Journal of Environmental Management 231 (2019) 429–438

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Research article

Operational offshore wind farms and associated ship traffic cause profound changes in distribution patterns of Loons (*Gavia* spp.)

Bettina Mendel[a], Philipp Schwemmer[a,*], Verena Peschko[a], Sabine Müller[a], Henriette Schwemmer[a], Moritz Mercker[b], Stefan Garthe[a]

[a] Research and Technology Centre (FTZ), University of Kiel, Hafentörn 1, 25761 Büsum, Germany
[b] BIONUM - Büro für Biostatistik, Finkenwerder Norderdeich 15 A, 21129 Hamburg, Germany

ARTICLE INFO

Keywords:
Environmental impact
Marine special protection area
Habitat loss
Management
Red-throated Loon
Avoidance behaviour

ABSTRACT

Seabirds select suitable habitats at sea, but these habitats may be strongly impacted by marine spatial planning, including the construction of offshore wind farms (OWFs) and the associated ship traffic. Loons (*Gavia* spp.) are particularly vulnerable to anthropogenic activities and are also of high conservation status, making them particularly relevant to marine planning processes. We investigated the effects of OWF construction and ship traffic on Loon distributions in the German North Sea on a large spatial scale, using a 'before–after' control impact analysis approach and a long-term data set. Many OWFs were built in or close to core areas of Loon distributions. Loons showed significant shifts in their distribution in the 'after' period and subsequently aggregated between two OWF clusters, indicating the remaining suitable habitat. The decrease in Loon abundance became significant as far as about 16 km from the closest OWF. Ship traffic also had a significant negative impact on Loons, indicating that OWFs deterred Loons through the combined effect of ship traffic and the wind turbines themselves. This study provides the first analysis of the extensive effects of OWFs and ships on Loons on a large spatial scale. The results provide an essential baseline for future marine spatial planning processes in the German North Sea and elsewhere.

1. Introduction

Shallow-shelf sea areas have long been used by humans. The North Sea is amongst the most-intensively utilised sea areas worldwide for activities including fishing, transport, oil and gas drilling, and gravel extraction (Emeis et al., 2015; Halpern et al., 2008). The installation of offshore wind farms (OWFs) in many sea areas throughout Europe and elsewhere represents a relatively new human use requiring considerable attention in terms of the marine planning process. In order to meet their climate goals, many European governments have started to install and plan further OWFs within relatively large sea areas (e.g. Breton and Moe, 2009; Langston, 2010). Germany intends to extend its offshore power generation to 6,500 MW by 2020 and to 15,000 MW by 2030, leading to a large increase in the number of OWF sites, mainly in the German North Sea, making Germany one of the countries with the most extensive plans for OWF installations (Beiersdorf and Radecke, 2014). Seventeen OWFs are currently (2018) in operation, with five further ones under construction and several more being approved in German sea areas (BSH, 2017).

In terms of the process of marine spatial planning, these permanent installations at sea represent a major addition to other types of marine human activities, whilst competing with sea areas assigned for nature conservation (Emeis et al., 2015; Moksness et al., 2009; Nolte, 2010) and potentially overlapping with areas used by resting and foraging seabirds. Previous studies have pointed out contrasting effects (negative or positive) of OWFs on seabirds that vary strongly among areas and species (Dierschke et al., 2016; Drewitt and Langston, 2006; Fox and Petersen, 2006; Furness et al., 2013; Garthe and Hüppop, 2004; Masden et al., 2009). In addition, the construction and maintenance of OWFs is further associated with a strong increase in shipping activities in and around OWFs (Exo et al., 2003).

OWFs may have direct effects on birds such as collision of individuals with the turbines, with subsequent impacts on the whole population (Fox et al., 2006; Goodale and Milman, 2014; Masden et al., 2009). Furthermore, the energy budget and condition of individual birds may also be affected indirectly through the effects of OWFs on habitat loss and reduced food availability (Drewitt and Langston, 2006; Fox et al., 2006; Stienen et al., 2007), though the long-term effects of these indirect effects at the population level are hard to estimate (Fox et al., 2006; Goodale and Milman, 2014; Searle et al., 2017). However,

* Corresponding author.
E-mail address: schwemmer@ftz-west.uni-kiel.de (P. Schwemmer).

https://doi.org/10.1016/j.jenvman.2018.10.053
Received 10 August 2018; Received in revised form 20 September 2018; Accepted 15 October 2018

birds have been shown to lose suitable resting and foraging habitats or to select less suitable sea areas (Stienen et al., 2007). Furthermore, they may need to increase their flight time by flying around OWFs on their way to suitable foraging sites (Drewitt and Langston, 2006; Masden et al., 2009). This study aimed to quantify the indirect effects (i.e. habitat loss by OWFs and associated ship traffic) on Loons (*Gavia* spp.) to provide baseline data for future studies that might address population consequences.

Loons belong to the most sensitive species group with respect to the avoidance of OWFs, as shown for single OWF sites in the North Sea (e.g. Dierschke et al., 2012, 2016; Leopold et al., 2010; Mendel et al., 2014; Petersen et al., 2006a, b; Welcker and Nehls, 2016). Furthermore, Red-throated Loons (*Gavia stellata*) are also very sensitive to ship traffic, demonstrating long flush distances in front of approaching vessels (Bellebaum et al., 2006) and significantly lower densities in areas with permanently higher ship traffic (Hüppop et al., 1994; Schwemmer et al., 2011). Their sensitive nature and the fact that a significant proportion of the biogeographic population occurs in European waters means that Loons are listed in Annex I of the EU Birds Directive and are considered to be particularly threatened with respect to human activities (e.g. Furness et al., 2013; Garthe and Hüppop, 2004). Negative effects on Loons at both the individual and population levels as a result of avoidance of OWFs cannot be ruled out (Dierschke et al., 2016, 2017), and Loons are therefore currently rated as a species group requiring particular consideration with respect to marine spatial planning in Germany and the UK (Busch et al., 2013).

Most Loons in the North Sea are Red-throated Loons (90%), with a minor proportion of Black-throated Loons (*G. arctica*; 10%) (Dierschke et al., 2012; Garthe et al., 2007). The German North Sea represents one of the most important resting sites for Loons with internationally important numbers, especially during spring migration (Garthe et al., 2007, 2015; Mendel et al., 2008; Skov et al., 1995), when around 20,200 Loons use German waters (Garthe et al., 2015). The 'Eastern German Bight' Special Protection Area (SPA) has been established to acknowledge the importance of this resting site and the high sensitivity of Loons with respect to human disturbances (Fig. 1). However, there is a potential conflict with the 'Butendiek' OWF, which was approved before but installed after the establishment of the SPA (Garthe et al., 2012), while further OWFs ('Helgoland Cluster') are located just south of the border of the SPA (Fig. 1).

Information on the long-term and large-scale effects of OWFs on Loons is currently limited and there has been no long-term comparison of their distributions before and after the installation of OWFs. Furthermore, the effects of increasing construction- and maintenance-related ship traffic have rarely been considered (Boon et al., 2010; Christensen et al., 2003).

We therefore hypothesized that Loons would avoid OWF areas and that their distribution patterns would differ before and after the installation of OWFs. We also hypothesized that the ship traffic associated with OWF sites would cause avoidance reactions among Loons. Against this background, this study aimed to shed light on five specific topics. (1) We had access to a long-term dataset covering the 14-year period before the installation of the OWFs ('before'). We therefore aimed to compare this information directly with the distribution of Loons after the installation of OWFs ('after'), using a long-term perspective not achievable in most previous studies. Mandatory operational monitoring of the four offshore windfarms in focus is still ongoing. (2) Most previous studies of the potential effects of OWFs on Loons have focussed on the effects of single OWF sites and their direct vicinities (see Dierschke et al., 2016). These therefore only allowed the reactions of Loons to be studied on a relatively small spatial scale, and could only show that Loon numbers were impacted within the respective site but could not show where they had moved to (Rexstad and Buckland, 2012). In contrast, the current study aimed to analyse the large-scale effects of multiple OWFs on Loon distribution, considering potential shifts between the 'before' and 'after' periods. (3) There is currently a need to

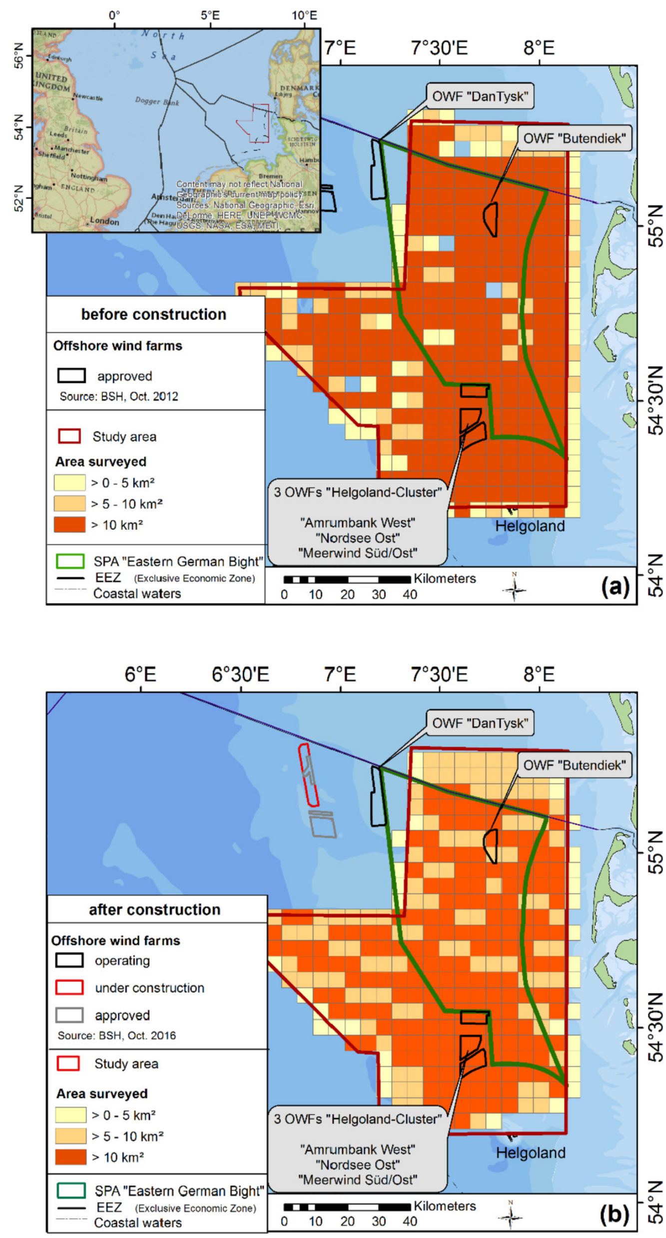

Fig. 1. Location of the study site within the south-eastern North Sea (inserted map in Fig. 1a) and in the eastern German Bight (North Sea) with locations of the different OWFs and the area surveyed for Loon abundance (yellow to red squares) across the 'Eastern German Bight' Special Protection Area (SPA; bold green line) for the 'before' (a) and 'after' periods of the analysis (b). Start of construction: 'Nordsee Ost' OWF during summer 2012; end of all construction works: 'Butendiek' OWF during summer 2015. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

disentangle the potential effects of OWFs from the effects of natural habitat characteristics that determine the distribution of Loons (Garthe, 1997; Winiarski et al., 2014). We therefore developed a model including stable natural parameters such as water depth and distance to land, as well as anthropogenic predictors such as distance to closest OWF and shipping traffic. (4) Given that the installation and maintenance of OWFs is associated with large increases in ship traffic, the effects of shipping need to be quantified and separated from the effects of the OWFs themselves. To date, this only has been analysed based on general ship densities (e.g. APEM, 2013; 2016; Leopold et al., 2014), while OWF ships present a dynamic source of disturbance for Loons.

This study therefore aimed to relate Loon and ship distributions at very high spatial and temporal scales by relating ship distributions derived from the Automatic Identification System (AIS) with Loon abundance assessed during aerial surveys. (5) Given a negative effect of OWFs on Loons, we aimed to quantify the avoidance distance to OWFs to draw conclusions about the degree of resulting (permanent) habitat loss.

In this study, we adopted two different approaches to analyse different aspects of the effects of OWFs on Loons: we used 'before' data to demonstrate the importance of the OWF areas before construction, and also focused on the simultaneous effects of OWFs and ships associated with OWFs after construction. The combined interpretation of these approaches allowed a comprehensive evaluation of the effects of OWFs on Loons.

2. Methods

2.1. Study area

The study was conducted within the eastern part of the Exclusive Economic Zone of the German North Sea, south of 55°17′ N, north of 54°11′, east of 6°30′ E, and west of 8°9′ E (Fig. 1a). The study site was located within an area 8–100 km off the Wadden Sea islands of northern Germany. The water depth ranged from 10 to 40 m. Loon distribution was recorded within the SPA 'Eastern German Bight' and beyond, and the study site therefore covered the core area of highest Loon densities within German waters (Garthe et al., 2015). The 'Butendiek' OWF is located in the core area of the SPA, while the 'Helgoland Cluster' OWFs are located at the border of the SPA and south of the core Loon distribution (Fig. 1a).

2.2. Recording Loon distribution and data processing

Loon distribution was recorded, both, in the period prior to OWF construction and in the period after construction:

(1) Before construction: These data cover the months of spring migration (i.e. March to April) of the years 2000–2013 and are the similar database as used by Garthe et al. (2015). The records originated from environmental impact assessment studies required for licensing procedures of offshore wind farms in the German EEZ and from seabird monitoring and research programmes (for details see Garthe et al., 2015; Fig. 1a). The data were recorded using visual aerial and ship-based surveys. Briefly, Loons were counted along transects of a known area, which allowed the densities to be computed (see Diederichs et al., 2002; Garthe et al., 2002 for a full description of both recording methods).
(2) After construction: These data also cover the months of spring migration (i.e. mainly March to April, but including the last week of February and the first week of May to enhance the sample size of surveys) of the years 2015–2017. Data originated from ongoing mandatory monitoring of the wind farms during operation, and from the 'Helbird' research project funded by the German Federal Ministry for Economic Affairs and Energy. Overall, data for the after period were based on 10 digital aerial surveys in 2015–2017 (Fig. 1b). Those data were obtained by video-based digital recordings instead of visual observations. Briefly, an aircraft sampled a transect of a known area using a video camera and all seabirds found were recorded and used to compute overall densities (for a detailed description of the method see Buckland et al., 2012; Thaxter and Burton, 2009). A change from visual to digital survey methods was mandatory for safety reasons because the flight altitude needed to be higher during the construction and operational phases of the turbines (168 m, instead of 91 m for visual observations), which excluded visual recordings.
(3) During construction: No data were considered in this study, as disturbance during the construction of the OWF is temporary and mainly associated with construction ships, and its contribution to the overall effect of the OWF on the Loon population was assumed to be of low importance in relation to the expected lifetime of the OWF (Christensen et al., 2003).

Visual observations of seabird distributions are known to underestimate birds in parts of the transect further from the observer (Buckland et al., 2001, 2015). We therefore applied a species-specific correction factor for aerial and ship-based observations, respectively (see Garthe et al., 2015 for details). However, no distance correction was necessary for the video-based digital surveys because the probability of detecting a bird was equal across the whole transect.

All three recording methods relied on the principle that transect sampling of birds could be used to compute densities. However, we did not compare absolute density values between the 'before' and 'after' periods, because the visual and digital methods have not been confirmed to produce the same absolute values (Buckland et al., 2012; Skov et al., 2016); this could only be tested by performing both methods at the same time, and no such dataset is currently available. Thus, both periods were compared by computing the relative deviance from the maximum density in each period in %, and using this to compare the distributions and locations of high-density areas of Loons between the two periods.

Data were spatially pooled in a grid with cells of 2.5 × 2.5 km for the 'before' and 'after' periods, for each of the three methods (visual aerial and ship-based surveys, video-based digital recordings), respectively. Bird numbers and monitored areas were each summed per grid cell, and eventually used to compute mean densities for each period, while geographical coordinates were averaged for each cell.

2.3. Integrating covariates for the 'before–after' control impact (BACI) approach

We related the average distribution data for Loons with environmental variables using ArcGIS (version 10.3; Environmental System Research Institute, 2016). The environmental variables included: (1) *dist_coast* = minimum distance to the mainland and larger islands (except Helgoland); (2) *dist_helgoland* = minimum distance to Helgoland; (3) *dist_owf* = minimum distance to the border of the OWF; and (4) *mean_depth* = mean water depth.

This first model, hereafter named the BACI approach, did not consider the effect of ships because ship data at a sufficiently high spatiotemporal resolution were only available for the 'after' period. To distinguish between the effect of the OWFs and the effect of ship traffic on Loons, we therefore developed a second model (ship model) using only the data from the 'after' period.

To merge the environmental variables with the bird-count data in an optimal way, we first pooled the covariates to a spatial grid of 2.5 × 2.5 km, and then fitted each covariate with a generalised additive model (GAM) using the function gam() in the R-package mgcv (R Core Team, 2017; R version 3.4.2; Wood, 2006). We used only latitude and longitude as a smooth 2D-predictor based on cubic splines with the maximal degree of freedom, so that the result represents a cubing interpolation on the given (possibly irregular) grid. Thirdly, we used the predict() function to predict the values straight to the coordinates as given in the pooled bird-count data. Finally, the additional categorical variable *owf_zone* for 'inside OWF-affected area' vs. 'outside OWF-affected area' was defined for two different zones: 1) inside: ≤ 3 km vs. outside: > 3 km (measured from the nearest turbine), given that OWF-associated ships operate mainly within a 3 km radius around the OWF and this distance class has been used in previous studies of the impact of single OWFs (Vanermen et al., 2015a; Welcker and Nehls, 2016); and 2) inside: ≤ 10 km vs. outside: > 10 km, because an initial analysis showed the greatest decrease in Loon densities up to a distance of 10 km from the turbines.

2.4. Set up and validation of regression models for the BACI approach

The BACI approach is based on surveying a potentially impacted situation and a control situation before the impact (variable '*period*'), and relative comparisons of spatial and temporal differences can then be used to extract the unbiased impact (Schwarz, 2014; Smith, 2002). We formulated the BACI approach within the framework of generalised additive mixed models (GAMMs), which are known to describe biological count data appropriately (Zuur et al., 2007, 2009; 2012). We used a continuous linear or smooth predictor measuring the distance to the border of the next OWF. This allowed us to estimate how the abundance of Loons changed in relation to the distance from the OWF and to estimate avoidance distances. Notably, we introduced a variable for the observation method ('visual ship-based surveys' vs. 'visual aerial surveys' vs. 'digital aerial surveys') as a random intercept to account for differences in detection among these methods. We were aware that this variable was partially collinear with the variable 'period' because only digital aerial surveys were used 'after' and only visual surveys were performed 'before'. Importantly, the estimation of the interaction term 'period x wind_farm' (see below) representing the BACI approach was not influenced by this, because only relative differences in Loon densities were evaluated.

This approach produced the following full model for the BACI approach (not yet thinned regarding its predictors; see below):

$$\log(y_{ij}) = \beta_0 + u_i + \mathrm{f(mean\_depth_j)} + \mathrm{f(dist\_coast_j)} + \mathrm{f(dist\_helgoland_j)} + \mathrm{s(latitude,longitude)} + [\mathrm{wind\ farm_j}] + \mathrm{period_j} + [\mathrm{wind\ farm_j}] \times \mathrm{period_j} + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \tag{1}$$

where $\varepsilon_{ij} \sim N(0, \sigma^2)$ and $u_i \sim N(0, \sigma_u^2)$ were independent and identically distributed. Here, y_{ij} is the vector of bird numbers, where the index j refers to the observation number and i is related to the method-ID. *f*() depicts either a linear term or a cubic regression spline *s*() (tested during predictor selection), where, in the case of a spline, the optimal number of knots was estimated via cross-validation. The variable [*wind_farm$_j$*] was either considered as a linear term, *dist_owf$_j$* measuring the distance to the next wind turbine, as an additive smoother, *s*(*dist_owf$_j$*), or as a bivariate variable, *owf_zone$_j$*, the latter distinguishing between 'inside OWF-affected area' and 'outside OWF-affected area'. For each model, an appropriate probability distribution was selected for y_{ij} via Akaike Information Criterion (AIC; Akaike, 1973) analysis (see below).

We modified the common selection and validation strategies to validate the optimal GAMM model (Field et al., 2012; Korner-Nievergelt et al., 2015; Zuur, 2012; Zuur et al., 2009, 2010; 2012) using the following steps: (1) Based on the entire model (1), we selected an appropriate probability distribution/stochastic part of the model using the AIC. Namely, we compared Poisson-, negative binomial-, Tweedie-, zero-inflated Poisson distribution, and observation-level random intercept Poisson models. All five probability distributions are known to describe the stochastic part in regression models of (overdispersed) count data reasonably well (Kokonendji et al., 2004; Korner-Nievergelt et al., 2015; Linden and Maentyniemi, 2011; Zuur et al., 2012). (2) The optimal model regarding the set of fixed-effect predictors was selected from the full model by comparing 16 different models. (3) Model validation was carried out by visual inspection of the residual plots to assess all the required model assumptions (Zuur et al., 2010). Corresponding auto-correlation structures were added to the model if required.

AIC favoured a negative-binomial distribution, and subsequent predictor selection produced the following final model:

$$\log(y_{ij}) = \beta_0 + u_i + \beta_1 \mathrm{dist\_coast_j} + \mathrm{s(latitude,longitude)} + [\mathrm{wind\_farm_j}] + \mathrm{period_j} + [\mathrm{wind\_farm_j}] \times \mathrm{period_j} + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \tag{2}$$

Residual analysis revealed no violation of linearity, homogeneity, independence, or normality of the random intercept.

2.5. Integrating covariates for the ship model

Ship traffic has been shown to have a significant effect on Loon distribution (Bellebaum et al., 2006; Schwemmer et al., 2011), and ship traffic in the study area has increased greatly due to the construction and maintenance of OWFs. It is therefore important to disentangle the effects of these two sources of anthropogenic activities (OWFs and ship traffic) on Loons. Ship traffic shows temporal inhomogeneity, with more traffic in the morning and evening hours, and it was therefore necessary to consider the data spatio-temporally instead of purely spatially, as with the BACI approach. Data were only used for five digital-survey flights from the 'after' period because no real-time ship data were available for the 'before' period or for any other survey days during the 'after' period. Bird data were spatially assigned to an optimal grid of 2.5 × 2.5 km for each survey day separately and treated as described above. To consider the time, we also calculated the mean time at which the Loon observations were recorded for each grid cell.

Data on ship distributions were recorded in parallel with the digital-survey flights to record Loon distribution using an AIS spotter (www.aisspotter.com). Because the ship data consisted of irregular position data in terms of time and space, they were linearly interpolated to obtain positions at least every minute. To merge the ship data with the Loon-distribution data, it was assumed that all ships within the time interval $[t - \delta_t, t]$ and within a circle around (x, y) with radius r may influence bird density, for each time point t and each pair of spatial coordinates (x, y). Given that the optimal values δ_t and r are not known *a priori*, we tested all existing combinations between δ_t ε {2, 60, 120, 180, 250, 300, 350, 400, 600, ∞} sec and r ε {1, 2, 3, 4, 5, 6, 7, 8, 9, 10} km, and created a separate variable counting all ships within the given time and space interval for each of the 100 combinations. Here, $\delta_t = \infty$ depicts a case where all available ship data have only been spatially correlated to bird-count data without considering temporal distance to the observations. We subsequently compared 100 resulting regression models (see below) to find the optimal values of δ_t and r. However, the AIC value was not appropriate for selecting the optimal model because the ship densities and OWF-related variables were collinear, and the model with only one of both variable types would be favoured due to the parsimony of the AIC-based selection. In contrast, we aimed to consider both (collinear) variables to distinguish explicitly between the unique effects of ships and wind turbines on Loon abundance. An appropriate measure should thus relate the effect size of the ship-dependent variable with its reliability. Hence, we selected the model with the highest $|\beta|/SE_\beta$ value, where β is the ship-related regression coefficient and SE_β is its standard error.

2.6. Set up and validation of regression models for the ship model

The GAMMs were set-up as described above for the BACI approach. Notably, the ID of the digital-survey flight was introduced as a random intercept to account for different numbers of birds or different monitoring conditions between surveys.

This produced the following GAMM structure of the ship model (not yet thinned regarding its predictors):

$$\log(y_{ij})\beta_0 + u_i + \mathrm{f(mean\_depth_j)} + \mathrm{f(dist\_coast_j)} + \mathrm{f(dist\_helgoland_j)} + [\mathrm{wind\_farm_j}] + [\mathrm{ship\_number_j}] + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \tag{3}$$

where $\varepsilon_{ij} \sim N(0, \sigma^2)$ and $u_i \sim N(0, \sigma_u^2)$ were independent and identically distributed. Here, y_{ij} is the vector of bird numbers, where the index j refers to the observation number and i is related to the survey flight ID. *f*() depicts a linear or smooth predictor (tested during AIC-based predictor selection). The variable [*wind_farm$_j$*] was either considered as a binomial predictor ('inside' vs. 'outside'), a linear term (distance to the

Journal of Environmental Management 231 (2019) 429–438

OWF border), or a cubic regression spline depending on the latter. The variable [*ship_number$_j$*] was considered as the total number of temporally and spatially related ships, additionally depending on the *a priori* defined parameters δ_t and r (see above). In contrast to the BACI approach, we did not consider a spatial smooth because this predictor would interfere with the correct estimation of [*wind_farm$_j$*]. GAMM-model selection and validation strategies were performed as described for the BACI approach (see above), including integration of the appropriate autoregression structures (if required).

AIC-based selection of the probability distribution again favoured a negative-binomial distribution. The optimal values of δ_t and r required to blend the observation and ship data showed that the highest (β/SE)-values (indicating high precision of the ship-related regression coefficient) were $\delta_t = 5$ min and $r = 5$ km. Subsequent predictor selection revealed the following final model:

$$\log(y_{ij}) = \beta_0 + u_i + \mathrm{f}(\mathrm{mean\_depth_j}) + \beta_1 \mathrm{dist\_coast_j} + \mathrm{s}(\mathrm{dist\_helgoland_j}) + [\mathrm{wind\_farm_j}] + [\mathrm{ship\_number_j}] + \mathrm{offset}(\log(\mathrm{area_j})) + \varepsilon_{ij} \quad (4)$$

where *s*() depicts the cubic regression splines with optimal degrees of freedom estimated via cross-validation.

Analysis using different sizes of the underlying spatial grid for spatio-temporal pooling revealed an optimal grid size of 2.5 × 2.5 km, leading to a temporal autocorrelation of model residuals of order 2 (in contrast to the model based on raw data, where the autoregressive order (AR order) was > 30). Model-validation plots indicated no violation of linearity or homogeneity, spatial residual plots and a semi-variogram indicated no violation of spatial independence, and a plot of the partial autocorrelation function (pACF-plot) revealed a temporal autocorrelation of approximately order 2, which was integrated as an AR(2)-structure into the model.

3. Results

3.1. Loon abundance before and after OWF installation

The spatial distribution patterns of Loons changed profoundly between the ‘before’ and ‘after periods (Fig. 2). During the ‘before’ period, the core area with the highest Loon densities clearly overlapped the area of the planned ‘Butendiek’ wind farm, while moderately high densities stretched out to the area of the planned ‘Helgoland Cluster’. In contrast, there was a clear shift to the area located between these two OWF sites during the ‘after’ period (Fig. 2). The areas of the OWFs themselves, as well as the immediate vicinities, showed extremely low abundances of Loons during the ‘after’ period. The core area of Loons during the ‘after’ period was thus still located in the centre of the SPA, but the birds were more aggregated within the still-undisturbed sea area.

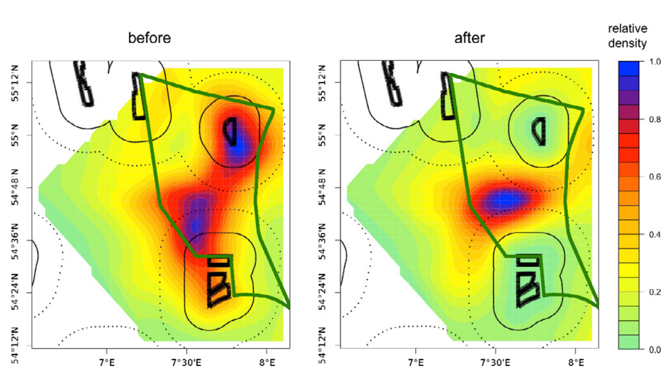

Fig. 2. Spatial density plots of predicted Loon distributions ‘before’ vs. ‘after’ the construction of OWFs, based on the BACI-GAMM. Bold black lines: OWFs; thin black lines: 10 km distance buffer; dotted black lines: 20 km distance buffer; bold green line: Special Protection Area.

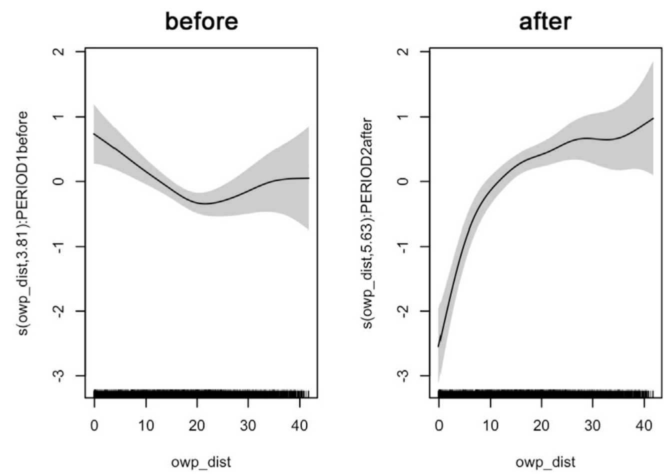

Fig. 3. Change in Loon abundance in relation to distance from the closest OWF site ‘before’ (left) and ‘after’ construction of the OWFs. Smoothed curve: predicted number of Loons at a given distance from the closest OWF; shaded area: 95% confidence interval; small lines on the x-axis: observations of Loons at a given distance from the OWF.

We also introduced the distance from the wind farm as a smooth term, estimated separately for each period. This revealed a striking difference between the two periods (Fig. 3): the ‘before’ plot suggested that the future wind farm areas were sites with naturally increased Loon abundance, while the ‘after’ plot showed a strong decline in Loon abundance due to the OWFs (Fig. 3). The start of this decline was already visible at > 20 km from the OWFs (see also dotted black lines in the ‘after’-plot in Fig. 2b). To determine the distance from the wind farm at which the decline in abundance was significant, we approximated the first derivative of the corresponding smooth (Fig. 3 ‘after’) by calculating its first finite difference.

To determine the distance at which the change in Loon density became significant, we calculated confidence intervals for the first derivatives via bootstrap analysis and subsequently evaluated where the lower confidence interval intersected with zero. This occurred at around 16.5 km from the OWFs (Fig. 4). However, the greatest decline in density was at distances within 10 km from the OWF (Figs. 3 and 4). Avoidance of wind farms within 10 km was also clearly visible in the distribution maps (solid black lines in Fig. 2b).

Additionally, the binomial wind farm-related variable *owf_zone* was highly significant for both radii (3 or 10 km, respectively). The abundance of Loons decreased highly significantly by 94.5% inside the 3 km zone around the OWFs within the study site (interaction term in Table 1; $\beta = -2.9$, $p < 0.001$), while the abundance was still decreased by 83.7% inside the 10 km zone (Table 2, $\beta = -1.8$, $p < 0.001$). The distance to land (*dist_coast*) had no significant effect on Loon densities (Table 1; Table 2).

3.2. Distinguishing between effects of ships and OWFs

Loon densities were still reduced if ships were included in the overall model as a predictor for the ‘after’ period, as was the case without considering the effect of ships, as shown above. Applying a 3 km radius around the wind farms, OWFs alone reduced the Loon density by 70.8% compared with the sea areas outside the OWFs ($p < 0.001$; Table 3). If the radius was extended to 10 km around the OWFs, the Loon density was still reduced by 44.5% ($p < 0.001$) by the OWFs alone.

When ships as single predictor were removed from the model, the estimated effect of OWFs (now combined with the effect of the ships) on Loons was 84% using a 3 km radius ($p < 0.001$). This suggested that

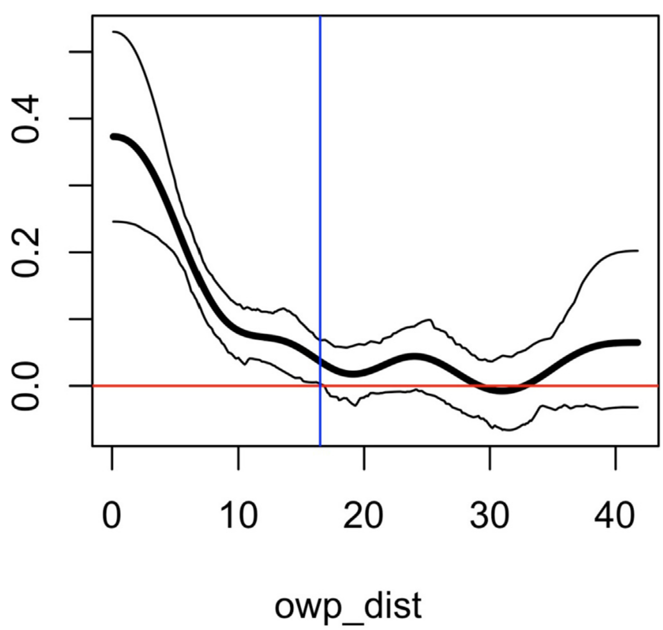

Fig. 4. First finite difference of the smooth depending on the distance from the closest OWF, partially evaluated for the 'after' period. Red line indicates a derivative of zero, blue line indicates distance at which the derivative was significant. Thick black line corresponds to the first derivative; thin black lines depict 95% confidence intervals.

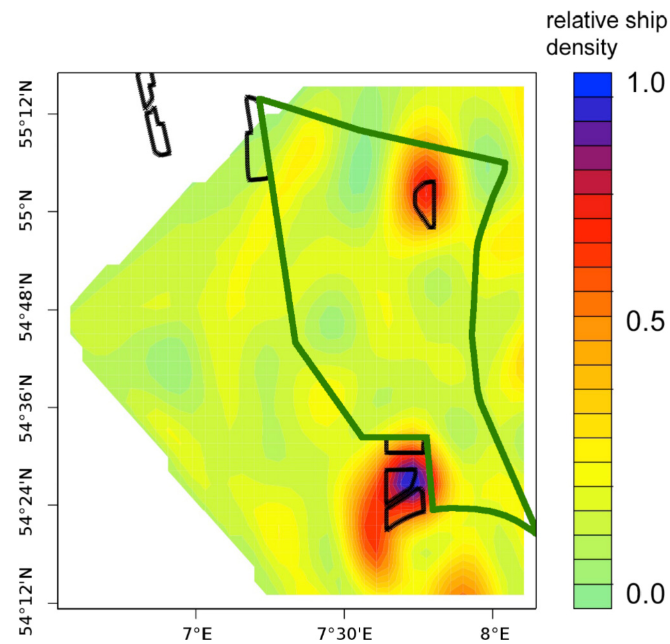

Fig. 5. Spatial density plot of ship distribution in the 'after' period based on AIS data.

Table 1
Regression results of the BACI approach–GAMM using the binomial variable 'inside wind farm' vs. 'outside wind farm' (*owf_zone*) for a radius of 3 km.

	Estimate	Std. Error	z value	Pr(> |z|)
(Intercept)	−1.05	1.17	−0.90	0.37
period[after]	0.40	0.85	0.47	0.64
owf_zone[a][inside]	0.70	0.13	5.07	< 0.001
dist_coast[b]	0.02	0.02	0.70	0.43
period[after]xowf_zone[inside]	−2.90	0.22	−13.16	< 0.001

[a] Offshore wind farm zone.
[b] Distance to coast.

Table 2
Regression results of the BACI approach–GAMM using the binomial variable 'inside wind farm' vs. 'outside wind farm' (*owf_zone*) for a radius of 3 km.

	Estimate	Std. Error	z value	Pr(> |z|)
(Intercept)	−1.41	1.17	−1.21	0.23
period[after]	0.73	0.95	0.76	0.45
owf_zone[a][inside]	0.66	0.12	5.59	< 0.001
dist_coast[b]	0.02	0.02	1.01	0.31
period[after] xowf_zone[inside]	−1.81	0.12	−15.26	< 0.001

[a] Offshore wind farm zone.
[b] Distance to coast.

ships also had a strong negative effect on Loon abundance, accounting for at least 14% of the joint OWF–ship effect.

Thus, in the ship model, the effect of OWFs alone was not as strong as estimated by the BACI approach (i.e. without considering ship traffic; > 94% and > 84%, respectively). There are two possible explanations for these different estimations. (1) the ship model was only fitted using data from the 'after' period because no ship data were available for the 'before' period. Hence, the estimated reduction in effect does not take account of the fact that bird densities within the OWFs showed the highest Loon abundances before the construction of the farms (see above), leading to a strong underestimation of the reduction effect. (2) Although the ship model considered the effect of ships, these were at least partially correlated with OWF location (Fig. 5). Thus the BACI approach actually estimated the joint reduction effect of OWFs and ships, whereas the ship model evaluated both impacts separately, which may have led to a reduction in the OWF effect compared with the BACI approach.

Indeed, the ship model showed a significant negative impact of ships on Loon abundance (Tables 3–4), with a highly significant decline of 31% in abundance for each additional ship in the spatio-temporal range of the Loons (i.e. 5 min and 5 km from the Loon sighting; see Methods) ($p < 0.001$). This suggests that one in three Loons left the area as one ship approached. The spatial component of ship disturbance was much stronger than the temporal component; i.e. our regression models selecting for the optimal δ_t and r revealed that ships within 5 km had a strong impact on Loon abundance, whereas the time lag between

Table 3
Regression results for the *ship-owf*-approach–GAMM distinguishing between the effect of ships and the effect of OWFs in the 'after' period for a radius of 3 km.

	Estimate	Std. Error	z value	Pr(> |z|)
(Intercept)	0.29	0.58	0.5	0.62
owf_zone[a][inside]	−1.23	0.31	−4.03	< 0.001
dist_coast[b]	−0.01	0.01	−0.55	0.58
n_ships[c]	−0.37	0.08	−4.82	< 0.001

[a] Offshore wind farm zone.
[b] Distance to coast.
[c] Number of ships.

Table 4
Regression results for the *ship-owf*-approach–GAMM distinguishing between the effect of ships and the effect of OWFs in the 'after' period for a radius of 10 km.

	Estimate	Std. Error	z value	Pr(> |z|)
(Intercept)	0.73	0.58	1.26	0.20
owf_zone[a][inside]	−0.59	0.17	−3.51	< 0.001
dist_coast[b]	−0.01	0.01	−1.00	0.32
n_ships[c]	−0.48	0.07	−6.44	< 0.001

[a] Offshore wind farm zone.
[b] Distance to coast.
[c] Number of ships.

the Loon sighting and the AIS signal of the ship was less relevant (with an optimum at approx. 5 min). This suggests that ships may affect Loons most strongly at a distance of ≤5 km.

As seen with the BACI approach, the distance to land had no significant influence on Loon abundance (Tables 3–4).

4. Discussion

4.1. Distribution patterns before and after OWF installation

Our results demonstrated that the distribution patterns of Loons, which had remained stable over a period of many years (Garthe et al., 2015), were substantially altered at both small and large spatial scales by the installation of OWFs in the German North Sea. We developed our BACI approach on a solid database including 14 years of large-scale surveys in the period 'before' OWF installation. To the best of our knowledge, all previous reports have been based on a maximum of 1–3 years of data prior to the construction of OWFs, and have mostly focused on the effect of a single OWF (e.g. Leopold et al., 2013; Petersen et al., 2014). Although we were unable to compute absolute differences in Loon populations between the two periods due to a change in survey methods, our results demonstrated profound large-scale shifts in distribution patterns, as well as significant avoidance of the OWF areas.

We observed a shift in the Loon-abundance hotspot to the western–central area of the SPA that remained undisturbed by OWFs in the 'after' period. This hotspot is located about 20 km distant from all surrounding OWFs. Several previous studies have highlighted the environmental parameters that are most important for determining Loon distribution patterns. Frontal systems are expected to increase prey availability for Loons (Skov and Prins, 2001), while nearshore and shallow sandy sea areas also play a major role (O'Brien et al., 2008; Skov and Prins, 2001; Skov et al., 2016). Our results suggest that the area of the 'Butendiek' OWF, which was installed in the northern part of the SPA, was of particular importance for Loons before the construction of this OWF, given that this was the area of maximum Loon abundance during the 'before' period. The 'Helgoland-Cluster' OWFs are located south-west of the border of the SPA, and our results showed that, in contrast to the 'Butendiek' area, Loon abundances in the 'before' period were significantly lower compared with abundances within the SPA. However, Loons are known to occur here regularly (Garthe et al., 2015).

One aim of this study was to disentangle the importances of natural habitat structures and anthropogenic pressures on Loons. Our modelling approach showed that natural habitat predictors, such as distance to the coast/Helgoland and water depth did not play major roles compared with the effects of OWFs and shipping (see below). This suggests that anthropogenic pressures are the most important factors driving the distribution patterns of Loons within their natural hot spots.

Still, we cannot completely rule out that undetected changes in ecological conditions might have additionally led to the shift in distribution patterns. For instance, it could be assumed that Loons might have followed shifts in their prey community. However, given that Loons are known to feed on a variety of fish species (Guse et al., 2009), a shift in fish distribution that could account for the change in Loon distribution seems highly unlikely. The reef effect is even known to likely increase benthic and fish communities inside OWFs (e.g. Vandendriessche et al., 2015; Vanermen et al., 2015a) which in turn may enhance the quality of these sites for piscivorous seabirds. However, given that these sites were avoided by Loons, despite of a likely higher fish availability and as Loon distribution patterns had been stable over a period of many years in the 'before' period (Garthe et al., 2015), it seems to be convincing that OWFs and associated ship traffic are the main factors explaining the shifts in distribution patterns.

Incorporating distance from the nearest OWF as a smoothed term in the model allowed us to highlight the fact that Loons reacted as far as 20 km from OWFs, with significant changes in densities at a distance of 16.5 km and the greatest changes in abundance within 10 km. These values were higher than those reported in previous studies (summarized in Dierschke et al., 2016; Welcker and Nehls, 2016). However, most previous studies only investigated local avoidance effects (often only up to 4 km distance; Leopold et al., 2013; Petersen and Fox, 2007, Petersen et al., 2006a,b; Welcker and Nehls, 2016) and were therefore unable to detect any larger-scale avoidance reactions. This highlights the importance of a sufficiently large-scale approach and the inclusion of multiple OWF sites (Rexstad and Buckland, 2012), as in the current study. To emphasize the importance of scale, we quantified the effects of OWFs on Loons by defining the affected sea areas by both 3 km and 10 km radii.

The 3 km distance class was chosen based on previous studies that showed avoidance distances for single OWFs up to this value (Vanermen et al., 2015a; Welcker and Nehls, 2016). However, our results suggest that this distance was too short, based on the effects of multiple OWFs on a larger spatial scale.

The reason for the relatively large-scale effect of OWFs on Loons detected in the current study is not completely clear. It is possible that visual cues are not the only reason for the large disturbance distance. Previous studies showed that OWFs not only affected seabirds and other marine wildlife directly (Bergström et al., 2014; Goodale and Milman, 2014; Lindeboom et al., 2011), but may additionally cause changes in the abiotic environment, such as sediment properties and water stratification due to turbulence caused by the piles (Carpenter et al., 2016; Nagel et al., 2018). Carpenter et al. (2016) pointed out that an individual OWF may enhance mixing of the water column, with a cascade of effects on the whole ecosystem in an area of 10–20 km from the OWF, though the physical–biological interactions remain unclear. This was in accordance with the disturbance distance of Loons found in the current study. Petersen et al. (2014) also showed significantly lower Loon abundances up to 13 km from OWFs, which also matched the results of the current larger-scale approach.

Finally, it is important to critically explore the question of the power of the data used in this study. For the type of data used, previous investigations have shown that high survey intensities are required to safely trace declines in seabird populations, mainly as a result of high variability in distribution patterns (e.g. MacLean et al., 2013; Vanermen et al., 2015b). However, compared to our study that was conducted over a large sea area, both studies mentioned above focussed on rather small study sites, likely enhancing small-scale variability in counting data. According to Vanermen et al. (2015b) the statistical power after 10 years of survey was sufficiently high to detect reliable changes. For the 'before' period, 13 years of data were available for our BACI approach, indicating a valid data base. In contrast, the 'after' period only consists of 10 aerial surveys over a period of three years, suggesting that the data base for the 'after' period may still be too weak. However, the significant negative and consistent effects of OWFs and associated ship traffic on Loon distribution during all surveys of the 'after' period indicates that the data base is sufficient to yield valid results. Nevertheless, it will be necessary to enhance the data base for the 'after' period by future surveys to confirm the results and to enhance the statistical power.

4.2. Distinguishing between the effect of ships and OWFs

The installation of OWFs causes a substantial increase in ship traffic in the surrounding area due to maintenance and service activities (Exo et al., 2003). Although ship traffic is known to affect the distribution patterns of seabirds and particularly of Loons (Bellebaum et al., 2006; Schwemmer et al., 2011), the combined effect of OWFs and their associated ship traffic has rarely been reported; however, the few available studies noted a significant impact of ship traffic on Loon distribution (APEM, 2013, 2016; Leopold et al., 2014; Skov et al., 2016). Loons have been shown to exhibit a behavioural response to approaching ships, and flight distances of up to 2 km have been documented (Bellebaum et al., 2006; Schwemmer et al., 2011). This corresponds to the current results, which suggested a significant reduction in Loon densities within a radius of up to 5 km from the vicinity of ships, with the temporal aspect of ship distribution having little effect.

Inclusion of ship abundance in the model showed a reduced density of Loons of up to 70% based on the 3 km distance zone. This reduction could be considered to reflect the effect of the OWFs alone. In contrast, the joint effect of OWFs and ships led to a reduction of 84%, indicating the additional negative impact of ships on Loon densities. The exact reduction in densities due to ships alone could not be computed reliably because of the collinearity of ship traffic and OWFs. Importantly, their mobile nature means that ships are both spatially and temporally variable predictors, and a reliable estimation of their overall effects on birds will always be biased. This issue will remain difficult to address even in future studies, given that ships aggregate strongly in the vicinity of OWFs and present no fixed predictor.

The greater reduction in Loon densities following inclusion of ship traffic in the model demonstrates the importance of reviewing the cumulative impact of multiple anthropogenic pressures in the marine environment. Previous studies have focussed on cumulative effects simply by investigating the combined effects of multiple OWFs (Busch et al., 2013; Desholm, 2009; Dierschke et al., 2003, 2006, Fox et al., 2006; King et al., 2009; Mendel and Garthe, 2010). However, given the strong effect of ships on Loon abundance, it seems necessary to include other anthropogenic pressures in estimates of cumulative effects on Loon abundance in general.

4.3. Conclusions

The large-scale avoidance effects of OWFs (and ships) on Loons suggest that Loons are unlikely to suffer from enhanced direct mortality, e.g. because of collisions (Leopold et al., 2010; Petersen et al., 2006a,b; this study). Furthermore, a low flight altitude of only up to 10 m above the sea surface (Van Bemmelen et al., 2011) reduces the collision risk for Loons. Indirect effects, such as habitat loss, are thus likely to be key factors affecting Loons in relation to OWFs. However, the consequences of such indirect effects e.g. on population levels of seabirds, and density-dependent effects are hard to assess, and appropriate methodologies are largely lacking (Green et al., 2016; Horswill et al., 2017). When assessing the consequences of habitat loss due to the installation of OWFs and the associated enhancements in ship traffic, it is essential to consider which alternative sea areas could be used as resting and foraging grounds. In the current case, alternative sites seemed to be very limited because the SPA was virtually surrounded by OWFs. This might explain why Loons tended to concentrate in the centre of the SPA rather than moving outside it.

Although it was not possible to compute absolute differences in abundance between the 'before' and 'after' periods in this study, it is hoped that this issue will be resolved when enough data become available from parallel digital and visual surveys of sea areas where visual observations are still allowed. However, the relative reduced densities of Loons with respect to OWFs and ship traffic as well as the avoidance distances provided in the current study will serve as a baseline for further studies. A suitable approach for quantifying the overall habitat loss for Loons would involve computing the relative proportion of habitat loss within a certain area (e.g. within the SPA). Dierschke et al. (2006) suggested summing the total OWF areas and adding an additional buffer zone to assess the overall habitat loss. Applying this approach to the current study allowed the minimum habitat loss due to the OWFs in the SPA to be computed, indicating that complete loss of the sea area within a 3 km radius around the OWFs for Loons (as strongly supported by the current study) would equate to a loss of 8.8% of the SPA (overall size 3,135 km^2) for Loons. This should be regarded as an absolute minimum, given that our results clearly showed that the density of Loons was greatly reduced beyond 3 km from the nearest OWF.

Although we are not able to compare absolute density values between the 'before' and 'after' periods, our results indicated that Loons aggregated in the centre of the SPA after OWF installation, representing an increase in Loon density in a much smaller sea area. Given that Loons tend to occur in comparatively small flocks, only occasionally exceeding 5–10 individuals/km^2 (Garthe et al., 2015; O'Brien et al., 2012), this change in distribution might promote density-dependent effects (Blanc et al., 2006; Horswill et al., 2017; Lewis et al., 2001). A possible shift towards suboptimal habitats may lead to suboptimal body conditions prior to breeding, which could in turn reduce the reproductive success and enhance mortality in adult birds (Coulson et al., 1983; Hüppop, 1995). Even a slight increase in the mortality of adult Loons of only 0.3% can have significant negative effects on population levels (Rebke, 2005).

To assess the role of habitat loss on Loons, it is crucial to know if habituation to OWFs will occur or if the habitat loss will be permanent. Although studies from the UK and The Netherlands have indicated slight (though insignificant) increases in Loon abundances after 4–5 years since construction, studies from Denmark have shown no signs of habituation (Petersen and Fox, 2007; Petersen et al., 2008). Similarly, the current study found no habituation 3 years after construction. However, the monitoring of the operating wind farms is still ongoing and thus results on habituation are preliminary. Given that the degree of habituation remains very unclear, we strongly recommend the need for long-term monitoring to assess any potential large-scale effects of cumulative anthropogenic drivers on Loon distribution, particularly within the most relevant sea areas for Loons (e.g. Vanermen et al., 2015a,b).

Funding

This study was funded by the Federal Ministry for Economic Affairs and Energy according to the decision of the German Bundestag (project HELBIRD, grant number 0325751). Data assembled within the project WEBTOP (grant number 10033325), funded by the German Maritime and Hydrographic Agency (BSH), could also be used for analysis.

Acknowledgements

The following environmental consulting companies recorded bird distributions: APEM, BioConsult SH GmbH, IBL Umweltplanung GmbH, and Institut für Angewandte Ökosystemforschung GmbH. NAVAMA handled the AIS data. K Borkenhagen assisted with figure layout.

References

Akaike, H., 1973. Information theory and an extension of the maximum likelihood principle. In: Petrov, B.N., Caski, F. (Eds.), Proceeding of the Second International Symposium on Information Theory. Akademiai Kiado, Budapest, pp. 267–281.
APEM Ltd, 2013. Aerial Bird Surveys in the Outer Thames Estuary SPA. APEM Scientific Report for Natural England, pp. 67 Final.
APEM Ltd, 2016. Assessment of Displacement Impacts of Offshore Windfarms and Other Human Activities on Red-throated Divers and Alcids. Natural England Commissioned Reports No, pp. 227.
Beiersdorf, A., Radecke, A. (Eds.), 2014. Ecological Research at the Offshore Windfarm Alpha Ventus: Challenges, Results and Perspectives. Springer, Wiesbaden.
Bellebaum, J., Diederichs, A., Kube, J., Schulz, A., Nehls, G., 2006. Flucht- und Meidedistanzen überwinternder Seetaucher und Meeresenten gegenüber Schiffen auf See. Ornithologischer Rundbrief Mecklenburg-Vorpommern 45, 86–90.
Bergström, L., Kautsky, L., Malm, T., Rosenberg, R., Wahlberg, M., Capetillo, N.Å., Wilhelmsson, D., 2014. Effects of offshore wind farms on marine wildlife - a generalized impact assessment. Environ. Res. Lett. 9, 034012.
Blanc, R., Guillemain, M., Mouronval, J.B., Desmonts, D., Fritz, H., 2006. Effects of non-consumptive leisure disturbance to wildlife. Rev. Ecol. Terre Vie 61, 117–133.
Boon, A.R., ter Hofstede, R., Klok, C., Leopold, M., Blacquiere, G., Poot, M.J.M., Kastelein, R.A., Camphuysen, C.J., 2010. Monitoring and Researching Ecological Effects of Dutch Offshore Wind Farms. Masterplan. Deltares, Utrecht/Delft.
Breton, S.P., Moe, G., 2009. Status, plans and technologies for offshore wind turbines in Europe and North America. Renew. Energy 34, 646–654.
BSH, 2017. Bundesfachplan Offshore für die deutsche ausschließliche Wirtschaftszone der

Nordsee 2016/2017 und Umweltbericht. https://www.offshore-stiftung.de/sites/offshorelink.de/files/documents/BFO_Nordsee_2016_2017.pdf, Accessed date: 1 August 2018.
Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L., Thomas, L., 2001. Introduction to Distance Sampling: Estimating Abundance of Biological Populations. Oxford University Press, New York.
Buckland, S.T., Burt, M.L., Rexstad, E.A., Mellor, M., Williams, A.E., Woodward, R., 2012. Aerial surveys of seabirds: the advent of digital methods. J. Appl. Ecol. 49, 960–967.
Buckland, S.T., Rexstad, E.A., Marques, T.A., Oedekoven, C.S., 2015. Distance Sampling: Methods and Applications. Springer International Publishing, Switzerland.
Busch, M., Kannen, A., Garthe, S., Jessopp, M., 2013. Consequences of a cumulative perspective on marine environmental impacts: offshore wind farming and seabirds at North Sea scale in context of the EU marine strategy framework directive. Ocean Coast. Manag. 71, 213–224.
Carpenter, J.R., Merckelbach, L., Callies, U., Clark, S., Gaslikova, L., Baschek, B., 2016. Potential impacts of offshore wind farms on North Sea stratification. PloS One 11, e0160830. https://doi.org/10.1371/journal.pone.0160830.
Christensen, T.K., Clausager, I., Petersen, I.K., 2003. Base-line Investigations of Birds in Relation to an Offshore Wind Farm at Horns Rev, and Results from the Year of Construction. NERI Report. Department of Bioscience, Aarhus University.
Coulson, J.C., Monaghan, P., Butterfield, J., Duncan, N., Thomas, C., Shedden, C., 1983. Seasonal changes in the herring gull in Britain: weight, moult and mortality. Ardea 71, 235–244.
Desholm, M., 2009. Avian sensitivity to mortality: prioritising migratory bird species for assessment at proposed wind farms. J. Environ. Manag. 90, 2672–2679.
Diederichs, 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., Garthe, S., Mendel, B., 2006. Possible conflicts between offshore wind farms and seabirds in the German sectors of North Sea and Baltic Sea. In: Köller, J., Köppel, H., Peters, W. (Eds.), Offshore wind energy. Research on environmental impacts. Springer, pp. 121–143.
Dierschke, V., Exo, K.M., Mendel, B., Garthe, S., 2012. Gefährdung von Sterntaucher *Gavia stellata* und Prachttaucher *G. arctica* in Brut-, Zug- und Überwinterungsgebieten–eine Übersicht mit Schwerpunkt auf den deutschen Meeresgebieten. Vogelwelt 133, 163–194.
Dierschke, V., Furness, R.W., Garthe, S., 2016. Seabirds and offshore wind farms in European waters: avoidance and attraction. Biol. Conserv. 202, 59–68.
Dierschke, V., Furness, R.W., Gray, C.E., Petersen, I.K., Schmutz, J., Zydelis, R., Daunt, F., 2017. Possible Behavioural, Energetic and Demographic Effects of Displacement of Red-throated Divers. JNCC Report 605, U.K. Joint Nature Conservation Committee, Peterborough.
Dierschke, V., Hüppop, O., Garthe, S., 2003. Populationsbiologische Schwellen der Unzulässigkeit für Beeinträchtigungen der Meeresumwelt am Beispiel der in der deutschen Nord- und Ostsee vorkommenden Vogelarten. Seevögel 24, 61–72.
Drewitt, A.L., Langston, R.H., 2006. Assessing the impacts of wind farms on birds. Ibis 148, 29–42.
Emeis, K.-C., van Beusekom, J., Callies, U., Ebinghaus, R., Kannen, A., Kraus, G., Kröncke, I., Lenhart, H., Lorkowski, I., Matthias, V., Möllmann, C., Pätsch, J., Scharfe, M., Thomas, H., Weisse, R., Zorita, E., 2015. the North sea — a shelf sea in the anthropocene. J. Mar. Syst. 141, 18–33.
Environmental Systems Research Institute (ESRI), 2016. ArcGIS V. 10.3 ESRI. Redlands, California.
Exo, K.M., Hüppop, O., Garthe, S., 2003. Birds and offshore wind farms: a hot topic in marine ecology. Wader Study Group Bull. 100, 50–53.
Field, A., Miles, J., Field, Z., 2012. Discovering Statistics Using R. SAGE Publications Ltd.
Fox, A.D., Desholm, M., Kahlert, J., Christensen, T.K., Petersen, I.K., 2006. Information needs to support environmental impact assessment of the effects of European marine offshore wind farms on birds. Ibis 148, 129–144.
Fox, A.D., Petersen, I.K., 2006. Assessing the degree of habitat loss to marine birds from the development of offshore wind farms. In: Boere, G.C., Galbraith, C.A., Stroud, D. (Eds.), Waterbirds Around the World. Edinburgh Stationery Office pp. 804–804.
Furness, R.W., Wade, H.M., Masden, E.A., 2013. Assessing vulnerability of marine bird populations to offshore wind farms. J. Environ. Manag. 119, 56–66.
Garthe, S., 1997. Influence of hydrography, fishing activity, and colony location on summer seabird distribution in the south-eastern North Sea. ICES J. Mar. Sci. 54, 566–577.
Garthe, S., Hüppop, O., Weichler, T., 2002. Anleitung zur Erfassung von Seevögeln auf See von Schiffen. Seevögel 23, 47–55.
Garthe, S., Hüppop, O., 2004. Scaling possible adverse effects of marine wind farms on seabirds: developing and applying a vulnerability index. J. Appl. Ecol. 41, 724–734.
Garthe, S., Markones, N., Mendel, B., Sonntag, N., Krause, J., 2012. Protected areas for seabirds in German offshore waters: designation, retrospective consideration and current perspectives. Biol. Conserv. 156, 126–135.
Garthe, S., Schwemmer, H., Markones, N., Müller, S., Schwemmer, P., 2015. Verbreitung, Jahresdynamik und Bestandsentwicklung der Seetaucher *Gavia* spec. in der Deutschen Bucht (Nordsee). Vogelwarte 53, 121–138.
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. Vogelwelt 128, 163–178.
Goodale, M.W., Milman, A., 2014. Cumulative adverse effects of offshore wind energy development on wildlife. J. Environ. Plann. Manag. 59, 1–21.
Green, R.E., Langston, R.H.W., McCluskie, A., Sutherland, R., Wilson, J.D., 2016. Lack of sound science in assessing wind farm impacts on seabirds. J. Appl. Ecol. 53, 1635–1641.
Guse, N., Garthe, S., Schirmeister, B., 2009. Diet of red-throated divers *Gavia stellata* reflects the seasonal availability of Atlantic herring *Clupea harengus* in the south-western Baltic Sea. J. Sea Res. 62, 268–275.
Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madi, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R., Watson, R., 2008. A global map of human impact on marine ecosystems. Science 319, 948–952.
Horswill, C., O´Brien, S.H., Robinson, R.A., 2017. Density dependence and marine bird populations: are wind farm assessments precautionary? J. Appl. Ecol. 54, 1406–1414.
Hüppop, O., 1995. Störungsbewertung anhand physiologischer Parameter. Ornithologischer Beobachter 92, 257–268.
Hüppop, O., Garthe, S., Hartwig, E., Walter, U., 1994. Fischerei und Schiffsverkehr: Vorteil oder Problem für See- und Küstenvögel? In: Lozán, J.L., Rachor, E., Reise, K., von Westernhagen, H., Lenz, W. (Eds.), Warnsignale aus dem Wattenmeer. Blackwell-Wissenschaftsverlag, Berlin, pp. 278–285.
King, S., Maclean, I., Norman, T., Prior, A., 2009. Developing Guidance on Ornithological Cumulative Impact Assessment for Offshore Wind Farm Developers. COWRIE Ltd.
Kokonendji, C.C., Demetroi, C.G.B., Dossou-Gbete, S., 2004. Overdispersion and Poisson-Tweedie exponential dispersion models. Monografias del Seminario Matematico Garcia de Galdeano 31, 365–374.
Korner-Nievergelt, F., Roth, T., von Felten, S., Guelat, J., Almasi, B., Korner-Nievergelt, P., 2015. Bayesian Data Analysis in Ecology Using Linear Models with R, BUGS, and Stan. Elsevier.
Langston, R.H.W., 2010. Birds and wind farms: where next? In: BOU Proceedings – Climate Change and Birds, . http://www.bou.org.uk/bouproc#002D;net/ccb/langston.pdf, Accessed date: 1 August 2018.
Leopold, M.F., Boonman, M., Collier, M.P., Davaasuren, N., Fijn, R.C., Gyimesi, A., de Jong, J., Jongbloed, R.H., Jonge Poerink, B., Kleyheeg-Hartman, J.C., Krijgsveld, K.L., Lagerveld, S., Lensink, R., Poot, M.J.M., van der Wal, J.T., Scholl, M., 2014. A First Approach to Deal with Cumulative Effects on Birds and Bats of Offshore Wind Farms and Other Human Activities in the Southern North Sea. Report C166/14. IMARES, Wageningen.
Leopold, M.F., Camphuysen, C.J., Verdaat, H., Dijkman, E.M., Meesters, H.W.G., Aarts, G.M., Poot, M.J.M., Fijn, R., 2010. Local Birds in and Around the Offshore Wind Park Egmond Aan Zee (OWEZ) (T-0 & T-1). Report C034/10. IMARES, Wageningen.
Leopold, M.F., van Bemmelen, R.S.A., Zuur, A.F., 2013. Responses of Local Birds to the Offshore Wind Farms PAWP and OWEZ off the Dutch Mainland Coast. Report C151/12. IMARES, Wageningen.
Lewis, S., Sherratt, T.N., Hamer, K.C., Wanless, S., 2001. Evidence of intra-specific competition for food in a pelagic seabird. Nature 412, 816–819.
Lindeboom, H.J., Kouwenhoven, H.J., Bergman, M.J.N., Bouma, S., Brasseur, S., Daan, R., Fijn, R.C., de Haan, D., Dirksen, S., van Hal, R., Hille Ris Lambers, R., ter Hofstede, R., Krijgsveld, K.L., Leopold, M., Scheidat, M., 2011. Short-term ecological effects of an offshore wind farm in the Dutch coastal zone; a compilation. Environ. Res. Lett. 6, 035101. https://doi.org/10.1088/1748-9326/6/3/035101.
Linden, A., Maentyniemi, S., 2011. Using the negative binomial distribution to model overdispersion in ecological count data. Ecology 92, 1414–1421. https://doi.org/10.1890/10-1831.1.
MacLean, I.M.D., Rehfisch, M.M., Skov, H., Thaxter, C.B., 2013. Evaluating the statistical power of detecting changes in the abundance of seabirds at sea. Ibis 155, 113–126.
Masden, E.A., Haydon, D.T., Fox, A.D., Furness, R.W., Bullman, R., Desholm, M., 2009. Barriers to movement: impacts of wind farms on migrating birds. ICES J. Mar. Sci. 66, 746–753.
Mendel, B., Garthe, S., 2010. Kumulative Auswirkungen von Offshore-Windkraftnutzung und Schiffsverkehr am Beispiel der Seetaucher in der Deutschen Bucht. Coast. Rep. 15, 31–44.
Mendel, B., Kotzerka, J., Sommerfeld, J., Schwemmer, H., Sonntag, N., Garthe, S., 2014. Effects of the alpha ventus offshore test site on distribution patterns, behaviour and flight heights of seabirds. In: Ecological Research at the Offshore Windfarm Alpha Ventus. Springer Fachmedien, Wiesbaden, pp. 95–110.
Mendel, B., Sonntag, N., Wahl, J., Schwemmer, P., Dries, H., Guse, N., Müller, S., Garthe, S., 2008. Profiles of seabirds and waterbirds of the German North and Baltic Seas. Distribution, ecology and sensitivities to human activities within the marine environment. In: Naturschutz und Biologische Vielfalt 61. Bundesamt für Naturschutz, Bonn – Bad Godesberg.
Moksness, E., Dahl, E., Støttrup, J., 2009. Integrated Coastal Zone Management. Wiley-Blackwell, Chichester.
Nagel, T., Chauchat, J., Wirth, A., Bonamy, C., 2018. On the multi-scale interactions between an offshore-wind-turbine wake and the ocean-sediment dynamics in an idealized framework – a numerical investigation. Renew. Energy 115, 783–796.
Nolte, N., 2010. Nutzungsansprüche und Raumordnung auf dem Meer. Hansa Int. Marit. J. 147, 79–83.
O'Brien, S., Wilson, L.J., Webb, A., Cranswick, P.A., 2008. Revised estimate of numbers of wintering red-throated divers *Gavia stellata* in Great Britain. Bird Study 55, 152–160.
O'Brien, S.H., Webb, A., Brewer, M.J., Reid, J.B., 2012. Use of kernel density estimation and maximum curvature to set Marine Protected Area boundaries: identifying a Special Protection Area for wintering red-throated divers in the UK. Biol. Conserv. 156, 15–21.
Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M., Fox, A.D., 2006a. Final Results of Bird Studies at the Offshore Wind Farms at Nysted and Horns Rev, Denmark. NERI Report. Ministry of Environment, Denmark.
Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M., Fox, A.D., 2006b. Final Results of Bird Studies at the Offshore Wind Farms at Nysted and Horns Rev, Denmark. NERI Report. commissioned by DONG Energy and Vattenfall A/S, DK, pp. 166.
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. NERI Report. commissioned by Vattenfall A/S, DK, pp. 166.

Petersen, I.K., Fox, A.D., Kahlert, J., 2008. Waterbird Distribution in and Around the Nysted Offshore Wind Farm, 2007. NERI Report. commissioned by DONG Energy, DK.

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. Aarhus Univ., Aarhus.

R Core Team, 2017. R: a Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/, Accessed date: 1 August 2018.

Rebke, M., 2005. Populationsmodelle zur Abschätzung der Auswirkungen additiver Vogelmortalität an Offshore-Windenergieanlagen. Diploma thesis. Univ. of Bremen.

Rexstad, E., Buckland, S., 2012. Displacement Analysis Boat Surveys Kentish Flats. CREEM, Univ. of, St. Andrews.

Schwarz, C.J., 2014. Analysis of BACI experiments. In: Course Notes for Beginning and Intermediate Statistics, Chapter 12. http://people.stat.sfu.ca/~cschwarz/Stat-650/Notes/PDFbigbook-R/R-part013.pdf, Accessed date: 1 August 2018.

Schwemmer, P., Mendel, B., Sonntag, N., Dierschke, V., Garthe, S., 2011. Effects of ship traffic on seabirds in offshore waters: implications for marine conservation and spatial planning. Ecol. Appl. 21, 1851–1860.

Searle, K.R., Mobbs, D.C., Butler, A., Furness, R.W., Trinder, M.N., Daunt, F., 2017. Finding out the fate of displaced birds. In: CEH Report to Marine Scotland FCR/2015/19.

Skov, H., Durinck, J., Leopold, M.F., Tasker, M.L., 1995. Important Bird Areas for Seabirds in the North Sea Including the Channel and the Kattegat. BirdLife International, Cambridge.

Skov, H., Heinänen, S., Thaxter, C.B., Williams, A.E., Lohier, S., Banks, A.N., 2016. Real-time species distribution models for conservation and management of natural resources in marine environments. Mar. Ecol. Prog. Ser. 542, 221–234.

Skov, H., Prins, E., 2001. Impact of estuarine fronts on the dispersal of piscivorous birds in the German Bight. Mar. Ecol. Prog. Ser. 214, 279–287.

Smith, E.P., 2002. BACI design. In: El-Shaarawi, H., Piegorsch, W.W. (Eds.), Encyclopedia of Environments. Wiley & Sons Ltd., Chichester, pp. 141–148.

Stienen, E., Van Waeyenberge, J., Kuijken, E., Seys, J., 2007. Trapped within the corridor of the Southern North Sea: the potential impact of offshore wind farms on seabirds. In: de Lucas, M., Janss, G.F.E., Ferrer, M. (Eds.), Birds and Wind Farms: Risk Assessment and Mitigation. Quercus, London, pp. 71–80.

Thaxter, C.B., Burton, N.H.K., 2009. High Definition Imagery for Surveying Seabirds and Marine Mammals: a Review of Recent Trials and Development of Protocols. British Trust for Ornithology Report Commissioned by COWRIE Ltd.

Van Bemmelen, R., Geelhoed, S., Leopold, M., 2011. Shortlist Masterplan Wind Ship-based Monitoring of Seabirds and Cetaceans. Report C099/11. IMARES, Wageningen.

Vandendriessche, S., Derweduwen, J., Hostens, K., 2015. Equivocal effects of offshore wind farms in Belgium on soft substrate epibenthos and fish assemblages. Hydrobiologia 756, 19–35.

Vanermen, N., Onkelinx, T., Courtens, W., Verstraete, H., Stienen, E.W., 2015a. Seabird avoidance and attraction at an offshore wind farm in the Belgian part of the North Sea. Hydrobiologia 756, 51–61.

Vanermen, N., Onkelinx, T., Verschelde, P., Courtens, W., Van de walle, M., Verstraete, H., Stienen, E.W.M., 2015b. Assessing seabird displacement at offshore wind farms: power ranges of a monitoring and data handling protocol. Hydrobiologia 756, 51–61.

Welcker, J., Nehls, G., 2016. Displacement of seabirds by an offshore wind farm in the North Sea. Mar. Ecol. Prog. Ser. 173–182.

Winiarski, K., Burt, M., Rexstad, E., Miller, D., Trocki, C., Paton, P., McWilliams, S., 2014. Integrating aerial and ship surveys of marine birds into a combined density surface model: a case study of wintering common loons. Condor 116, 149–161.

Wood, S.N., 2006. Generalized Additive Models: an Introduction with R. Chapman and Hall, London.

Zuur, A.F., 2012. A Beginner's Guide to Generalized Additive Models with R. Highland Statistics Ltd, Newburgh.

Zuur, A.F., Ieno, E., Smith, G.M., 2007. Analysing Ecological Data. Springer Science & Business Media, LLC.

Zuur, A.F., Ieno, E.N., Elphick, C.S., 2010. A protocol for data exploration to avoid common statistical problems. Methods Ecol. Evol. 1, 3–14.

Zuur, A.F., Ieno, E.N., Walker, N.J., Saveliev, A.A., Smith, G.M., 2009. Mixed Effect Models and Extensions in Ecology with R. Springer Science & Business Media, LLC.

Zuur, A.F., Saveliev, A.A., Ieno, E.N., 2012. Zero Inflated Models and Generalized Linear Mixed Models with R. Highland Statistics Ltd, Newburgh.