-after approach for the northern area included only fully operational wind farms in the after-period, and thus a slightly stronger displacement effect can be expected. However, when comparing the two before time-periods (2001 – 2005 & 2008 – 2011), a difference of about 2 km in displacement distance, as well as different area gains and losses were found, indicating the uncertainty of this approach. Because of these shortcomings, it was decided to use the main approach (only-distance model), which included all available data to estimate displacement distance. Furthermore, the before-after approach cannot be applied in most areas other than the northern sub-area because wind farms close to each other were constructed over the whole study period since the construction of the first wind farm ‘alpha ventus’. This makes a separation between baseline and construction/operational phases impossible. Thus, a model assessing the effect of the distance to wind farms for every phase and year and using the average density over 10 years as reference represents a more realistic scenario.

6.4 Comparison with other studies

Although earlier studies used different approaches and statistical methods, relatively similar values for displacement range and stock size were found. There are solid indications, that the basic data and the restrictions on the data set used has the strongest influence on the results, rather than the statistical method. Especially regarding diver abundance, the present study showed that when survey effort and coverage of the study area are high, results are relatively consistent between studies but when coverage was inconsistent, so were the results (compare Schwemmer et al. 2019 and Garthe et al. 2015). Both Schwemmer et al. (2019) and the present study found no clear trends in numbers after 2010 in the main concentration area. For the whole German North Sea, Schwemmer et al. (2019) found an increase in spring numbers until 2012 followed by a constant decline until 2017 (no data for 2018). A similar pattern was found in our dataset only when analyzing the southern sub-area in spring separately and not for the whole German North Sea, where an increase in abundance for 2018 was seen. Schwemmer et al. (2019) explained the maximum number of divers in spring 2012 by unusually high numbers of divers in the Natura2000- area “Borkum Riffgrund”. Despite the good coverage of this area, the magnitude of this high diver concentration cannot be confirmed by our dataset, even though 2012 was the year with highest

diver densities in the southern sub-area as well. This indicates, that the most important issue regarding population size estimations is the data base which is used. Differences in other studies compared with our analyses might result from the fact that part of the aerial survey data from the wind farm monitoring which has been used in our study was not available to the others studies. Also, our analysis included the year 2018, which showed a higher diver abundance than the previous years, especially in spring. We further suspect that including ship surveys might cause substantial bias in the analysis, as ship surveys cause disturbance to divers (Bellebaum et al. 2006, Schwemmer et al. 2011), and cover far smaller study areas. We therefore do not consider data from ship observations suitable for the analysis of displacement effects for this species.

Since the monitoring programs for the development of offshore wind farms are connected with an extensive data collection based on aerial surveys, we were able to almost completely cover the study area and only four years (out of 16) were covered by less than 50%, all of them before 2009. Our results show no clear population trend for the spring population of divers either in the whole German North Sea or in the diver concentration area. Thus, our results contradict the hypothesis of Schwemmer et al. (2019) and Garthe et al. (2018) that the expansion of offshore wind power has been accompanied by a significant decline in diver population in this area.

Regarding displacement, recent studies in the German Bight using a large study area consistently found strong effects up to about 10 - 15 km distance from the closest OWF and weaker effects reaching even further (Heinänen 2016, Garthe et al. 2018, Mendel et al. 2019; Table 4-1). Studies using smaller survey areas (e.g. Welcker & Nehls 2016), commonly found weaker displacement effects, indicating that a large survey area around a wind farm is needed for a proper analysis of displacement.

An important difference between studies (Table 6-1) is the definition of a reference value to use for the calculation of displacement distance. We have shown that when using baseline data as reference, displacement effects varied depending on the “before” time-period used. Therefore, if no long-term baseline dataset is available, displacement should better be estimated based on the model curve from a model including distance to wind farm. Variations in displacement distance can have multiple causes, as also local or seasonal factors may play a role in the extent of disturbance. These factors are currently not well understood and more research into local and seasonal patterns is needed for a proper understanding of the variation found.

In contrast to previous studies using a Frequentist approach (MRSea or GAMM), we used a state-of-the-art Bayesian approach (GMRF with INLA-SPDE) in our study, which can include prior information in the modelling process and does not require predictors for a reliable estimation of abundance and spatial distribution of divers. This, however, requires a fair survey coverage of the study area and some overlap between data collection methods during at least some of the years investigated.

Table 6-1 Overview of recent studies investigating displacement of divers using a large survey area in the North Sea.

Study	Study area/OWF	Estimated response distance	Data	Statistical method	Reference for displacement effect
Petersen et al. 2014	Horns Rev 2, Denmark	5 - 6 km	Aerial surveys	Pre- MRSea	Before-after comparison
Webb et al. 2015	Lincs, UK	2 - 6 km	Aerial surveys	MRSea	Before-after comparison
Heinänen et al. 2016	North-eastern German Bight	10 - 15 km	Aerial surveys	GAMM	Comparison with abundance in 15 - 20 km & > 20 km distance from OWF
Garthe et al. 2018	German North Sea	9 - 12 km	Aerial and ship surveys	GAMM	Before-after comparison
Mendel et al. 2019	German North Sea	16 km	Aerial and ship surveys	GAMM	Model curve in operational phase (first derivative of the smooth)
This study	German North Sea	9 - 10 km	Aerial surveys	INLA-SPDE (bayesian)	Intersection of model curve with average density

The calculated theoretical habitat loss around a wind farm in the German North Sea of 5 km as calculated for the total study area during spring was similar to an earlier study (5.5 km; Garthe et al. 2018) and considerably higher than the previously assumed 2 km (BSH 2013). However, for the southern sub-area, habitat loss in spring was estimated at only 2 km. Only slight difference in radius for theoretical habitat loss were found when using different footprints, including the shape of an OWF cluster. However, by aggregating OWF in clusters, the overall number of divers being affected will be reduced relative to single OWF of the same size.

6.5 Conclusions

This study gives an overarching picture of how the diver population has responded to the expansion of wind energy within the German North Sea and is intended to be considered as a whole. As there is evidence that each of the observed issues - population size, spatial distribution and displacement - are related to each other, we emphasize that they should be regarded together. While an unexpectedly strong avoidance reaction by divers to windfarms was determined, in some sub-regions more than in others, the overall abundance of divers within the region has remained stable and without any clear trend within the inter-annual fluctuations. No effect was found at population level, indicating that the carrying capacity of the available habitat - especially of the northern sub-region - has not been reached. However, further studies into habitat quality and food availability could help our understanding of the delicate population dynamics in the area. Furthermore, studies into whether the seasonal phenology of divers has shifted over the years could further our understanding of their usage of the German Bight now and in the future.

7 LITERATURE

- BACHL, F. E., LINDGREN, F., BORCHERS, D. L. & ILLIAN, J. B. 2019. inlabru: an R package for Bayesian spatial modelling from ecological survey data. *Methods in Ecology and Evolution* 10:760–766.
- BELLEBAUM, J., DIEDERICH, A., KUBE, J., SCHULZ, A. & NEHLS, G. 2006. Flucht- und Meidedistanzen überwinternder Seetaucher und Meeresenten gegenüber Schiffen auf See. *Orn. Rundbrief Mecklenburg-Vorpommern* 45, Sonderheft 1 (Tagungsband 5. deutsches See- und Küstenvogelkolloquium):86-90.
- BERNARDO, J. M., SMITH, A. F. M. & BAYES, T. 2000. Bayesian theory. Wiley, Chichester. 586 pp.
- BIRDLIFE INTERNATIONAL. 2004. Birds in Europe: populations estimates, trends and conservation status. P. (BirdLife International, Ed.). BirdLife International, Cambridge (UK). 374 pp.
- BIRDLIFE INTERNATIONAL. 2017. European birds of conservation concern: populations, trends and national responsibilities. Cambridge, UK.
- BLANGIARDO, M. & CAMELETTI, M. 2015. Spatial and spatio-temporal Bayesian models with R-INLA. John Wiley and Sons, Inc, Chichester, West Sussex. 1 pp.
- BMU. 2009. Positionspapier des Geschäftsbereichs des Bundesumweltministeriums zur kumulativen Bewertung des Seetaucherhabitatverlusts durch Offshore-Windparks in der deutschen AWZ der Nord- und Ostsee als Grundlage für eine Übereinkunft des BfN mit dem BSH. Einführung eines neuen fachlich begründeten Bewertungsverfahrens. Berlin (DEU).
- BMUB. 2017. Verordnung über die Festsetzung des Naturschutzgebietes 'Sylter Außenriff - Östliche Deutsche Bucht' (NSG SylV). Pp. 3423–3429. Bundesministerium für Umwelt, Naturschutz, Bau und Reaktorsicherheit, Berlin.
- BROOKS, S. (Ed.). 2011. Handbook for Markov chain Monte Carlo. Taylor & Francis, Boca Raton. 592 pp.
- BSH. 2003. Standarduntersuchungskonzept (StUK) für die Untersuchung und Überwachung der Auswirkungen von Offshore Windenergieanlagen (WEA) auf die Meeresumwelt. 2. Fortschreibung. Bundesamt für Seeschifffahrt und Hydrographie. Hamburg und Rostock.
- BSH. 2007. Standard - Untersuchung der Auswirkungen von Offshore-Windenergieanlagen auf die Meeresumwelt (StUK3). Bundesamt für Seeschifffahrt und Hydrographie (BSH), Hamburg & Rostock.
- BSH. 2008. Offshore-Windenergieparkvorhaben in der AWZ der deutschen Nordsee - Kumulative Betrachtungsweise der Seetauchernbestände.
- BSH. 2013. Standard - Untersuchung der Auswirkungen von Offshore-Windenergieanlagen auf die Meeresumwelt (StUK4). P. 86. Bundesamt für Seeschifffahrt und Hydrographie (BSH), Hamburg & Rostock.
- BSH. 2019. Flächenentwicklungsplan 2019 für die deutsche Nord- und Ostsee.
- BURGER, C., SCHUBERT, A., HEINÄNEN, S., DORSCH, M., KLEINSCHMIDT, B., ŽYDELIS, R., MORKŪNAS, J., QUILLFELDT, P. & NEHLS, G. 2019. A novel approach for assessing effects on distributions and movements of seabirds in relation to shipping traffic. *Journal of Environmental Management* 251:109511.
- BUSCH, M. 2015. Kurzbeschreibung der Methodenstandards von APEM Digitalflugerfassungen. P. 2. APEM Ltd.
- CAMPHUYSEN, K. C. J., FOX, T. A. D., LEOPOLD, M. M. F. & PETERSEN, I. K. 2004. Towards standardised seabirds at sea census techniques in connection with environmental impact assessments for offshore wind farms in the U.K. Koninklijk Nederlands Instituut voor Onderzoek der Zee Royal Netherlands Institute for Sea Research.
- COOK, A. S. C. P., JOHNSTON, A., WRIGHT, L. J. & BURTON, N. H. K. 2012. A review of flight heights and avoidance rates of birds in relation to offshore wind farms. P. 59. Strategic Ornithological Support Services Project SOSS-02, British Trust for Ornithology, Norfolk.

- DIEDERICH, A., NEHLS, G. & PETERSEN, I. K. 2002. Flugzeugzählungen zur großflächigen Erfassung von Seevögeln und marinen Säugern als Grundlage für Umweltverträglichkeitsstudien im Offshorebereich. *Seevögel* 23:38–46.
- DIERSCHKE, V., EXO, K.-M., MENDEL, B. & GARTHE, S. 2012. Gefährdung von Sterntaucher (*Gavia stellata*) und Prachtaucher (*G. arctica*) in Brut-, Zug- und Überwinterungsgebieten - eine Übersicht mit Schwerpunkt auf den deutschen Meeresgebieten. *Die Vogelwelt* 133:163–194.
- DIERSCHKE, V., FURNESS, R. W. & GARTHE, S. 2016. Seabirds and offshore wind farms in European waters: Avoidance and attraction. *Biological Conservation* 202:59–68.
- DORSCH, M., BURGER, C., HEINÄNEN, S., KLEINSCHMIDT, B., MORKUNAS, J., NEHLS, G., QUILLFELDT, P., SCHUBERT, A. & ZYDELIS, R. 2019. DIVER – German tracking study of seabirds in areas of planned Offshore Wind Farms at the example of divers. Final report on the joint project DIVER, FKZ 0325747 A/B, funded by the Federal Ministry of Economics and Energy (BMWi) within the framework of the 6th Energy Research Programme of the German Federal Government.
- EEG 2017, 2014. Erneuerbare-Energien-Gesetz vom 21. Juli 2014 (BGBl. I S. 1066), das zuletzt durch Artikel 1 des Gesetzes vom 17. Juli 2017 (BGBl. I S. 2532) geändert worden ist.
- ENGELHARD, G. H., RIGHTON, D. A. & PINNEGAR, J. K. 2014. Climate change and fishing: a century of shifting distribution in North Sea cod. *Global Change Biology* 20:2473–2483.
- EUROPÄISCHES PARLAMENT UND RAT DER EUROPÄISCHEN UNION. 2013. Richtlinie 2009/147/EG des Europäischen Parlaments und des Rates der Europäischen Union vom 30. November 2009 über die Erhaltung der wildlebenden Vogelarten (kodifizierte Fassung) (ABl. L 020, 26.1.2010, p.7), zuletzt geändert durch: Richtlinie 2013/17/EU des Rates vom 13. Mai 2013 (ABl. L 158, 10.6.2013, p. 193).
- FLIESSBACH, K. L., BORKENHAGEN, K., GUSE, N., MARKONES, N., SCHWEMMER, P. & GARTHE, S. 2019. A ship traffic disturbance vulnerability index for northwest european seabirds as a tool for marine Spatial planning. *Frontiers in Marine Science* 6:192.
- FRID, A. & DILL, L. 2002. Human-caused disturbance stimuli as a form of predation risk. *Conservation Ecology* 6.
- FURNESS, R. W., WADE, H. M. & MASDEN, E. A. 2013. Assessing vulnerability of marine bird populations to offshore wind farms. *Journal of Environmental Management* 119:56–66.
- GARTHE, S., MÜLLER, S., SCHWEMMER, H. & SCHWEMMER, P. 2015. Verbreitung, Jahresdynamik und Bestandsentwicklung der Seetaucher *Gavia spec.* in der Deutschen Bucht (Nordsee). *Vogelwarte* 53:121–138.
- GARTHE, S., SCHWEMMER, H., MÜLLER, S., PESCHKO, V., MARKONES, N. & MERCKER, M. 2018. Seetaucher in der Deutschen Bucht: Verbreitung, Bestände und Effekte von Windparks. Bericht für das Bundesamt für Seeschifffahrt und Hydrographie und das Bundesamt für Naturschutz, Kiel.
- GARTHE, S., SONNTAG, N., SCHWEMMER, P. & DIERSCHKE, V. 2007. Estimation of seabird numbers in the German North Sea throughout the annual cycle and their biogeographic importance. *Die Vogelwelt* 128:163–178.
- GARTHE & SCHWEMMER. 2005. Seabirds at Sea – Untersuchungen in den deutschen Meeresgebieten. *Vogelwelt* 126:67–74.
- GELMAN, A., HWANG, J. & VEHTARI, A. 2014. Understanding predictive information criteria for Bayesian models. *Statistics and Computing* 24:997–1016.
- GILL, J. A., NORRIS, K. & SUTHERLAND, W. J. 2001. Why behavioural responses may not reflect the population consequences of human disturbance. *Biological Conservation* 97:265–268.
- HEINÄNEN, S. 2016. Distribution modelling of Red-throated diver based on aerial digital surveys & hydrodynamics. Diver Workshop 2016, Hamburg.
- HEINÄNEN, S., ŽYDELIS, R., KLEINSCHMIDT, B., DORSCH, M., BURGER, C., MORKUNAS, J., QUILLFELDT, P. & NEHLS, G. unpublished data. Strong displacement of red-throated divers

- (*Gavia stellata*) from offshore windfarms confirmed by two independent data types, satellite telemetry and digital aerial surveys.
- JENOUVRIER, S., BAURBRAUD, C., CAZELLES, B. & WEIMERSKIRCH, H. 2005. Modelling population dynamics of seabirds: importance of the effects of climate fluctuations on breeding proportions. *Oikos*:511–522.
- KEOGAN, K., DAUNT, F., WANLESS, S., PHILLIPS, R. A., WALLING, C. A., AGNEW, P., AINLEY, D. G., ANKER-NILSSEN, T., BALLARD, G., BARRETT, R. T., BARTON, K. J., BECH, C., BECKER, P., BERGLUND, P.-A., BOLLACHE, L., BOND, A. L., BOUWHUIS, S., BRADLEY, R. W., BURR, Z. M., CAMPHUYSEN, K., CATRY, P., CHIARADIA, A., CHRISTENSEN-DALSGAARD, S., CUTHBERT, R., DEHNHARD, N., DESCAMPS, S., DIAMOND, T., DIVOKY, G., DRUMMOND, H., DUGGER, K. M., DUNN, M. J., EMMERSON, L., ERIKSTAD, K. E., FORT, J., FRASER, W., GENOVART, M., GILG, O., GONZÁLEZ-SOLÍS, J., GRANADEIRO, J. P., GRÉMILLET, D., HANSEN, J., HANSEN, S. A., HARRIS, M., HEDD, A., HINKE, J., IGUAL, J. M., JAHNCKE, J., JONES, I., KAPPES, P. J., LANG, J., LANGSET, M., LESCROËL, A., LORENTSEN, S.-H., LYVER, P. O., MALLORY, M., MOE, B., MONTEVECCHI, W. A., MONTICELLI, D., MOSTELLO, C., NEWELL, M., NICHOLSON, L., NISBET, I., OLSSON, O., ORO, D., PATTISON, V., POISBLEAU, M., PYK, T., QUINTANA, F., RAMOS, J. A., RAMOS, R., REIERTSEN, T. K., RODRÍGUEZ, C., RYAN, P., SANZ-AGUILAR, A., SCHMIDT, N. M., SHANNON, P., SITTler, B., SOUTHWELL, C., SURMAN, C., SVAGELJ, W. S., TRIVELPIECE, W., WARZYBOK, P., WATANUKI, Y., WEIMERSKIRCH, H., WILSON, P. R., WOOD, A. G., PHILLIMORE, A. B. & LEWIS, S. 2018. Global phenological insensitivity to shifting ocean temperatures among seabirds. *Nature Climate Change* 8:313–318.
- KLEINSCHMIDT, B., BURGER, C., DORSCH, M., NEHLS, G., HEINÄNEN, S., MORKUNAS, J., ŽYDELIS, R., MOORHOUSE-GANN, R. J., HIPPERSON, H., SYMONDSON, W. O. C. & QUILLFELDT, P. 2019. The diet of red-throated divers (*Gavia stellata*) overwintering in the German Bight (North Sea) analysed using molecular diagnostics. *Marine Biology* 166:77.
- LINDGREN, F., RUE, H. & LINDSTRÖM, J. 2011. An explicit link between Gaussian fields and Gaussian Markov random fields: the stochastic partial differential equation approach: Link between Gaussian Fields and Gaussian Markov Random Fields. *Journal of the Royal Statistical Society: Series B (Statistical Methodology)* 73:423–498.
- LINDLEY, D. V. 2006. Understanding uncertainty. Wiley, Hoboken, N.J. 250 pp.
- MARTINO, S. & RUE, H. 2008. Implementing Approximate Bayesian inference using Integrated Nested Laplace Approximation: A manual for the INLA program. Technical Report, Norwegian University of Science and Technology, Trondheim (NOR).
- MENDEL, B., PESCHKO, V., KUBETZKI, U., WEIEL, S. & GARTHE, S. 2018. Untersuchungen zu möglichen Auswirkungen der Offshore-Windparks im Windcluster nördlich von Helgoland auf Seevögel und Meeressäuger (HELBIRD). P. 166. Schlussbericht, Forschungs- und Technologiezentrum Westküste, Christian-Albrechts-Universität zu Kiel, Büsum (DEU).
- MENDEL, B., SCHWEMMER, P., PESCHKO, V., MÜLLER, S., SCHWEMMER, H., MERCKER, M. & GARTHE, S. 2019. Operational offshore wind farms and associated ship traffic cause profound changes in distribution patterns of Loons (*Gavia spp.*). *Journal of Environmental Management* 231:429–438.
- MENDEL, B., SONNTAG, N., WAHL, J., SCHWEMMER, P., DRIES, H., GUSE, N., MÜLLER, S. & GARTHE, S. 2008. Artensteckbriefe von See- und Wasservögeln der deutschen Nord- und Ostsee: Verbreitung, Ökologie und Empfindlichkeiten gegenüber Eingriffen in ihrem marinen Lebensraum. Bundesamt für Naturschutz, Bonn-Bad Godesberg (DEU). 436 pp.
- NEHLS, G., BURGER, C., KLEINSCHMIDT, B., QUILLFELDT, P., HEINÄNEN, S., MORKUNAS, J. & ŽYDELIS, R. 2018. From effects to impacts: Analysing displacement of Red-throated Divers in relation to their wintering home ranges. In: LPO (2018). Actes du Séminaire Eolien et Biodiversité, Artigues-près-Bordeaux, 21 et 22 novembre 2017. Pp. 113–115 *Actes du Séminaire Eolien et Biodiversité, Artigues-près-Bordeaux, 21 et 22 novembre 2017*.
- NEWTON, I. 2004. Population limitation in migrants - Review. *Ibis* 146:197– 226.

- NOER, H., CHRISTENSEN, T. K., CLAUSAGER, I. & PETERSEN, I. K. 2000. Effects on birds of an offshore wind park at Horns Rev: Environmental impact assessment. P. 112. Technical Report, Department of Coastal Zone Ecology, Denmark.
- PERCIVAL, S. 2010. Kentish Flats offshore wind farm: diver surveys 2009-10. P. 31. On Behalf of Vattenfall Wind Power, Ecology Consulting, Old Cassop, Durham.
- PERCIVAL, S. 2014. Kentish Flats Offshore Wind Farm: Diver Surveys 2011-12 and 2012-13. P. 25. Vattenfall Ecology Consulting, Durham (UK).
- PETERSEN, CHRISTENSEN, K. C., KAHLERT, J., DESHOLM, M. & FOX, A. D. 2006. Final results of bird studies at the offshore wind farms at Nysted and Horns Rev, Denmark. P. 161. National Environmental Research Institute (NERI), Aarhus University, Aarhus (DNK).
- PETERSEN, I. K. & FOX, A. D. 2007. Changes in bird habitat utilisation around the Horns Rev 1 offshore wind farm, with particular emphasis on Common Scoter. P. 36. National Environmental Research Institute (NERI), University of Aarhus, Aarhus (DNK).
- PETERSEN, I. K., NIELSEN, R. D. & MACKENZIE, M. L. 2014. Post-construction evaluation of bird abundances and distributions in the Horns Rev 2 offshore wind farm area, 2011 and 2012. P. 54. Aarhus University, DCE – Danish Centre for Environment and Energy, ST. Andrews (UK).
- PETTIT, L. I. 1990. The Conditional Predictive Ordinate for the Normal Distribution. *Journal of the Royal Statistical Society. Series B (Methodological)* 52:175–184.
- R CORE TEAM. 2019. A Language and Environment for Statistical Computing. R Foundation for Statistical Computing.
- ROBERT, C. P. & CASELLA, G. 2010. Monte Carlo statistical methods (2. ed., softcover reprint of the hardcover 2. ed. 2004). Springer, New York, NY. 645 pp.
- RUE, H. & HELD, L. 2005. Gaussian Markov random fields: theory and applications. Chapman & Hall/CRC, Boca Raton. 263 pp.
- RUE, H., MARTINO, S. & CHOPIN, N. 2009. Approximate Bayesian inference for latent Gaussian models by using integrated nested Laplace approximations. *Journal of the Royal Statistical Society: Series B (Statistical Methodology)* 71:319–392.
- SÆTHER, B.-E. & BAKKE, Ø. 2000. Avian Life History Variation and Contribution of Demographic Traits to the Population Growth Rate. *Ecology* 88:642–653.
- SCHWEMMER, H., MARKONES, N., MÜLLER, S., BORKENHAGEN, K., MERCKER, M. & GARTHE, S. 2019. Aktuelle Bestandsgröße und -entwicklung des Sterntauchers (*Gavia stellata*) in der deutschen Nordsee. Bericht für das Bundesamt für Seeschifffahrt und Hydrographie und das Bundesamt für Naturschutz. FTZ.
- SCHWEMMER, P., MENDEL, B., DIERSCHKE, V. & GARTHE, S. 2011. Effects of ship traffic on seabirds in offshore waters: Implications for marine conservation and spatial planning. *Ecological Applications* 21:1851–1860.
- SIM, I. M. W., REBECCA, G. W., LUDWIG, C., GRANT, M. J. & REID, J. M. 2011. Characterizing demographic variation and contributions to population growth rate in a declining population. *Journal of Animal Ecology* 80:159–170.
- SKOV, H. & PRINS, E. 2001. Impact of estuarine fronts on the dispersal of piscivorous birds in the German Bight. *Marine Ecology Progress Series* 214:279–287.
- SPIEGELHALTER, D. J., BEST, N. G., CARLIN, B. P. & VAN DER LINDE, A. 2002. Bayesian measures of model complexity and fit. *Journal of the Royal Statistical Society: Series B (Statistical Methodology)* 64:583–639.
- SZOSTEK, K. L. & BECKER, P. H. 2015. Survival and local recruitment are driven by environmental carry-over effects from the wintering area in a migratory seabird. *Oecologia* 178:643–657.
- UNEP / AEWA SECRETARIAT. 2016. Agreement on the conservation of African-Eurasian Migratory Waterbirds (AEWA). Text and Annexes (as amended by MOP6). Pp. 1–59. Bonn.
- VISSER, M. E. & BOTH, C. 2005. Shifts in phenology due to global climate change: the need for a yardstick. *Proceedings of the Royal Society B: Biological Sciences* 272:2561–2569.

- WATANABE, S. 2010. Asymptotic Equivalence of Bayes Cross Validation and Widely Applicable Information Criterion in Singular Learning Theory. *The Journal of Machine Learning Research* 11:3571–3594.
- WEBB, A., MACKENZIE, M., CANECO, B. & DONOVAN, C. 2015. Lincs Wind Farm - Second annual post-construction aerial ornithological monitoring report. HiDef Aerial Surveying Limited.
- WELCKER, J. & NEHLS, G. 2016. Displacement of seabirds by an offshore wind farm in the North Sea. *Marine Ecology Progress Series* 554:173–182.
- ZYDELIS, R., DORSCH, M., HEINÄNEN, S., NEHLS, G. & WEISS, F. 2019. Comparison of digital video surveys with visual aerial surveys for bird monitoring at sea. *Journal of Ornithology*.

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A APPENDIX

A.1 Supplementary material

A.1.1 Visual aerial surveys - Distance models

Table -1 Overview.

Source	years	N divers observed
IBL	2004-2010	706
IfAÖ GmbH	2003-2018	3186
BioConsult SH	2001-2010 (period 1)	3519
	2010-2014 (period 2)	2167
FTZ Westküste	2002-2016	2378

Table A-2 Distance models for visual aerial surveys conducted by IBL.

Model	AIC	ΔAIC
Hazard rate		
Sea state+ group size	593.8	0
Group size	594.7	0.9
Sea state+ group size+project	595.3	1.5
Sea state	598.0	4.2
–	599.6	5.8
Project	601.2	7.4
Glare	602.0	8.2
Sight conditions	602.5	8.7
Half-normal		
–	623.1	29.2

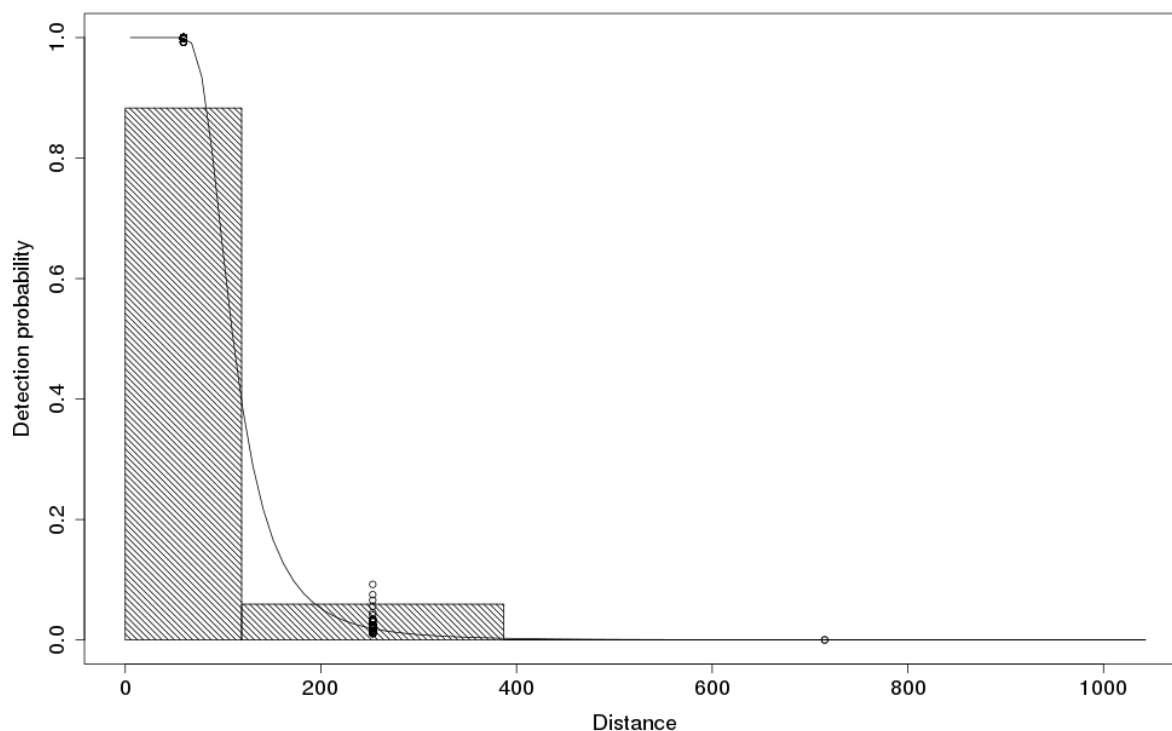


Figure A-1 Distance-dependent detection probability from the top-ranking model for visual aerial surveys conducted by IBL.

Table A-3 Estimates from the top-ranking model for visual aerial surveys conducted by IBL.

Parameter	estimate	se
Intercept	4,724	0,122
log group size	0,169	0,074

Table A-4 Distance models for visual aerial surveys conducted by IfAÖ.

Model	AIC	ΔAIC
Hazard rate		
Project + year + group size	5095,5	0
Year + group size	5100,3	4,8
Project + group size	5172,7	77,2
Group size	5176,1	80,6
Sight conditions	5190,3	94,8
Project	5193,6	98,1
–	5195,6	100,1
Glare	5198,6	103,1
Half-normal		
–	5274,8	179,3

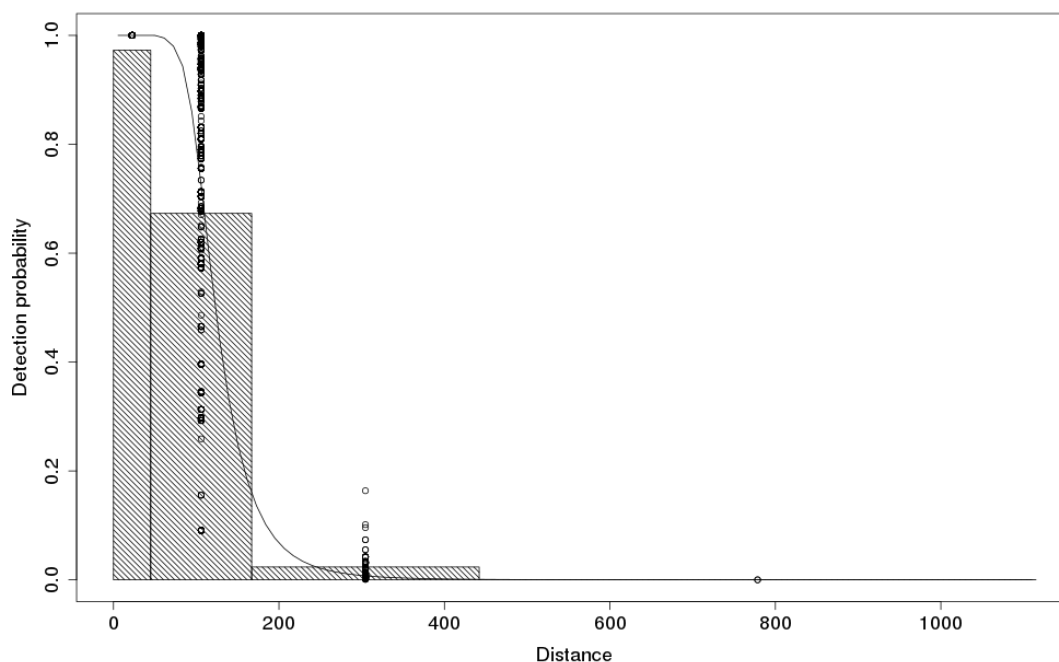


Figure A-2 Distance-dependent detection probability from the top-ranking model for visual aerial surveys conducted by IfAÖ.

Table A-5 Estimates from the top-ranking model for visual aerial surveys conducted by IfAÖ.

Parameter	estimate	se
Intercept	5,417	0,108
year 2004	-0,508	0,110
year 2005	-0,490	0,155
year 2008	-0,759	0,139
year 2009	-0,681	0,133
year 2010	-0,911	0,113
year 2011	-0,672	0,103
year 2012	-0,661	0,109
year 2013	-0,442	0,104
year 2015	-0,294	0,125
year 2016	-0,141	0,122
year 2017	-0,218	0,123
year 2018	-0,054	0,102
project 2	0,058	0,103
project 3	-0,056	0,054
project 4	-0,248	0,142
project 5	-0,274	0,114
project 6	-0,222	0,140
project 7	0,033	0,087

Parameter	estimate	se
project 8	-0,271	0,067
log size	0,151	0,033

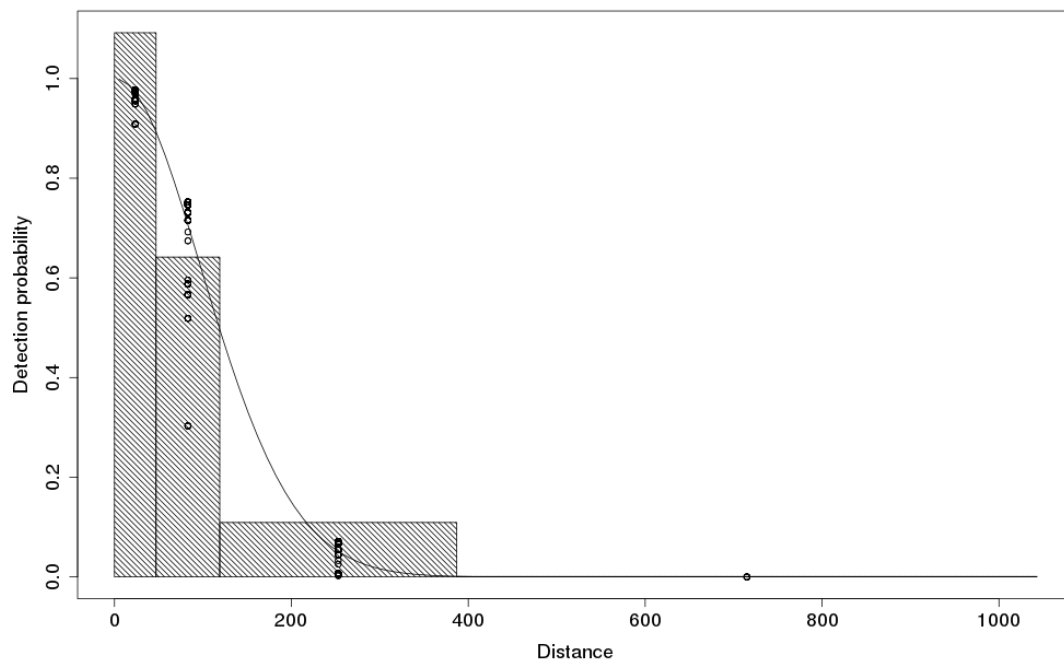
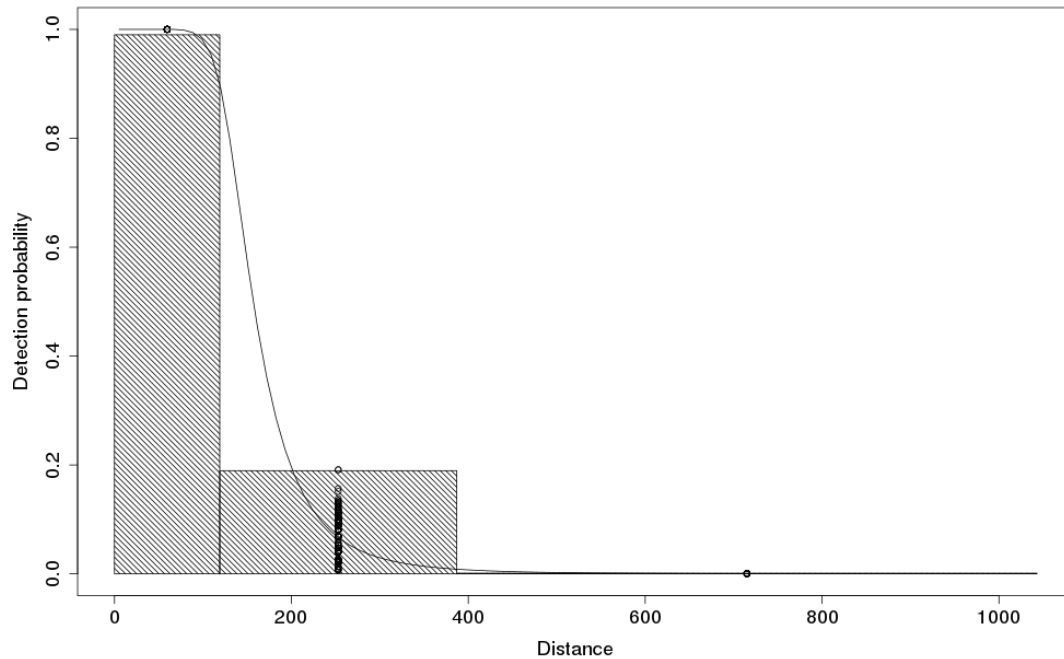


Figure A-3 Distance-dependent detection probability from the top-ranking models for visual aerial surveys conducted by BioConsult SH.

Table A-6 Distance models for visual aerial surveys conducted by BioConsult SH.

Model	AIC	ΔAIC
period 1		
Hazard rate		
Project + group size + sight conditions	4467,1	
Project + group size + sea state + sight conditions	4467,6	0,4
Project + sea state + sight conditions	4470,0	2,8
Project	4482,8	15,7
Sight conditions	4485,5	18,4
Sea state + group size	4495,1	28,0
Sea state	4498,2	31,0
Group size	4500,1	33,0
–	4505,2	38,0
Half-normal		
–	4529,8	62,6
period 2		
Half-normal		
Project + sight conditions	4803,6	
Sight conditions	4804,5	0,9
Project	4823,6	20,0
Sea state	4824,9	21,3
–	4828,4	24,9
Group size	4830,4	26,9
Hazard rate		
–	4833,5	29,9

Table A-7 Estimates from the top-ranking models for visual aerial surveys conducted by BioConsult SH.

Parameter	estimate	se
period 1		
Intercept	5,218	0,189
project 2	-0,152	0,234
project 3	-0,223	0,188
project 4	-0,470	0,196
project 5	-0,333	0,191
log size	0,050	0,023
sight conditions 2	-0,234	0,058
sight conditions 3	-0,039	0,183
period 2		
Intercept	4,687	0,024
project 2	-0,068	0,111
project 3	-0,405	0,155
project 4	0,013	0,052

Parameter	estimate	se
project 5	-0,034	0,032
sight conditions 2	-0,299	0,058
sight conditions 3	-0,115	0,198

Table A-8 Distance models for visual aerial surveys conducted by FTZ Westküste.

Model	AIC	Δ AIC
Hazard rate		
Visibility + group size	5270,0	
Group size	5271,6	1,6
Visibility	5275,7	5,7
–	5276,9	7,0
Sea state	5278,8	8,9
Half-normal		
–	5334,4	64,4

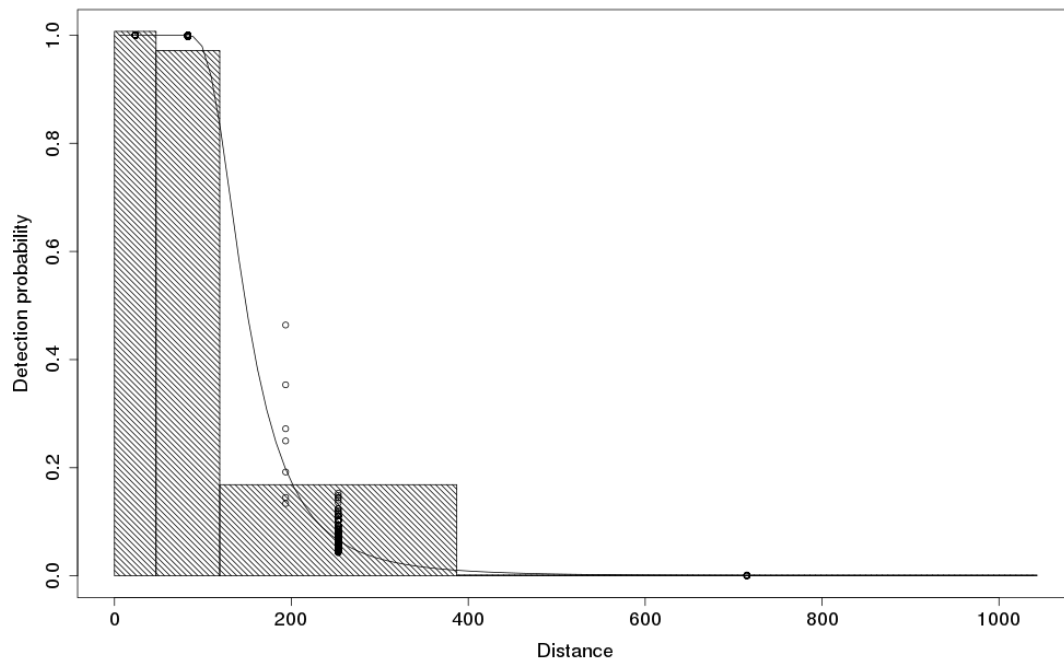


Figure A-4 Distance-dependent detection probability from the top-ranking model for visual aerial surveys conducted by FTZ Westküste.

Table A-9 Estimates from the top-ranking model for visual aerial surveys conducted by FTZ Westküste.

Parameter	estimate	se
Intercept	4,815	0,052
Visibility	0,010	0,005
log size	0,082	0,030

A.1.2 Data

Table -10 Number of surveys per project, for visual surveys

Project with visual surveys	Number of surveys (spring & winter)
Albatros	3
Alpha Ventus	11
Amrumbank West	2
Borkum Riffgrund	4
Butendiek_Basis	8
Butendiek_3.UJ	4
Dan Tysk	6
FTZ Monitoring	15
Global Tech I	6
Gode Wind	6
Horizont	5
North Sea Windpower	3
Northern Energy	5
OWP Bard Offshore 1	6
OWP Bard Offshore 1 und weitere, OWP Notos und weitere	1
OWP Borkum Riffgrund 1	1
OWP Meerwind Süd, OWP Meerwind Ost	4
OWP Nordsee Ost	8
Riffgat	17
Sandentnahme	10
Summe	125

Table -11 Number of surveys per project, for digital surveys

Project with digital surveys	Number of surveys (spring & winter)
ABW	4

Project with digital surveys	Number of surveys (spring & winter)
BARD Deutsche Bucht and Veja Mate	11
BARD Offshore 1	3
Butendiek	25
Cluster Helgoland	15
Cluster Nördlich Borkum	17
Cluster Östlich Austergrund	13
DanTysk, Sandbank	22
DIVER	4
Global Tech I	3
Helgoland Cluster	4
Nördlich Borkum	4
OWP Nordergründe	16
Summe	141

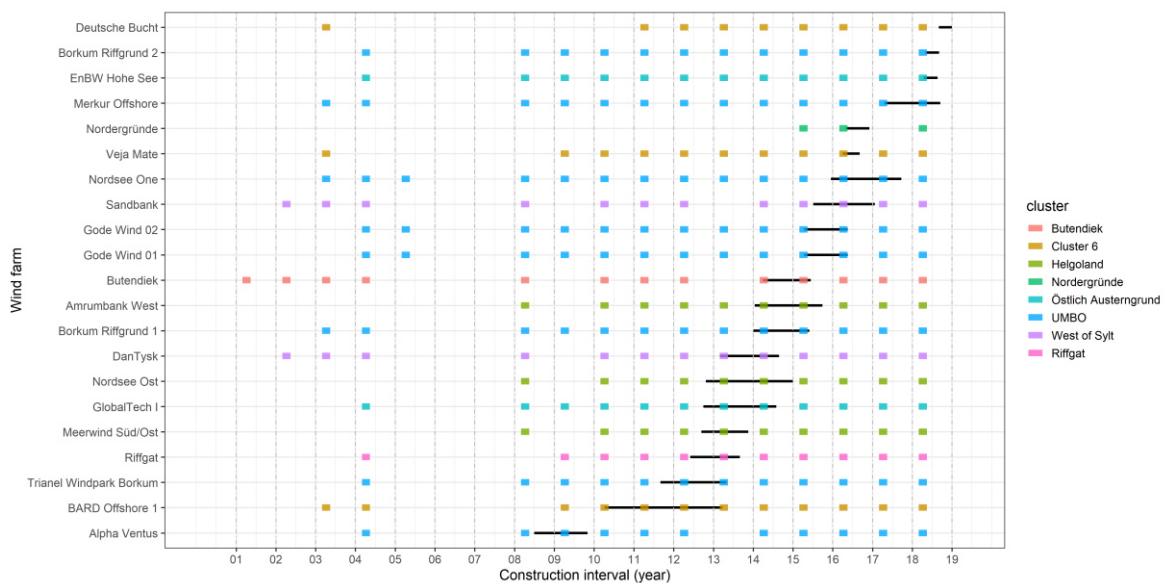


Figure A-5 Years with survey data for wind farm projects within the German North Sea and indication of construction periods. For Spring season.

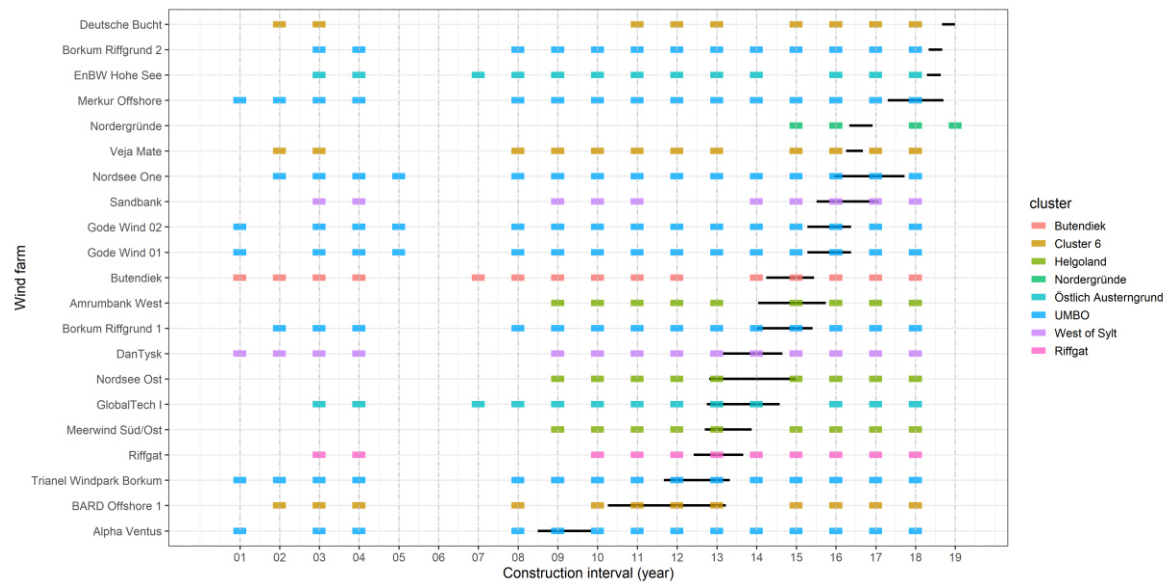


Figure A-6 Years with survey data for wind farm projects within the German North Sea and indication of construction periods. For Winter season.

A.1.3 Results

A.1.4 Observation effort

Observation effort in spring

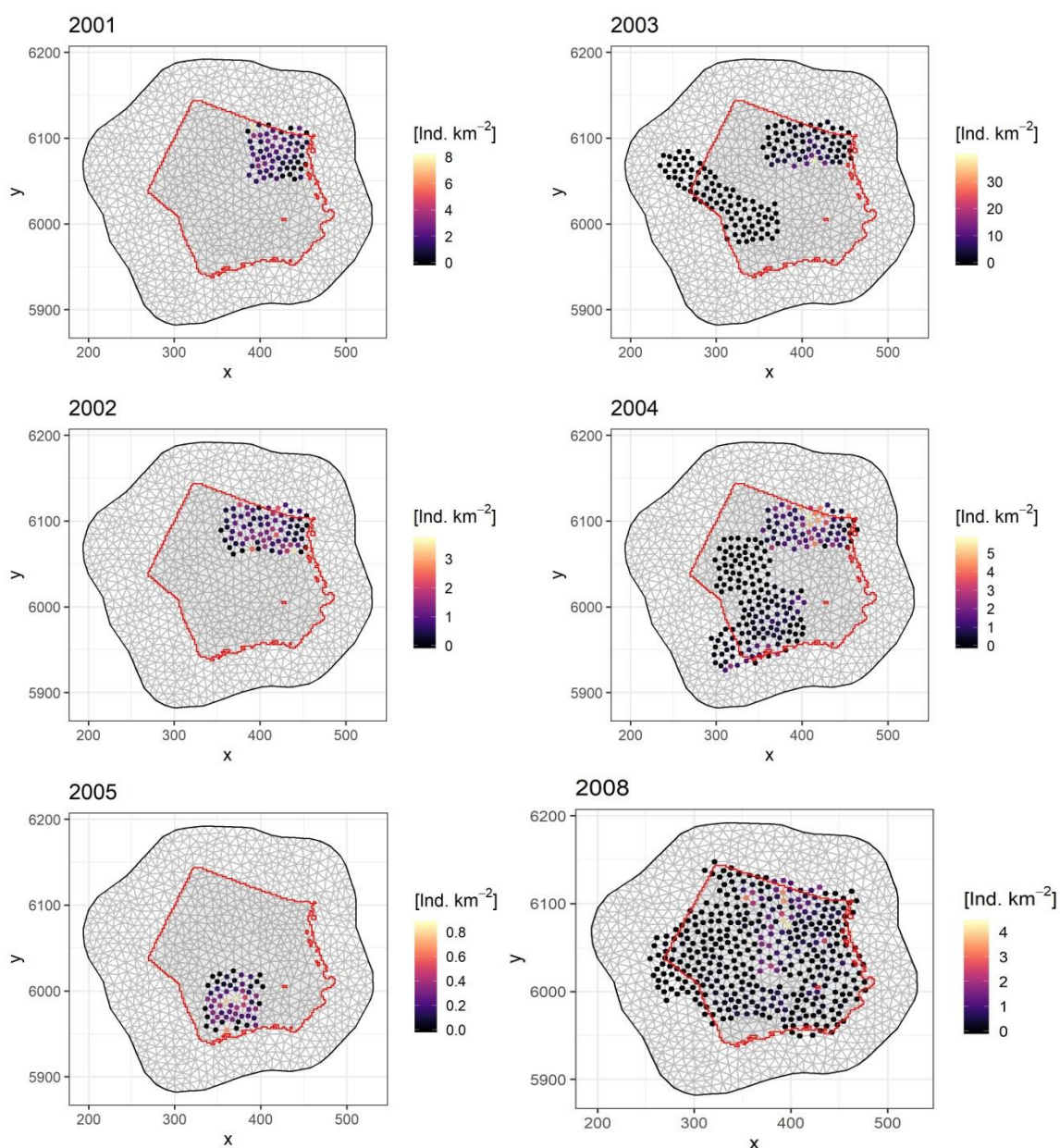


Figure A-7 5-km mesh for spatio-temporal model for spring. Colours mark nodes where data was available and give densities for that location.

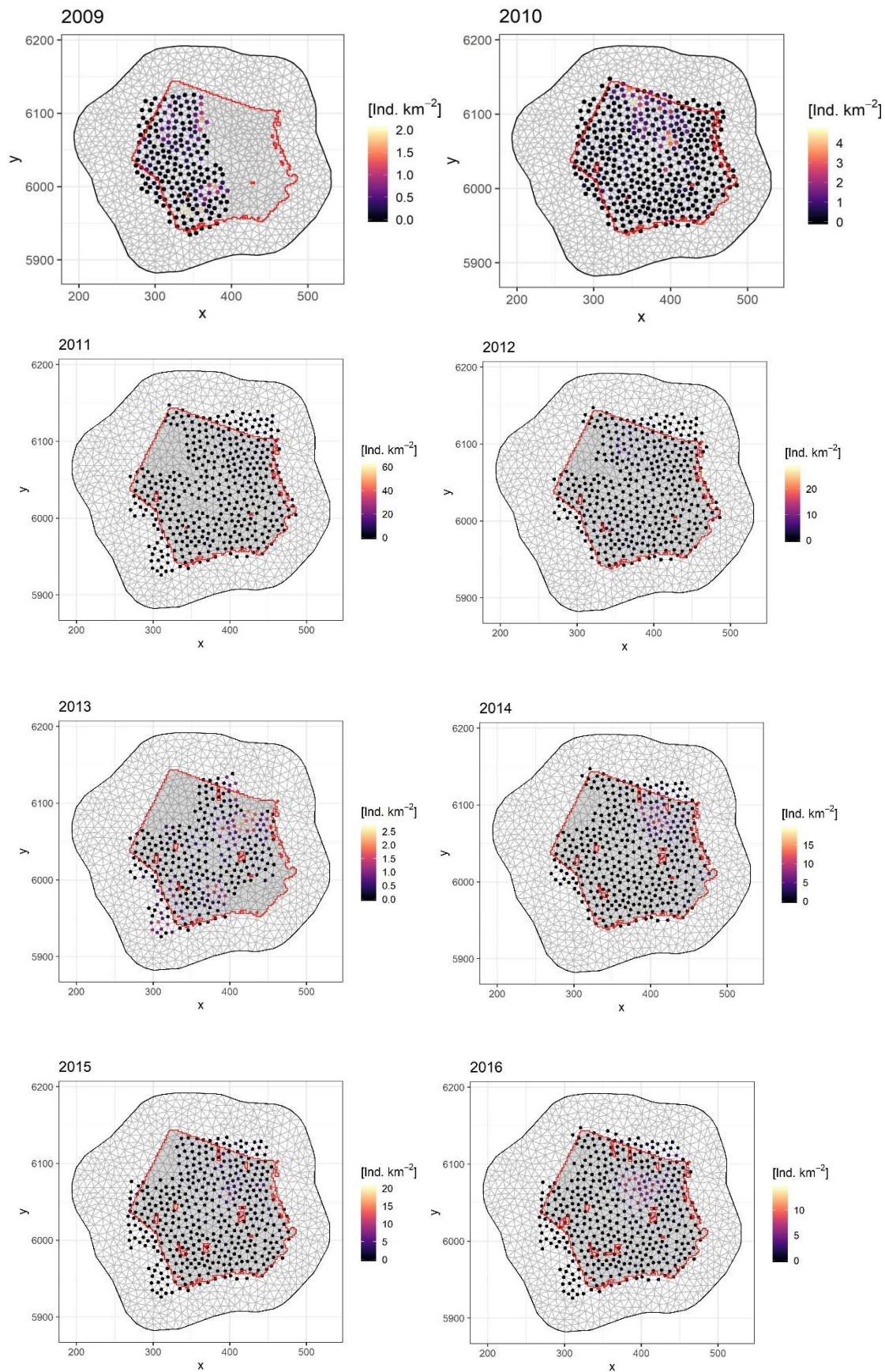


Figure A-8 5-km mesh for spatio-temporal model for spring. Colours mark nodes where data was available and give densities for that location.

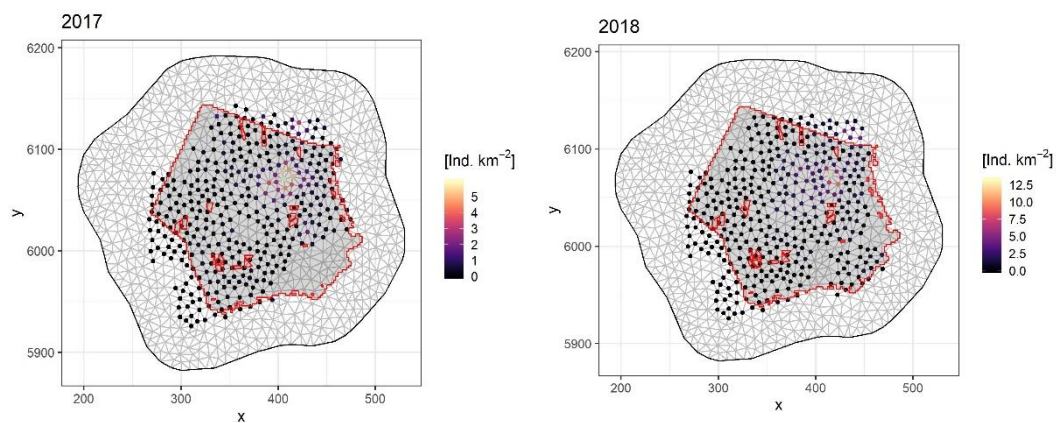


Figure A-9 5-km mesh for spatio-temporal model for spring. Colours mark nodes where data was available and give densities for that location.

Observation effort in winter:

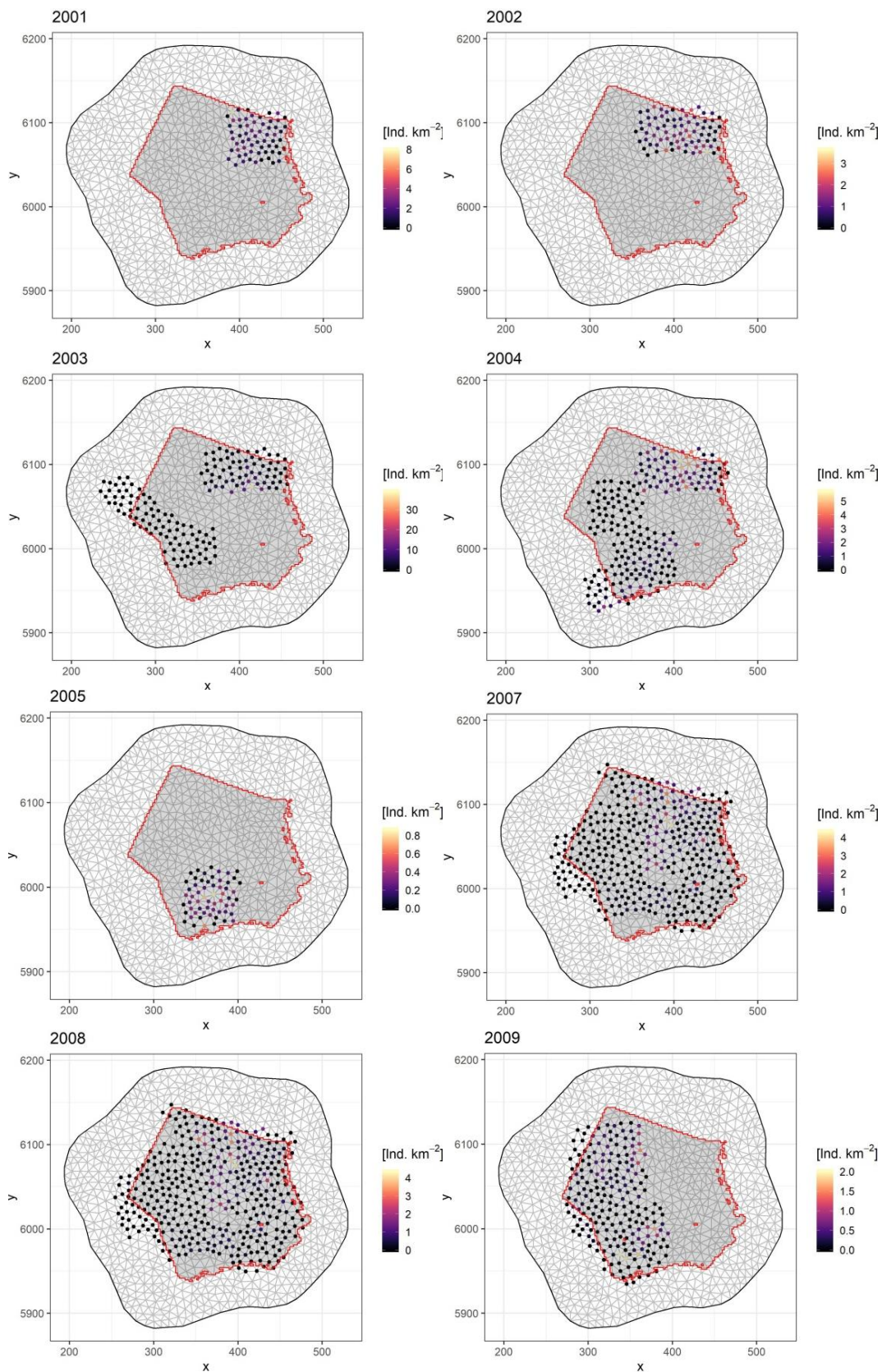


Figure A-10 5-km mesh for spatio-temporal model for winter. Colours mark nodes where data was available and give densities for that location.

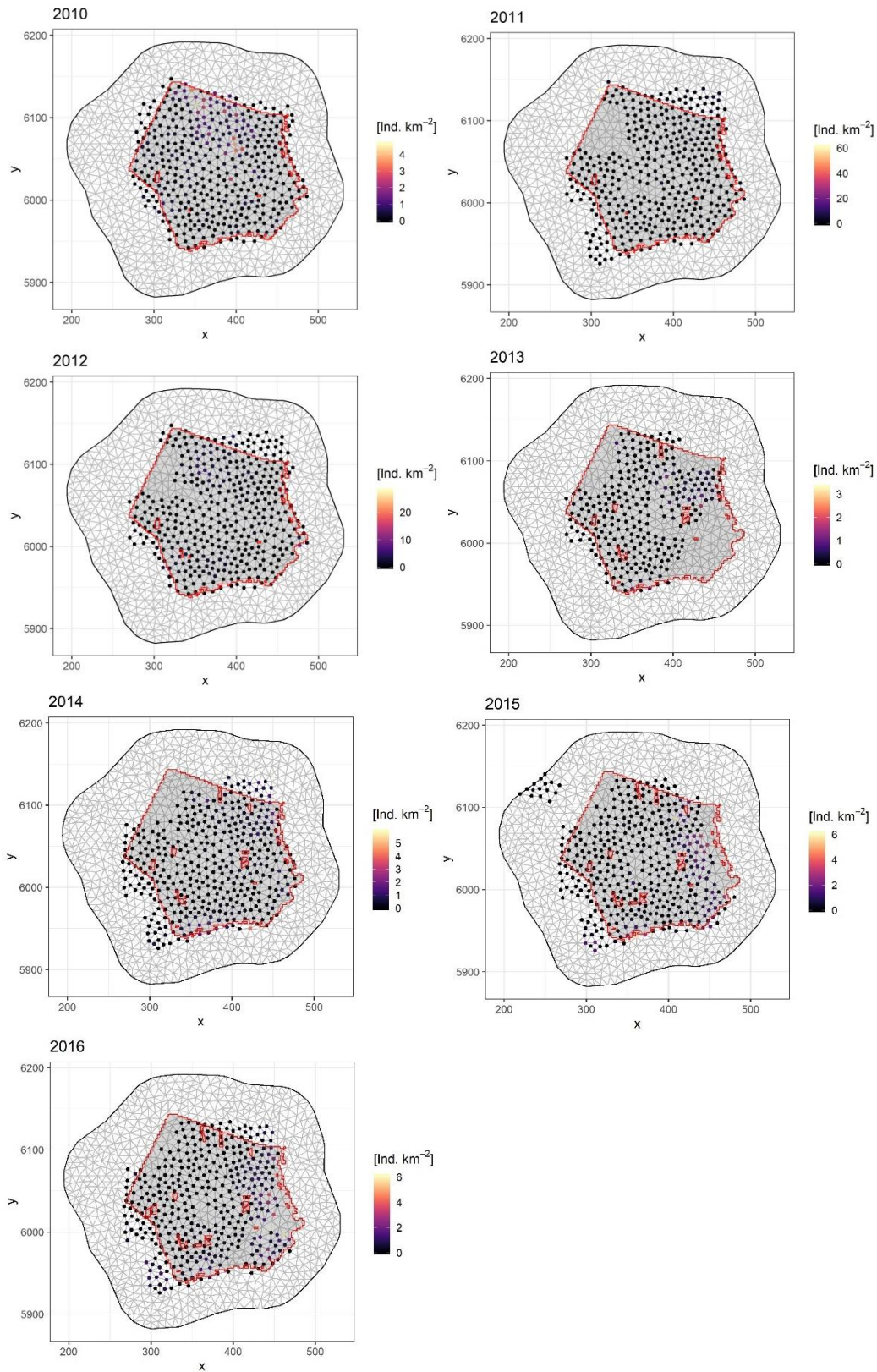


Figure A-11 5-km mesh for spatio-temporal model for winter. Colours mark nodes where data was available and give densities for that location.

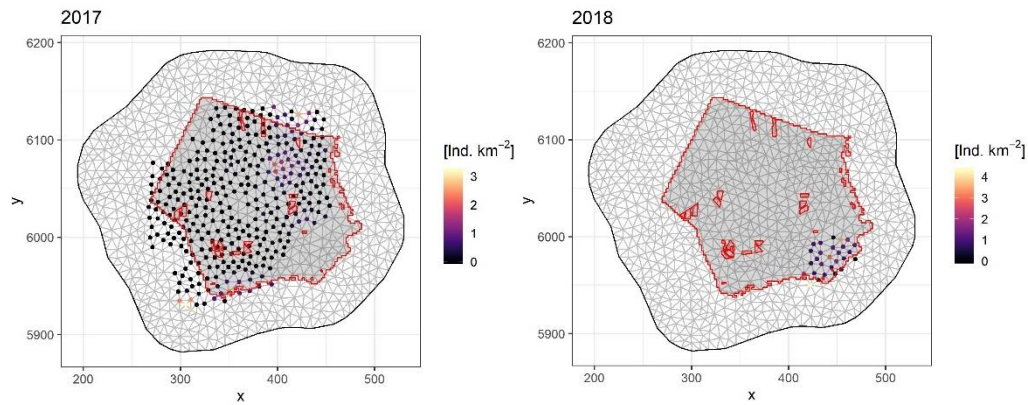
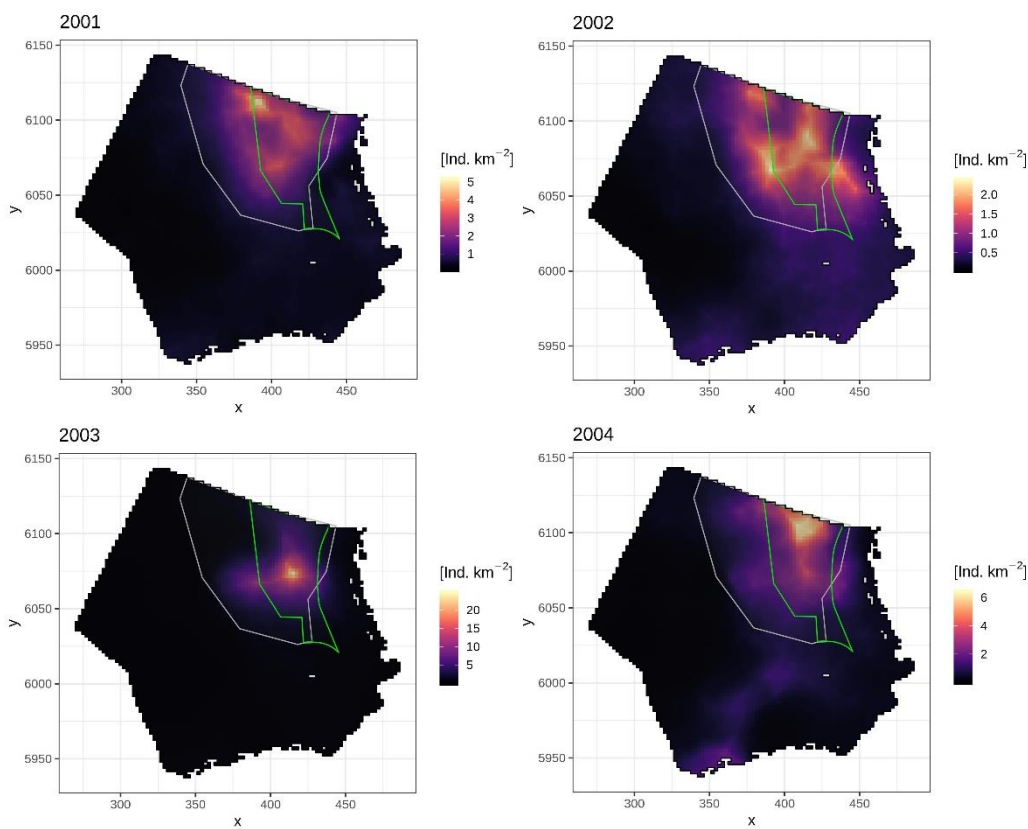
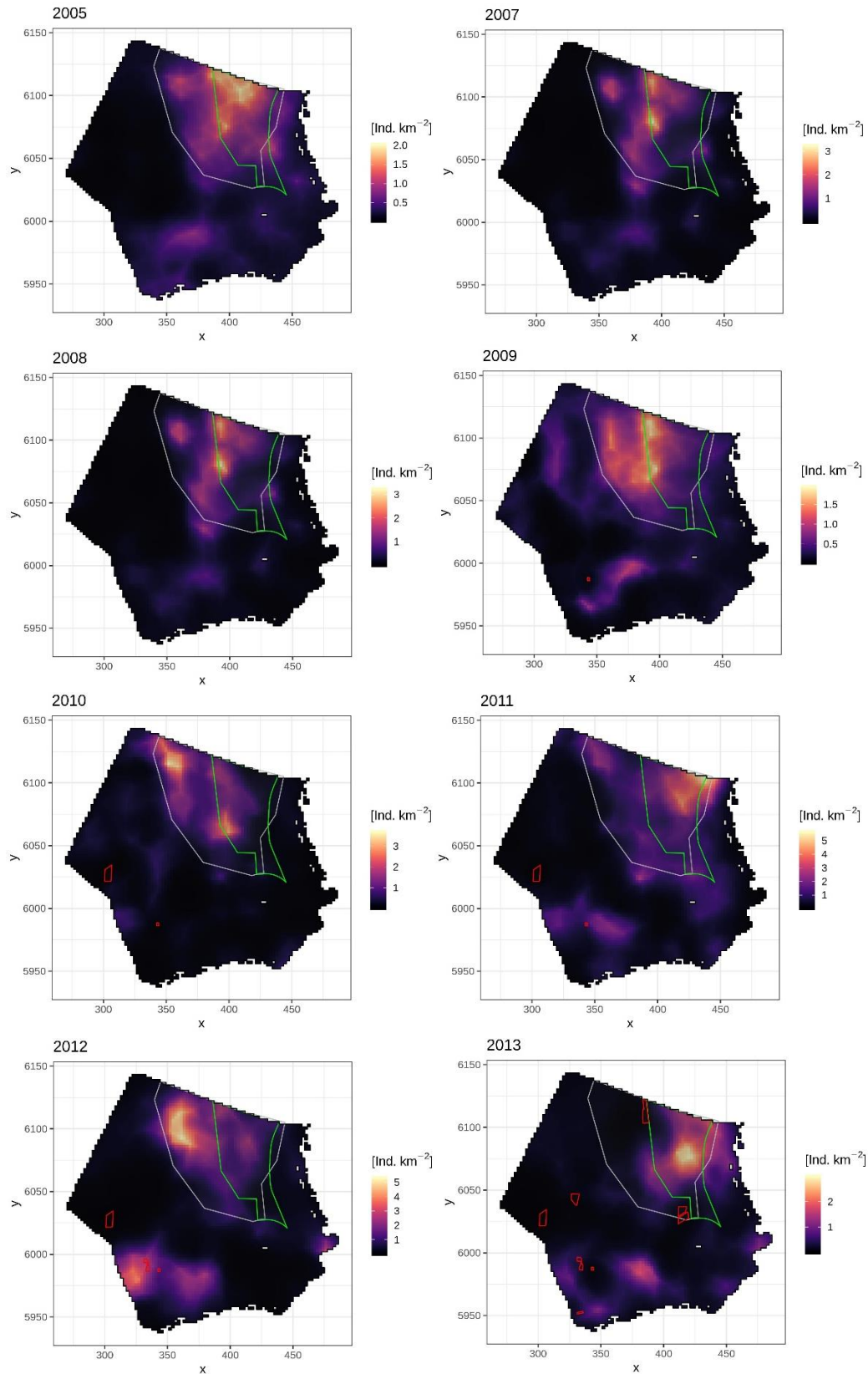


Figure A-12 5-km mesh for spatio-temporal model for winter. Colours mark nodes where data was available and give densities for that location.

A.1.5 Model validation (bathymetry)





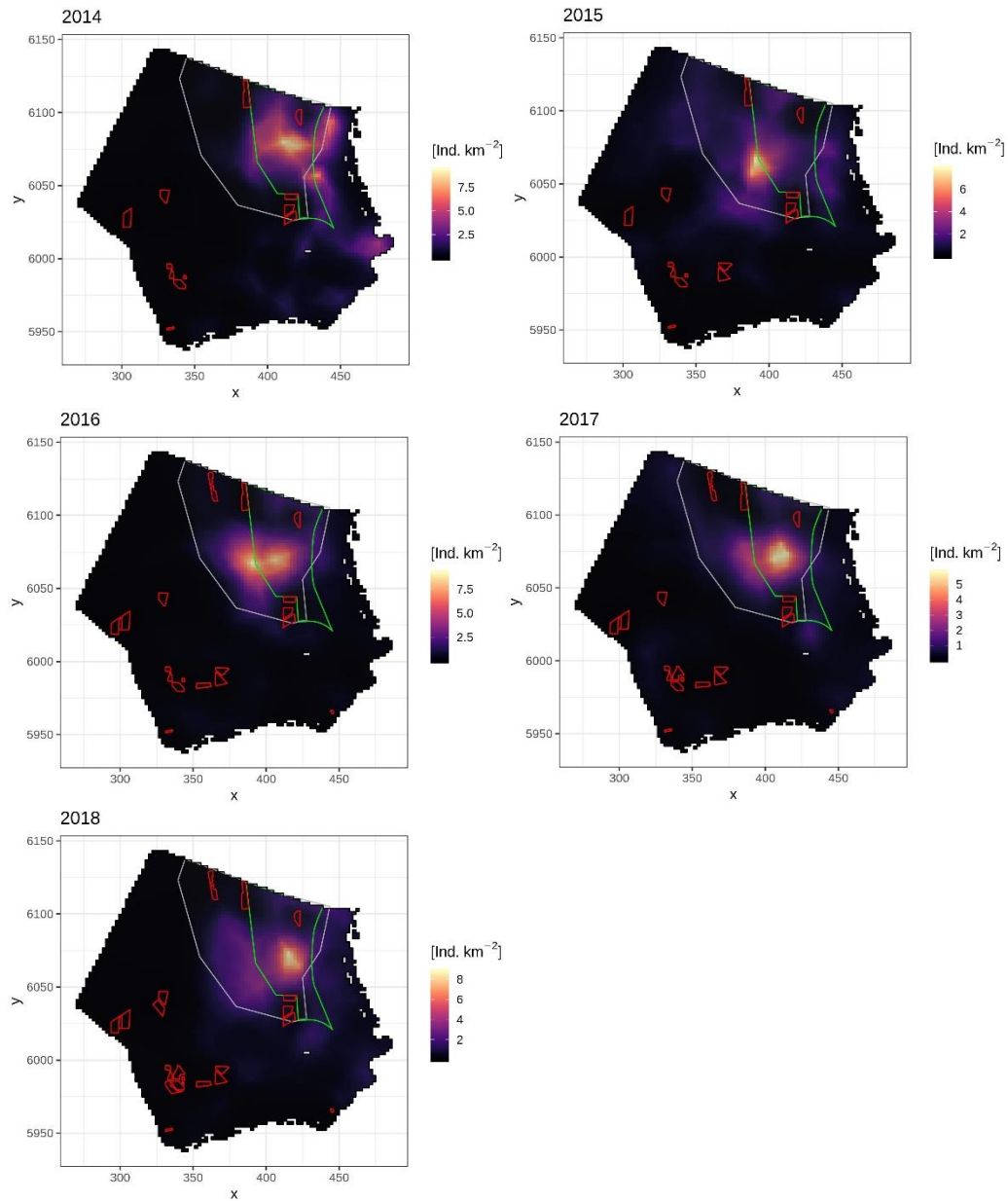
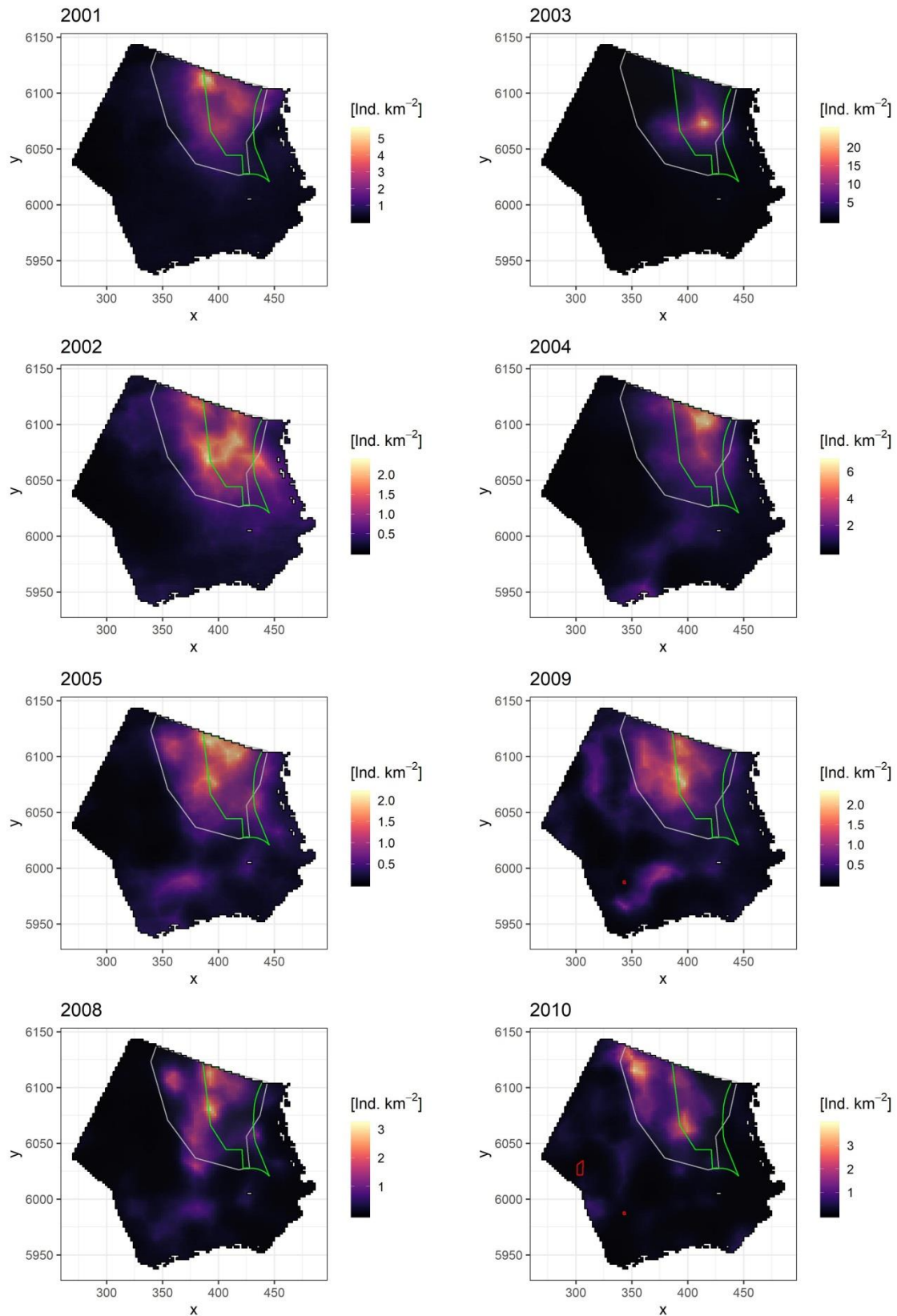


Figure A-13 Predicted densities for spring from model with linear effect of bathymetry. Note varying scales for each phase/year.



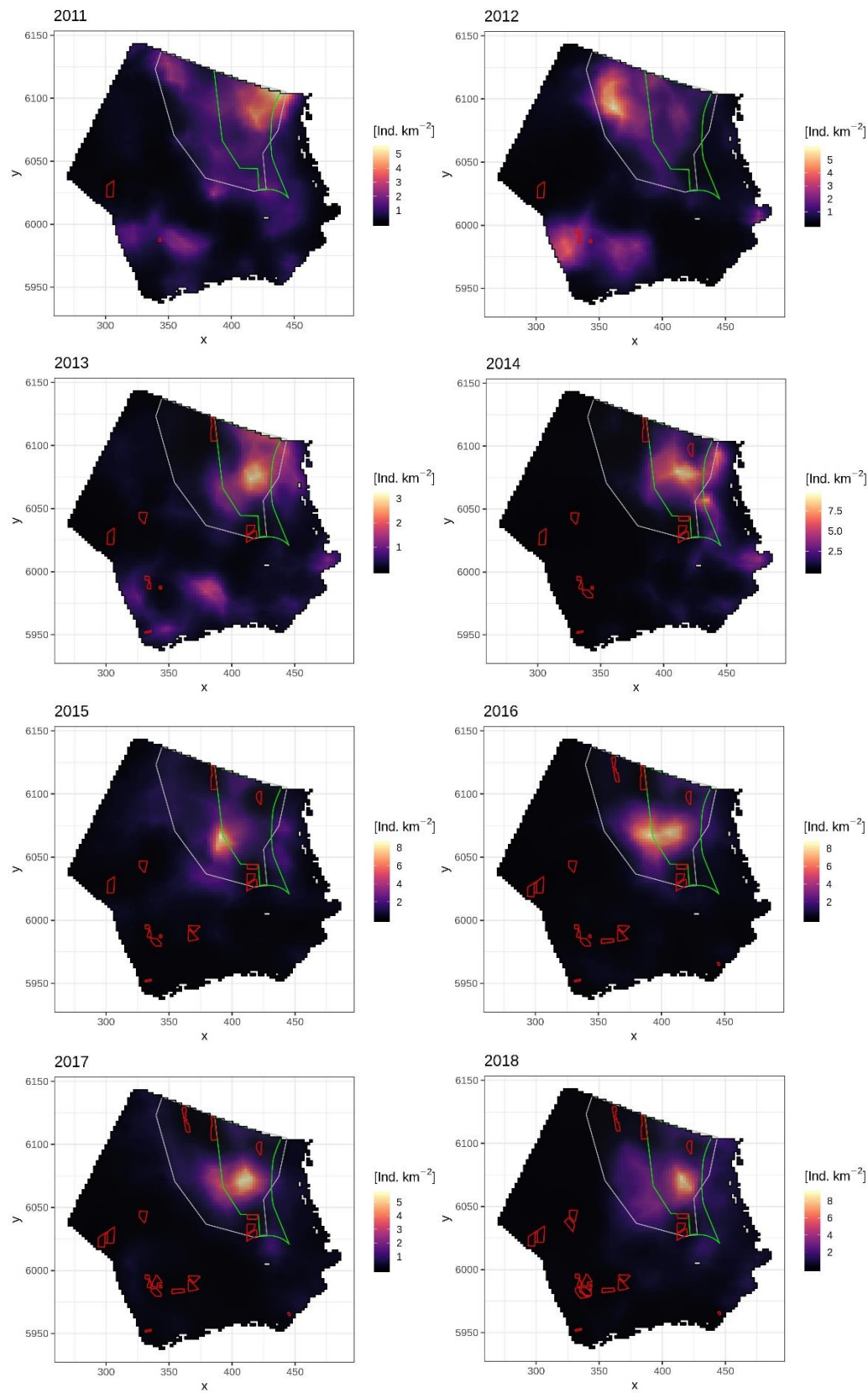
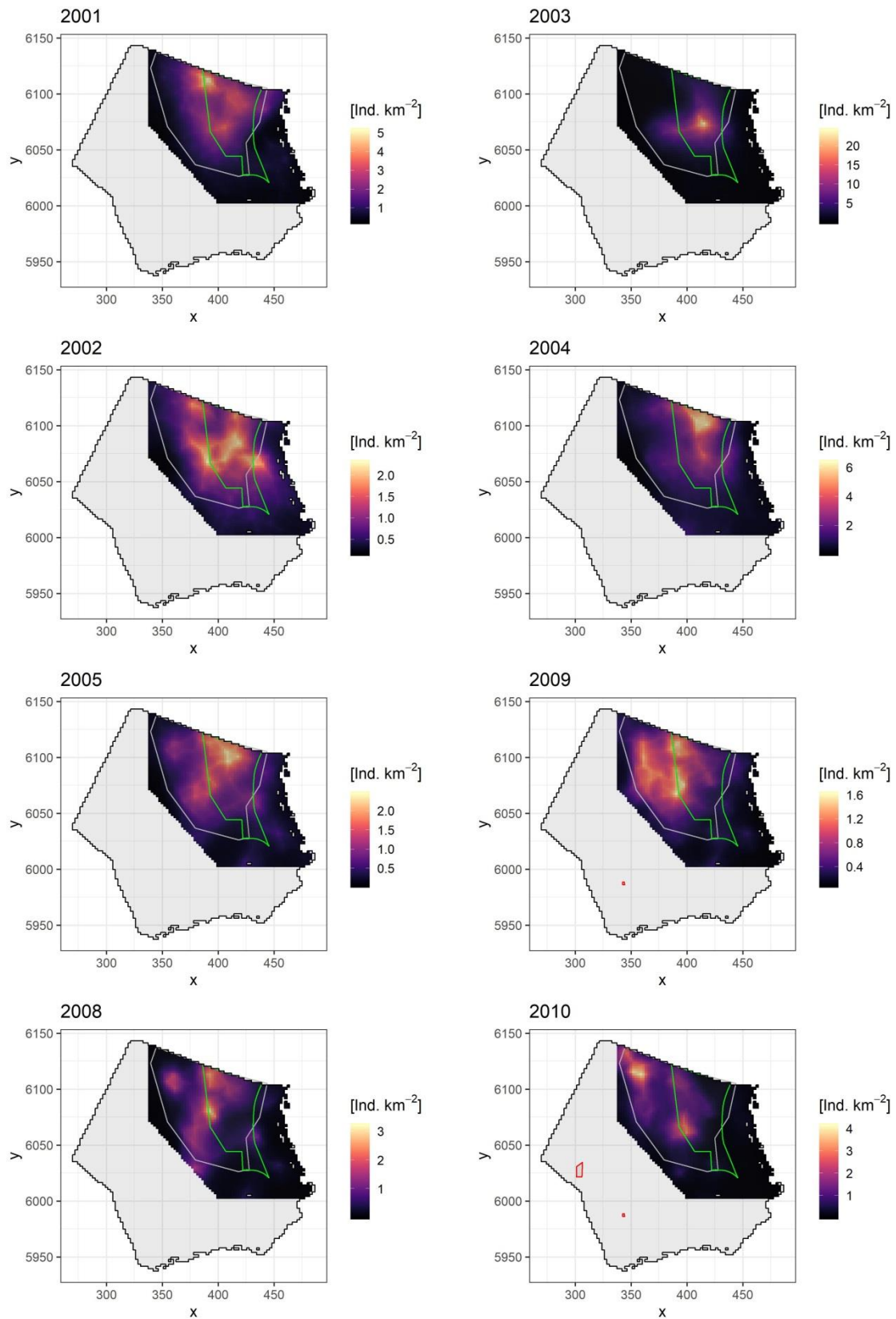


Figure A-14 Predicted densities for spring from model with non-linear effect of bathymetry. Note varying scales for each phase/year.

A.1.6 Results from spatio-temporal model



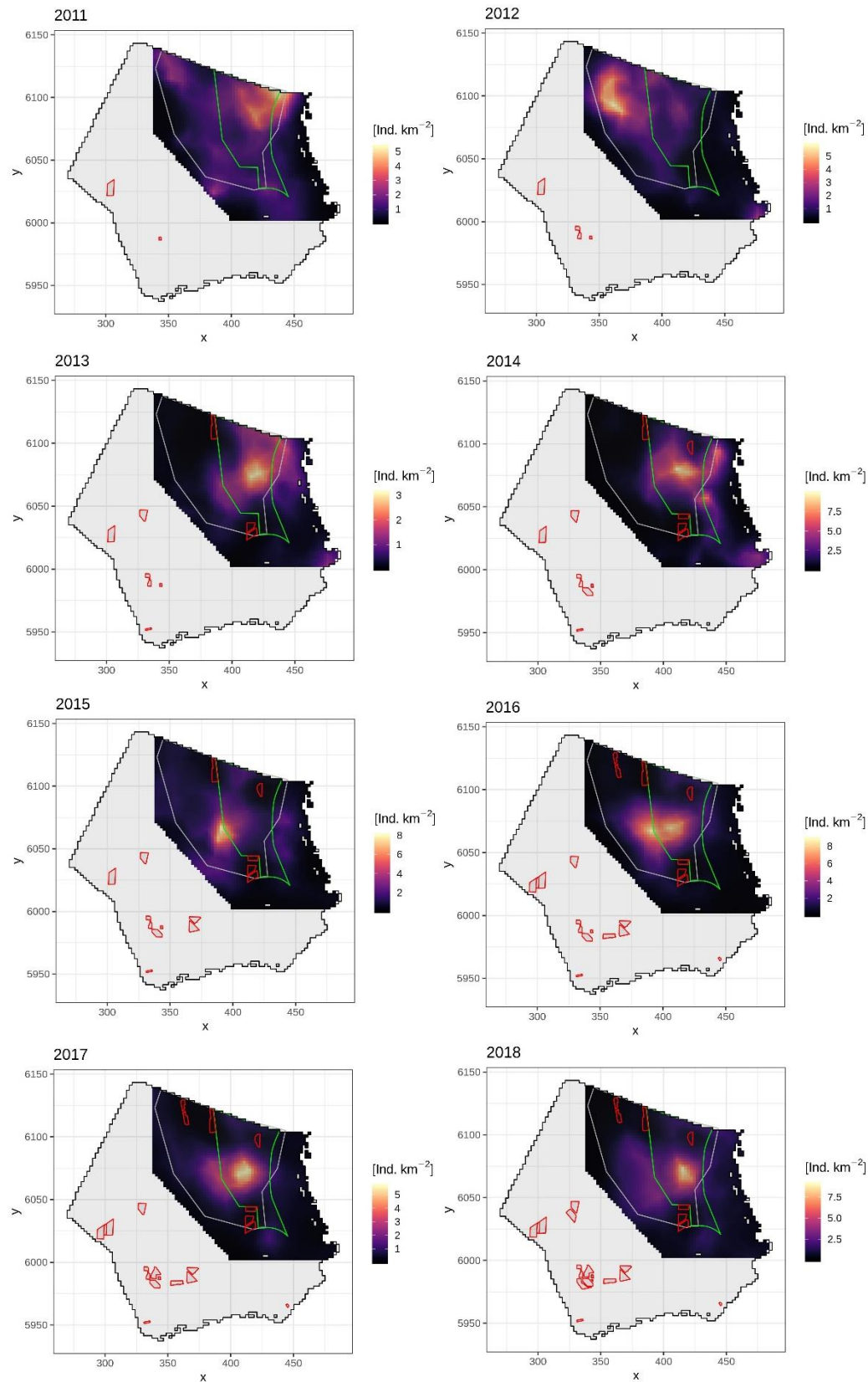
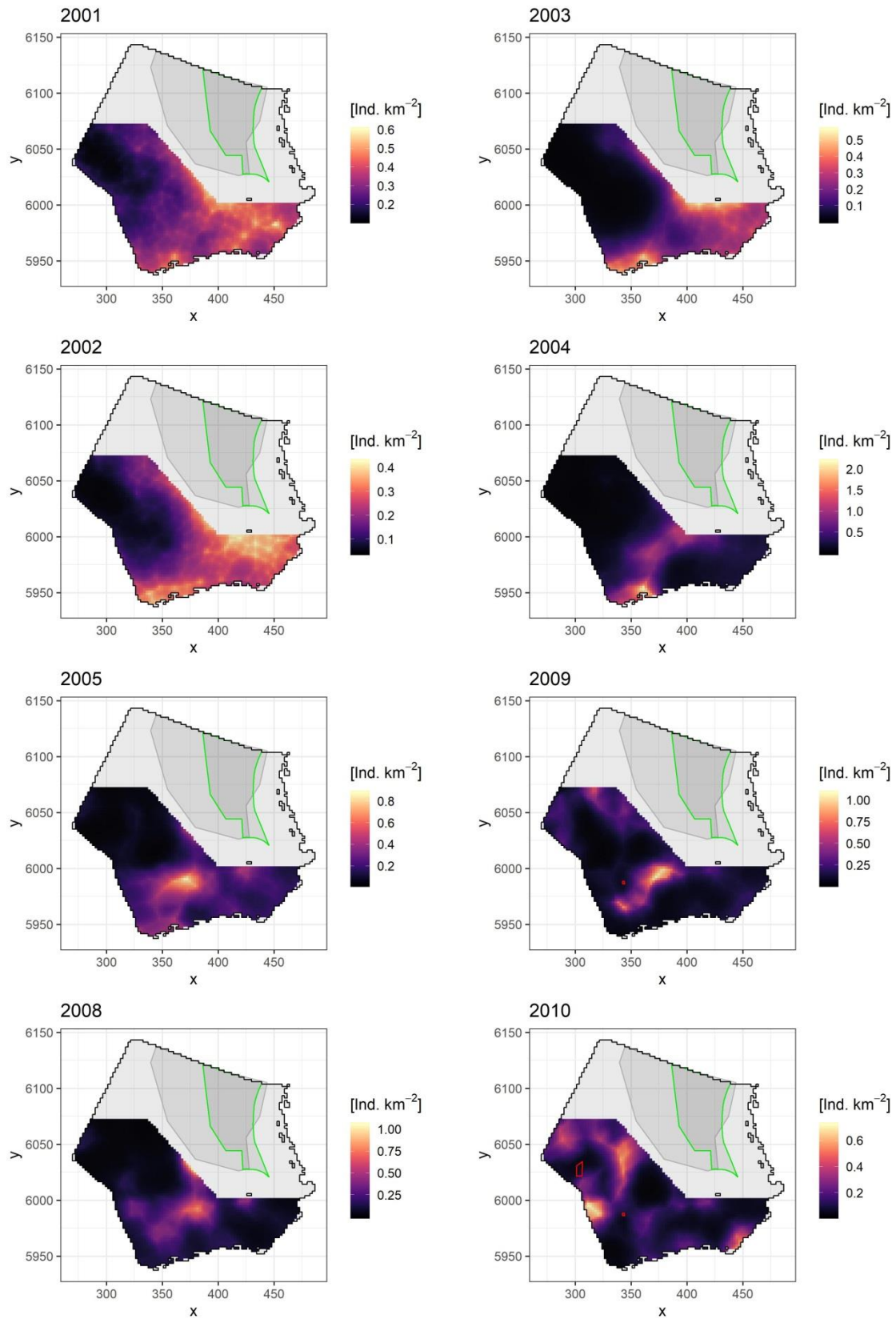


Figure A-15 Predicted densities for spring for the northern study area. Note varying scales for each phase/year. Red borders indicate wind farms under construction or in operation. Green line depicts border of SPA "Eastern German Bight", white line depicts main concentration area for divers as defined by BMU (2009).



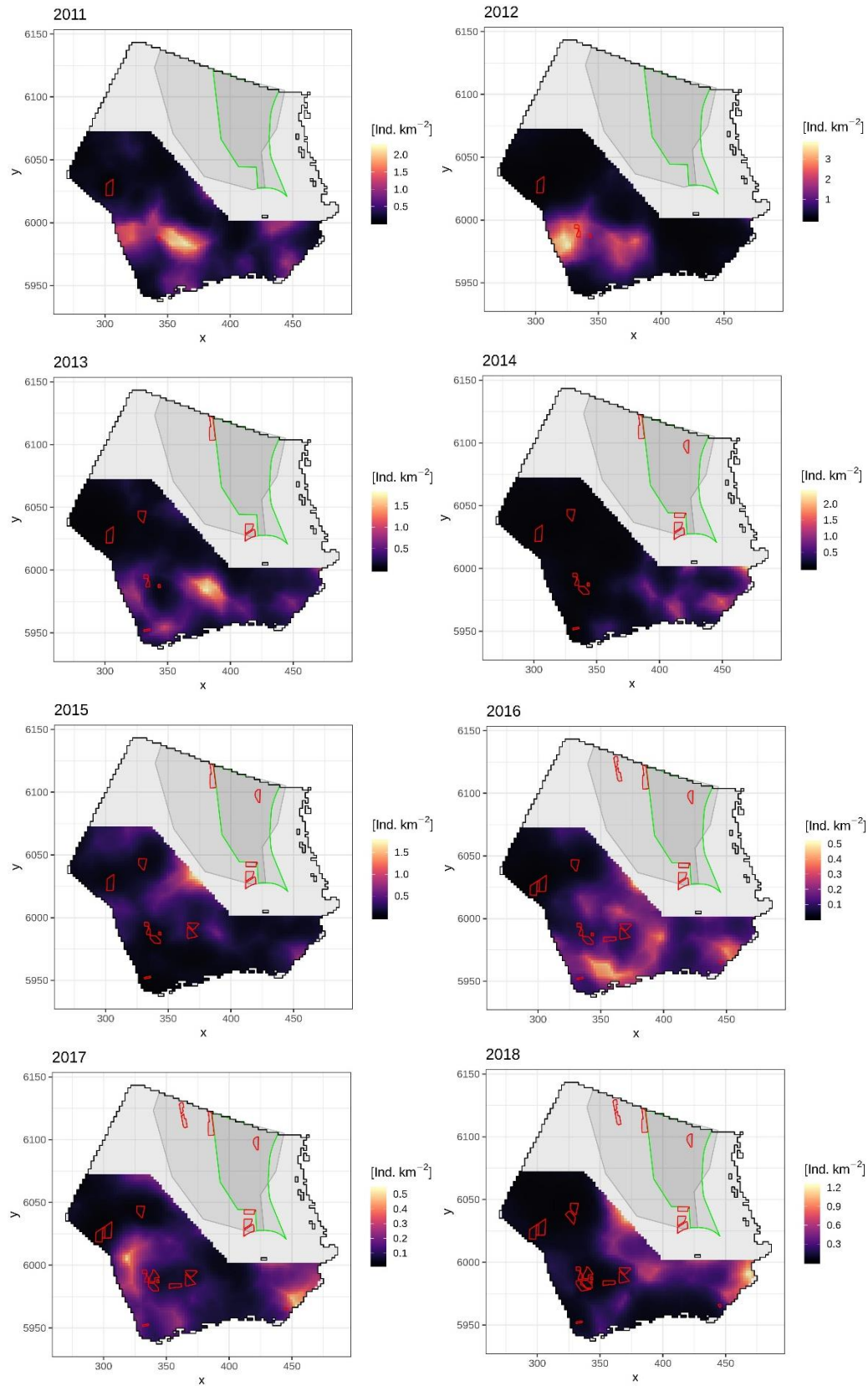


Figure A-16 Predicted densities for spring for the southern study area. Note varying scales for each phase/year. Red borders indicate wind farms under construction or in operation.