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Original Article

Looking at the bigger picture: the importance of considering annual cycles in impact assessments illustrated in a migratory seabird species

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Accurate assessment of the overall impacts of anthropogenic activities on mobile, migratory species requires cumulative year-round impact assessments covering their entire annual cycle. This study considers the type of information needed and the assessment tools required to implement such assessments. The developed concept is demonstrated by modelling year-round collisions of Black-legged Kittiwakes (*Rissa tridactyla*) breeding on the German North Sea island of Helgoland with constructed and planned offshore wind farms across the species distributional range, in order to assess the endangerment status of the local population.

Keywords: cumulative impact, kittiwake, offshore wind, year-round impact assessment.

Introduction

Environmental impact assessments (EIAs) aim to determine the potential impacts of anthropogenic developments on the environment (e.g. Wood, 2003; Glasson *et al.*, 2012), e.g. the likely effects of a development on a population of animals whose habitat overlaps or interacts with the development area. Cumulative impact assessments (CIAs) are important aspects of EIAs that consider the potential combined effects of several developments in relatively close proximity to each other (Masden *et al.*, 2010), considering both existing and future projects for which sufficient information is available (Masden *et al.*, 2014). It is important that the spatial scale of CIAs adequately reflects the biological characteristics of the features/species assessed (Masden *et al.*, 2010). However, this may be challenging in the case of mobile and particularly migratory species, for which the biologically appropriate assessment scale varies throughout the annual cycle of the species assessed.

In this context, we demonstrate the need for impact studies that encompass the complete annual cycle, in order to allow estimation and assessment of year-round impacts. We therefore provide conceptual thoughts and consider methods for assessing the cumulative potential threats faced by a species that visits several spatially detached habitats (and politically determined units of sea areas) during their annual cycle. Given that local impacts at the wintering site may be compensated for by favourable foraging and reproduction conditions at the breeding site, knowledge on the wintering conditions and the sensitivity of the population during this stage of its annual cycle are crucial for accurately assessing the potential impact of a development that may interfere with these suitable conditions at the breeding site. A realistic assessment of the capacity of a population to cope with pressures such as additional mortality or habitat loss is thus only possible through awareness of the year-round population context. However, such a synopsis of (sub)population-specific information may be difficult to compile for many species. Ideally, comprehensive tracking data can provide detailed information on the spatial and temporal movements of animals, though such data are missing for most species that reproduce, stage or winter in remote or poorly monitored areas, making potential threats at certain stages of their annual cycle difficult to identify. Nevertheless, the challenges associated with collecting year-round impact assessment data should not reduce efforts to collect it, given that such data represent a precondition for making an informed assessment.

We demonstrate the implementation of such an assessment using the Black-legged Kittiwake (*Rissa tridactyla*, hereafter referred to as kittiwake), a medium-sized, pelagic gull species which has a well-studied distribution across its entire annual cycle (Frederiksen *et al.*, 2012). The winter distribution of regional breeding populations is analysed, and the contributions of several regional sub-populations to the composition of the wintering population are estimated.

Offshore wind farm (OWF) developments impose specific pressures on kittiwakes (collision risk, displacement) (Furness *et al.*, 2013), and comprehensive developments in numerous countries suggest that specific kittiwake sub-populations are likely to interact with different OWFs in several countries during different seasons of their annual cycle.

Empirical evidence for frequent collisions between seabirds and offshore wind turbines is largely missing, probably because collisions within the marine environment are difficult to record (Collier *et al.*, 2011; 2012). However, high collision rates have been documented locally, e.g. in situations where wind turbines were built on breakwaters intercepting the main flight corridors of gulls and terns between their breeding colonies and foraging habitats (Everaert and Stienen, 2006). Gulls and other birds regularly collide with wind turbines at coastal sites (Langgemach and Dürr, 2016), and it is therefore reasonable to assume that seabirds may also collide with turbines at sea, such that local populations may be negatively affected. In this context, the potential impacts due to collision risk with OWFs for the Helgoland kittiwake population, which represents the entire German population, must include estimates for each stage of the species' annual cycle. The isolated location of the Helgoland kittiwake colony in the centre of the German Bight, with several OWFs located at different distances from the island, provides an interesting case study. Clearly, collisions with offshore wind turbines represent only one of several risks introduced by anthropogenic activities (e.g. fisheries, climate change, pollution, displacement), and impact assessments should ideally consider the combined effects of all such risks. However, mortality due to collisions with offshore turbines may represent the single most important new, immediate risk faced by kittiwakes, while other sources of human-caused mortality may, to a certain degree, already be reflected in current adult survival rates and therefore in the current population status (Busch and Garthe, 2016).

Importantly, rather than providing a transferable assessment framework, this study uses kittiwakes as an example to demonstrate the relevance, suitable assessment tools, and type of conclusions derived from year-round assessments, to highlight the need for a holistic approach to impact assessments.

Material and methods

An assessment of year-round collision risk for a defined sub-population requires the combination of an appropriate methodology for estimating collision risk with information on the phenology, distribution, and size of the population being studied.

Band collision risk model

Collision risk modelling is a common analytical step in OWF EIAs and habitat regulation assessments (HRAs). Various collision risk models (CRMs) have been developed and used in the past (e.g. Volkert model (Masden, 2015), for a comprehensive review of CRM approaches see Madsen and Cook, 2016), the CRM developed by Band (2012) in relation to the Strategic Ornithological Support Services programme (SOSS), initiated by The Crown Estate, is the current standard tool for collision risk assessments in the United Kingdom, and is frequently used in other European countries (e.g. Brabant *et al.*, 2015). This so-called Band model is used to derive quantitative estimates of collisions of bird species within operating OWFs. It provides a standardized approach to collision risk assessment, thus enabling comparisons of collision estimates from different OWFs (Band, 2012). The latest update of the Band model developed its application using a simulation approach to incorporate the variability and uncertainty associated with several input parameters, making it possible to derive average collision estimates with associated confidence intervals (Masden, 2015), using the mean and SD of the input parameters rather than the absolute values, where available. This model update has been used to derive collision estimates for the OWFs assessed.

Bird collision risk can be estimated when bird-related parameters (e.g. body length, flight speed, nocturnal activity, etc.) are combined with technical turbine specifications (e.g. turbine dimensions, rotation speed, etc.) and information on the number of turbines in the CRM. The CRM parameters used are summarized in Tables 1 and 2. Information on monthly kittiwake densities within the boat-survey areas comprising the OWFs in the vicinity of Helgoland are based on environmental impact studies focusing on the third investigation year prior to construction (Braasch *et al.*, 2011; Schuchardt *et al.*, 2011; Piper *et al.*, 2012). However, the OWF abundance data were collected between 2010 and 2012, and were published with no information on the uncertainty of the values.

Discussions are taking place between the OWF industry and statutory nature conservation bodies to ensure that nature conservation legislation is accounted for within EIA and HRA processes, in relation to several key parameters of the CRM. Notably, several different opinions exist regarding the avoidance rate, which strongly influences the final collision estimate. Brabant *et al.* (2015) pointed out that the number of estimated collision victims is proportional to the percentage of birds that fails to implement avoidance actions, making this value a key parameter in the CRM. The flight speed of the assessed species also has a strong influence on the calculated flux of birds through an OWF, with a higher flux increasing the expected number of potential collision victims (Krijgsveld *et al.*, 2009).

The assessment in this article follows the current advice of UK statutory nature conservation bodies, which recommend using an avoidance rate of 98.9% (±2 SD) for kittiwakes, and Option 1 of the Band model as the default (SNCBs, 2014). Option 1 of the model assumes a uniform distribution of flight heights between the lowest and highest level of the rotor for the proportion of birds identified to fly at risk height. In contrast, Option 2 uses the proportion of birds at risk height based on generic flight height information, also assuming a uniform distribution of flights across the collision risk window. Site-specific data regarding the proportion of birds at risk height should be used if available, but this information is currently not available for the German OWFs, and a generic value (15.7%) based on data from 25 sites in the United Kingdom, the Netherlands, and Belgium was therefore used (Cook *et al.*, 2012). Although Johnston *et al.* (2014) recently

Table 1. Bird-related parameters used in the updated Band CRM (Masden, 2015).

Species	Avoidance rate	Body length (m)	Wingspan (m)	Flight speed (m/s)	Nocturnal activity	Flight	Proportion of flights at potential collision height
Black-legged Kittiwake (*R. tridactyla*)	Mean = 0.989; SD = 0.002[a]	Mean = 0.39[b]; SD = 0.005[c]	Mean = 1.08[d]; SD = 0.04[e]	Mean = 7.26; SD = 1.5[f]	Mean = 0.033; SD = 0.0045[g]	Flapping[h]	Mean = 0.157[i]; SD = 0.0536[j]

Mean and SD of input parameters were used to capture uncertainty within the data where available.
[a]Recommended by UK statutory nature conservation bodies (SNCBs, 2014).
[b]BTO (2015).
[c]Masden (2015).
[d]BTO (2015).
[e]Masden (2015).
[f]Masden (2015).
[g]Masden (2015).
[h]Applied in similar assessments e.g. Dong Energy (2013).
[i]Cook *et al.* (2012).
[j]Approximation of SD.

Table 2. Wind farm and turbine parameters used in the updated Band CRM (Masden, 2015).

Name of OWF	Meerwind Süd/Ost	Nordsee Ost	Amrumbank West
Latitude	54.392° [a]	54.444° [a]	54.522° [a]
Tidal offset	2.0 m[b]	2.0 m[b]	2.0 m[b]
Target power	288 MW (288 MW/3.6 = 80 turbines)[c]	295.2 MW (295.2 MW/6.15 MW = 48 turbines[a]	288 MW (288/3.6 = 80 turbines[d]
Turbine model	Siemens SWT-3.6-120	Senvion 6.2M 126	Siemens SWT-3.6-120
Hub height	89 m[c]	96.5 m[a]	89 m[c]
Rotor radius	60 m[c] (diameter 120 m)	63 m[a] (diameter 126 m)	60 m[c] (diameter 120 m)
Pitch	15° [e]	15° [e]	15° [e]
Max. blade width	4.2 m[f]	4.2 m[h]	4.2 m[a]

[a]OWF database: www.4coffshore.com.
[b]Difference between highest astronomical tide and mean sea level: www.fino3.de/fino3/hydrologie.
[c]Meerwind Süd/Ost developer webpage: www.windmw.de/meerwind.html.
[d]http://www.eon.com/content/dam/eon-com/en/downloads/1/111216_Factsheet_Amrumbank_dt.pdf.
[e]Recommended average value (Band, 2012).
[f]Figure used in another CRM for Siemens SWT-3.6-120 (Dong Energy, 2013).
[g]Calculated based on mean wind speed data for 2013 and 2014 from Helgoland survey station [DWD Climate Data Center (CDC), 2015].
[h]Value represents an estimate, as no information available from OWF developer or turbine manufacturer.

provided modelled flight height distribution data, the model validation for gulls and kittiwakes in particular was poor (<50%). We therefore opted to use the generic value of Cook *et al.* (2012) in order to remain precautionary and to allow the results to be compared with modelling results for other OWFs, which were mainly calculated before the Band model update by Masden (2015). Option 3 of the model, which uses generic flight height information and does not assume uniform distribution of flight heights across the collision risk window, was not considered appropriate for assessing kittiwakes because of uncertainties about the modelled flight height distribution of the species, with could lead to underestimation of the potential collision risk (SNCBs, 2014).

In addition to site-specific information on bird abundance, biometric and behavioural aspects of the respective species, as well as technical information about the wind turbine type assessed, the operational time, defined as the proportion of time a turbine is rotating, needs to be calculated. Operational time is affected by two features: the proportion of time the wind speed is above the cut-in and below the cut-out wind speed for the respective turbine type, and the time required for maintenance activities (when the turbines are shut down, e.g. for repair etc.). Operational time and rotor rotation speed were calculated based on wind speed data collected (averages based on data from 2013 and 2014) at a weather station on the island of Helgoland (DWD Climate Data Center (CDC), 2015) located 20–40 km from the assessed OWFs. The down-time for operation and maintenance of the OWFs was also considered; according to the European Environment Agency (EEA, 2009), this takes place during about 10% of the year, with increased access for maintenance activities during the summer months due to the wind conditions (Band, 2012). This was reflected by setting the availability factor to 80% from May to the end of August, and 90% for the remaining months of the year. All parameters informing the CRM are provided in Tables 1 and 2.

Definition of kittiwake bio-seasons for assessment

Defining the different stages of the kittiwakes' annual cycle is a precondition for the implementation of a year-round impact assessment. About 70% of the birds breeding in the North Sea region also spend the winter in the North Sea (Frederiksen *et al.*, 2012), and there is thus no distinct migration season. Individuals breeding in Helgoland therefore tend to disperse across the North Sea in winter, and the year can simply be divided into a colony-attendance season (Busch and Garthe, 2016) and a non-breeding season for the Helgoland kittiwake population. Based on colony-specific data (Dierschke *et al.*, 2001, 2016), the colony-attendance season can be defined as roughly from February to August, with the non-breeding season covering the remaining months from September to the end of January.

Assessing collision risk during colony-attendance season

Kittiwakes are central-place foragers during the colony-attendance season (Baird, 1991; Burke and Montevecchi, 2009), and only OWFs within the colony-specific foraging range will therefore impose a potential threat to the Helgoland population during this period. Thaxter *et al.* (2012) reported a mean foraging range of 24.8 km, with a mean maximum of 60 km and a maximum of 120 km for kittiwakes, based on pooled data from 216 tracked individuals at seven colonies. Kittiwakes were among the species with the highest confidence in foraging range in the study by Thaxter *et al.* (2012), which assessed various seabird species. For the Helgoland population, Dierschke *et al.* (2004) calculated the concentration of kittiwakes within a 35 km radius of the island based on boat surveys, and a single aerial survey on 7 June 2003 found that the majority of kittiwakes were located within only 10 km of the colony.

There are three OWFs (the so-called Helgoland Cluster) within an $\sim$40 km radius around Helgoland: Meerwind Süd/Ost (24 km), Nordsee Ost (30 km), and Amrumbank West (35 km), all of which are located north-west of the island (see Figure 1). However, more OWFs are to be constructed within the mean maximum foraging range reported by Thaxter *et al.* (2012) (60 km) in the near future [Nordergründe (construction planned to start 2016), Gode Wind I, II, VI (Gode Wind under construction), Area C III (consent application submitted)].

Several studies have confirmed that seabird foraging ranges are related to colony size, with breeding adults from smaller colonies having shorter foraging distances than birds from larger colonies (Ainley *et al.*, 2003; Hemerik *et al.*, 2014), as a consequence of density-dependent intraspecific competition for food (Lewis *et al.*, 2001; Ford *et al.*, 2007). Birds breeding in island colonies typically forage closer to the colony because they can access much larger marine foraging areas than birds from coastal colonies within short distances, due to the ability to forage in a 360° angle around their breeding site. The above evidence together with colony-specific information on the foraging range of kittiwakes from Helgoland suggest that collision risk during the colony-attendance season will be restricted to the three OWFs in the Helgoland Cluster, and only these three OWFs are likely to pose any threat to the Helgoland breeding birds during this period.

There are no other kittiwake colonies in the vicinity of Helgoland, and these birds therefore do not share their foraging areas with individuals from other colonies. There is thus no need to apportion collisions during the colony-attendance season within the OWF cluster among different kittiwake colonies.

Assessing winter collision risk

Improved knowledge of the composition of meta-populations, migration, and the distribution of restricted regional populations during different seasons, as well as seasonal abundances within season-specific habitats, represent preconditions for well-informed year-round impact assessments. Combination of this knowledge with structured and pooled information on the potential threats/anthropogenic sources of mortality for restricted populations allow the potential impacts at each stage of the annual cycle of a species to be assessed and estimated.

The contribution of the Helgoland kittiwakes to the overall breeding population of the North Sea can be estimated based on the size of the Helgoland colony in relation to the overall North Sea population. Kittiwake densities for the OWFs of the Helgoland Cluster were available for 2010–2012, and the mean kittiwake population for the same period was therefore used for collision risk assessment. The mean population (2010–2012) of 6372 pairs (Inst. Avian Biol., J. Dierschke, pers. comm.) represents 12 744 breeding birds.

According to Wetlands International (2006), a factor of 1.5 can be used to scale up from a population of breeding adults to estimate the overall population size, including immature birds. Applying this rule suggests that $\sim$19 116 birds were associated with the Helgoland colony during this time frame. Given that the regional North Sea population was estimated to comprise 311 290 apparently occupied nests (AONs) (Frederiksen *et al.*, 2012) translating to 933 879 individuals [$(\text{AON} \times 2) \times 1.5$], the Helgoland colony may account for about 2.05% of the North Sea population.

The kittiwake population on Helgoland, in line with most North Sea colonies, has declined considerably in recent years (see Figure 2). Kittiwake populations in the United Kingdom, which supports the majority of breeding North Sea kittiwakes, declined by 63% between 1986 and 2014, and by 47% between 2000 and 2014 (Hayhow *et al.*, 2015). The population on Helgoland showed an opposing trend, increasing until about 2000, peaking in 2001 (8600 breeding pairs), and remaining relatively stable until 2010; however, the Helgoland population has also declined considerably from 2011 onwards (Inst. Avian Biol., J. Dierschke, pers. comm.; Dierschke *et al.*, 2001). The North Sea sandeel fishery and a tendency towards warmer winters, which negatively affect the reproduction of sandeels, are thought to be the drivers of population declines across the North Sea, where kittiwakes are almost completely dependent on sandeels during the breeding season (Frederiksen *et al.*, 2004; Mitchell *et al.*, 2004).

According to Frederiksen *et al.* (2012), $\sim$70% of the North Sea breeding kittiwake population does not cross to the western Atlantic, and most of these birds remain in the North Sea throughout the winter, with only a minority moving to the Celtic–Biscay Shelf. A precautionary assessment would therefore assume that about 70% of the Helgoland kittiwakes remain in the North Sea, representing 13 381 individuals ($19\,116 \times 0.7$). Considering that 225 261 adult kittiwakes are estimated to spend the winter in the North Sea [modelled for December 2009 (Frederiksen *et al.*, 2012)], Helgoland birds would represent 6% [$(13\,381/225\,261)\times 100$] of this North Sea winter population.

This knowledge of the composition of the seasonal kittiwake population in the North Sea, combined with information on the cumulative modelled collisions with offshore turbines, would allow kittiwake collisions to be apportioned to specific sub-populations (e.g. Helgoland) based on the relative contributions of the individual colonies to the overall non-breeding season North Sea population, assuming an even mixing of those birds spending the winter in the North Sea.

Information on non-breeding season kittiwake collisions are available for most UK North Sea OWFs, but this information is missing for most OWFs operated in other North Sea riparian states territorial waters, where collision risk modelling does not represent a standard/obligatory EIA assessment step.

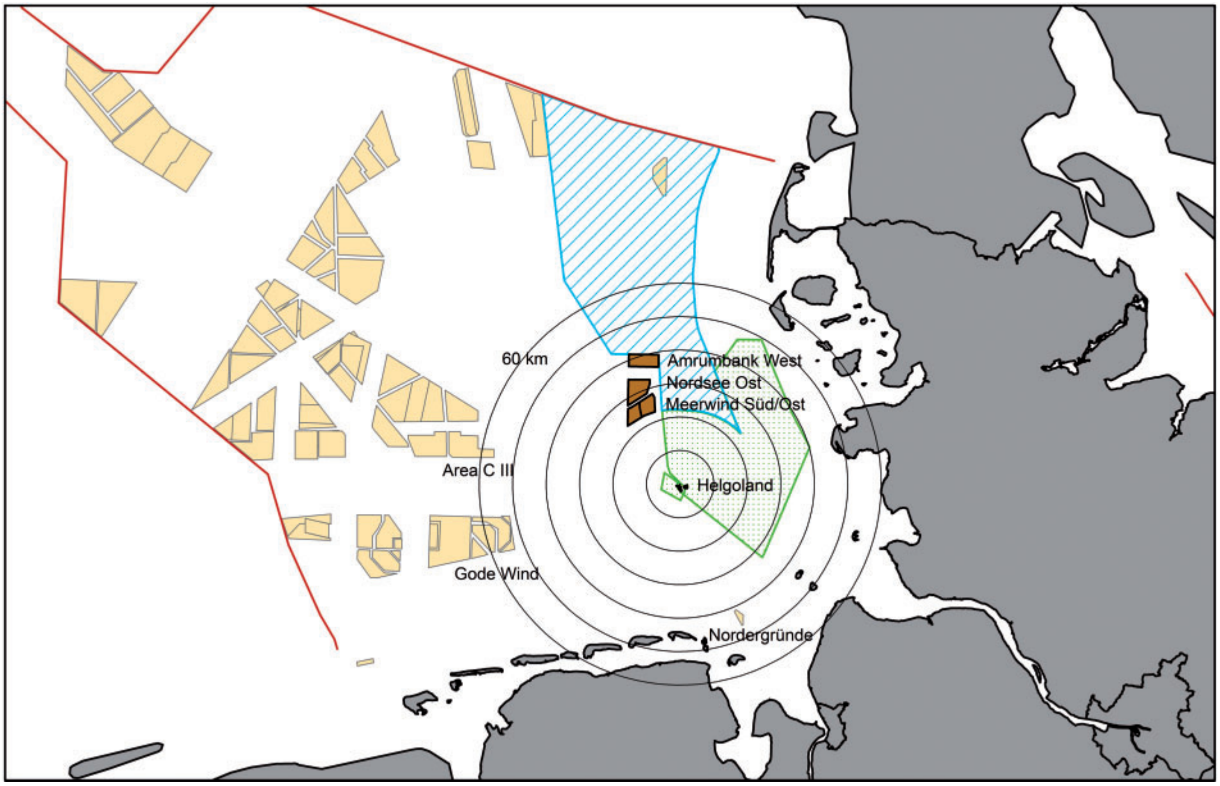

Figure 1. Location of Helgoland and relevant OWFs in the German Bight. The hatched area represents the Special Protection Area (SPA) “Eastern German Bight” and the dotted area the SPA “Helgoland Seabird Protection Area”.

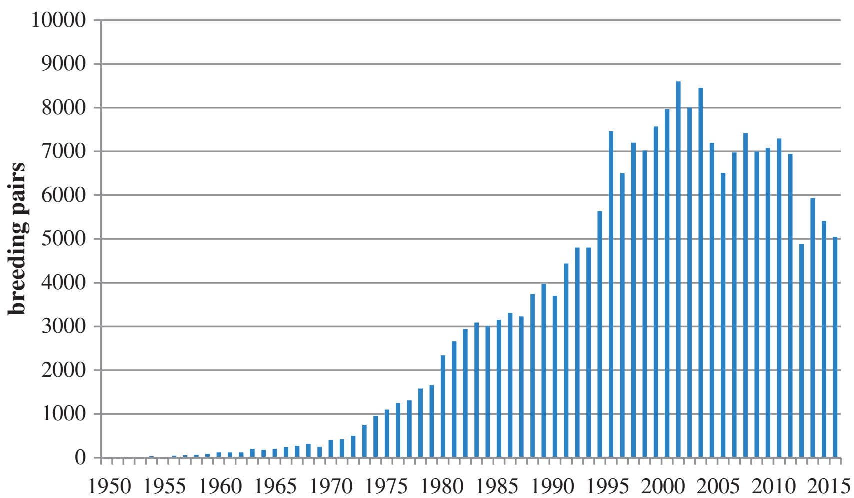

Figure 2. Development of kittiwake colony size on Helgoland (Inst. Avian Biol., J. Dierschke, pers. comm.).

Potential biological removal

The potential biological removal (PBR) approach (Wade, 1998) is a simple population modelling approach for estimating the number of fatalities (above natural mortality) that a defined population can sustain each year. In this assessment, PBR analysis was used to assess the cumulative year-round collision estimate in the context of the potential of the Helgoland colony to sustain those victims. PBR has become a common tool for assessing the potential impacts of OWFs on seabirds, especially in the United Kingdom, and detailed descriptions of the methodology, strengths, and weaknesses of the approach can be found in Dillingham and Fletcher (2008) or Busch and Garthe (2016). The key parameters used to calculate the PBR value for the Helgoland kittiwake population are summarized in Table 3. The mean population for the period 2010–2012 was used as the reference population, and OWF-specific abundance data were available for this period.

Results

We estimated the year-round collision risk for kittiwakes from Helgoland by performing collision risk assessments for both

Table 3. Parameters used for PBR analysis.

Reference population	Age of first breeding	Adult survival rate	N_{min} (conservative population size estimate)	R_{max} (max. recruitment rate)	Measurement error
19 116 individuals	4 years (BTO, 2015)	0.882 (Harris *et al.*, 2000)	15 487 individuals	0.1331	25% error and the lower bound of a 60% CI

Table 4. Estimated kittiwake collisions and SD at Helgoland Cluster OWFs.

Month	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total (annual)	Total (Feb–Aug)
Meerwind Süd/Ost														
Mean	1.6	2.7	5.2	4.6	2.6	4.1	3.2	0.3	0.0	0.0	1.7	0.2	26	23
SD	0.7	1.2	2.3	2.0	1.2	1.8	1.4	0.1	0.0	0.0	0.8	0.1		
Nordsee Ost														
Mean	1.2	2.2	7.4	0.8	2.4	1.5	0.7	0.3	0.1	0.1	0.9	0.4	18	15
SD	0.5	0.9	3.1	0.3	1.0	0.6	0.3	0.1	0.0	0.0	0.4	0.2		
Amrumbank West														
Mean	3.9	3.6	11.9	0.9	1.3	1.8	0.0	0.0	0.1	1.4	0.6	1.2	27	20
SD	1.8	1.6	5.3	0.4	0.6	0.8	0.0	0.0	0.0	0.6	0.3	0.5		
Cumulative collision estimate for Helgoland OWF Cluster														
Mean	6.7	8.5	24.4	6.3	6.4	7.4	4.0	0.6	0.1	1.4	3.2	1.8	71	58

Highlighted months indicate the colony-attendance season.

partial habitats occupied by this sub-population over the course of a year.

Collision risk assessment in relation to the Helgoland OWF cluster

The CRM (Masden, 2015) for the OWFs of the Helgoland Cluster using the recommended avoidance rate of 98.9% estimated 71 annual kittiwake collisions. The influence of the avoidance rate on the model output is illustrated by the collision estimates based on a range of avoidance rates presented in Table 4. As described in "Assessing collision risk during colony-attendance season" section, all collisions during the colony-attendance period can be associated with the Helgoland colony, because there are no other breeding sites in the foraging range. Accordingly, collisions modelled to occur between February and the end of August will exclusively impact the Helgoland population. A total of 58 kittiwakes per year were estimated to collide with turbine blades during this period.

Non-breeding season collision estimate

Modelled collision estimates from all UK North Sea OWFs were acquired for the non-breeding season. Data were compiled according to the HRA assessment for the Dogger Bank Teesside A & B OWF project (Forewind, 2014) using the Band (2012) CRM. However, because the Forewind (2014) collision estimates were based on an avoidance rate of 99%, the figures were adjusted to the recommended avoidance rate of 98.9%. Collision figures for the only additional proposed OWF not specified in the Forewind (2014) listing, the planned OWF Hornsea Project 2, were not added to Table 5 because no collisions are estimated for this site during the period from September to the end of January. All operational, consented, and planned OWFs for which collision estimates exist were considered in the assessment.

A cumulative total of 1620 collisions (at 98.9% avoidance rate) of North Sea kittiwakes with UK North Sea OWFs are estimated for the non-breeding season defined for the Helgoland kittiwake population (beginning of September to end of January). Based on the contribution of the Helgoland birds to the overall North Sea winter population (Frederiksen *et al.*, 2012), 6% of these collisions can be apportioned to kittiwakes originating from Helgoland (see "Assessing winter collision risk" section), suggesting that about 97 individuals from the Helgoland population are likely to collide with these UK OWFs (see Table 5), where CRM results are based on site-specific bird abundance data.

Numerous additional OWFs in German, Dutch, Belgian, and Danish North Sea waters also need to be considered within a comprehensive assessment, but because CRM is not required for EIAs in all North Sea riparian states, collision risk estimates are lacking for most of these sites. To obtain a rough estimate of collision mortality at non-UK OWFs during the non-breeding season, kittiwake collisions per year and per turbine were estimated by extrapolation of UK figures to other North Sea riparian states, based on the number of operating, consented, and planned offshore turbines.

Information on the number of operating, consented, and planned offshore wind turbines were collated from an online OWF database (http://www.4coffshore.com). The considered UK North Sea OWFs comprised roughly 3471 turbines. The modelled 1620 non-breeding kittiwake collisions represented 0.47 (1620/3471) non-breeding season collisions per year per turbine. Belgium, the Netherlands, Germany, and Denmark currently have a further 7014 operational, consented, or planned offshore wind turbines in the North Sea (as of April 2016). Extrapolating the non-breeding season collisions per year per turbine modelled for UK North Sea OWFs suggests that these non-UK turbines would account for an additional 3274 kittiwake collisions, resulting in a total of 4894 kittiwakes colliding with 10 485 offshore wind turbines across the North Sea during the non-breeding season, which figure needs to be incorporated in any year-round impact assessment. Based on the calculated 6% contribution of Helgoland kittiwakes to the North Sea wintering population, an estimated 294 Helgoland

Table 5. Overview of modelled annual kittiwake collisions at UK North Sea OWFs, based on Forewind (2014).

Name	Number of turbines	Modelled monthly kittiwake collisions											
		Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Greater Gabbard	140	0	2.222	6.281	2.937	0.396	0.044	0.616	0.605	0.869	0	2.189	11.341
Gunfleet	48	0	0	0	0	0	0	0	0	0	0	0	0
Kentish Flats	30	0	0	0	0	0	0	0	0	0	0	0	0
Lincs	75	0.231	0.231	0.231	0.231	0.231	0.231	0.231	0.231	0.231	0.231	0.231	0.231
London Array	175	0.462	0.462	0.462	0.462	0.462	0.462	0.462	0.462	0.462	0.462	0.462	0.462
Lynn and Inner Dowsing	54	0	0	0	0	0	0	0	0	0	0	0	0
Scroby Sands	30	0	0	0	0	0	0	0	0	0	0	0	0
Sheringham Shoal	88	0	0	0	0	0	0	0	0	0	0	0	0
Thanet	100	0.088	0.088	0.088	0.088	0.088	0.088	0.088	0.088	0.088	0.088	0.088	0.088
Teesside	27	0.44	0.66	1.43	12.43	3.41	22.11	12.87	11.77	10.67	0.55	0.66	0.33
Humber Gateway	73	0.638	0.638	0.638	0.638	0.638	0.638	0.638	0.638	0.638	0.638	0.638	0.638
Westermost Rough	35	0.044	0.044	0.044	0.044	0.044	0.044	0.044	0.044	0.044	0.044	0.044	0.044
Blyth Demonstration	2	0.462	0.462	0.462	0.462	0.462	0.462	0.462	0.462	0.462	0.462	0.462	0.462
Dudgeon	67	0	0	0	0	0	0	0	0	0	0	0	0
EOWDC (Aberdeen OWF)	11	0.44	0.033	0	0.616	1.309	5.478	4.972	2.365	0.913	2.101	0.473	0
Galloper	56	6.93	7.205	12.76	4.95	4.29	0.495	1.485	1.1	0.22	0.55	1.21	24.695
Race Bank	91	4.059	1.023	0.506	0	0.506	0	1.353	7.117	6.105	4.576	0	6.105
Triton Knoll	288	8.481	4.235	32.67	4.862	3.366	1.122	15.29	37.29	54.989	18.7	17.952	9.977
Beatrice	2	2.849	5.434	4.367	27.137	51.095	32.494	11.11	2.233	1.089	6.105	0	1.32
Moray	186	0.044	1.056	10.274	23.628	28.908	10.901	3.806	2.288	1.144	0.341	0.077	0.022
East Anglia ONE	102	32.186	37.554	34.87	0	26.829	0	2.684	2.684	2.684	13.42	182.38	93.885
Dogger Bank Creyke Beck	200	56.276	88.77	149.93	67.386	86.834	80.476	53.57	28.16	14.685	33.55	27.907	30.8
Dogger Bank Teesside A & B	360	28.787	59.873	128.26	39.655	44.297	31.834	21.12	11.22	10.714	22.55	17.402	28.809
Dogger Bank Teesside C & D	520	16.401	28.809	69.069	37.906	45.067	31.845	19.69	11.44	14.168	18.48	12.243	14.476
Firth of Forth Alpha	75	37.246	37.246	37.246	22	22	22	22	22	37.246	37.246	37.246	37.246
Firth of Forth Bravo	75	28.281	28.281	28.281	29.04	29.04	29.04	29.04	29.04	28.281	28.27	28.281	28.281
Hornsea Project One	171	6.05	6.05	8.8	3.85	1.65	15.4	26.95	14.3	14.85	6.6	14.3	3.85
Inch Cape	110	14.3	3.355	44.858	0.99	1.694	4.048	7.37	0.902	104.016	66.22	29.403	24.266
Neart na Gaoithe	64	0.55	0.55	0.55	2.75	9.295	3.575	20.02	1.65	9.35	11	1.65	33
Navitus	100	6.6	8.8	0.55	1.65	0.55	1.1	1.1	0.55	0	2.2	6.05	9.35
Rampion	116	12.375	13.332	2.552	1.474	0.385	51.7	2.343	11.308	0.396	9.35	4.906	11.462
total	3471	264.22								314.314	283.734	386.254	371.14
Overall non-breeding season collisions	**1620**												

Modelled collisions presented at 99% avoidance rate by Forewind (2014) were transformed to 98.9% avoidance rate. Months highlighted in grey indicate the non-breeding season (Sep. to Jan.). Where a range of turbine numbers was given for a development, an average value was used for the assessment. Operating, consented, and planned OWF projects were considered. Source for information for number of turbines per OWF: www.4coffshore.com.

Table 6. Extrapolating non-breeding season kittiwake collisions to North Sea scale based on a non-breeding season collision per turbine per year factor of 0.47 derived from CRM results for UK North Sea OWFs.

Country	Number of operating, consented and planned turbines	Number of non-breeding season kittiwake collisions
United Kingdom	3471	1620
Belgium	492	230
Netherlands	703	328
Germany	5512	2573
Denmark	307	143
Total	**10 485**	**4894**
Total except United Kingdom	**7014**	**3274**
Kittiwakes Helgoland colony (overall)		**294**
Kittiwakes Helgoland colony (except United Kingdom)		**196**

Operating, consented, and planned OWF projects across North Sea riparian states were considered. The apportionment of kittiwake collisions to the Helgoland colony was made assuming a 6% contribution to the overall North Sea non-breeding season population. Source of information for number of turbines per OWF: www.4coffshore.com.

Table 7. Year-round collision estimates for kittiwakes originating from the colony on Helgoland.

Season	Collision estimate	Comment
Colony-attendance season	58	
Non-breeding	97	Kittiwakes Helgoland colony (United Kingdom only)
Non-breeding	294	Kittiwakes Helgoland colony (entire North Sea)
Year-round estimate (modelled)	155	Estimate based on own and available modelling results based on site-specific data
Year-round estimate (extrapolated)	352	Estimate based on extrapolating impacts modelled for UK North Sea OWFs to other North Sea riparian states

kittiwakes per year may collide with OWF turbines during the non-breeding season, if the development plans of the North Sea wind industry are realized (see Table 6).

Year-round collision estimate

Summarizing the collision figures in "Collision risk assessment in relation to the helgoland OWF cluster" and "Non-breeding season collision estimate" sections produced an annual year-round collision estimate of 155 kittiwakes originating from the Helgoland colony, considering modelled breeding season and non-breeding season collisions at UK North Sea OWFs (see Table 7).

Extrapolating the non-breeding season collision estimate to the North Sea scale as described in "Non-breeding season collision estimate" section, suggest that there could be 352 collisions of Helgoland kittiwakes annually if the OWF development intentions of the North Sea riparian states are realized. Although this figure should be treated with caution, it may represent a rough estimate of the magnitude of the impact (see Table 7).

Collision estimate in context of PBR

Kittiwakes breeding on Helgoland represent a protected, designated SPA population that has shown considerable declines in recent years, suggesting that a precautionary f value, which reflects the recovery factor, should be applied for PBR modelling. According to Dillingham ans Fletcher (2008), an f value of 0.1 can be used to ensure a minimal increase in recovery time for a depleted population, maintain a population close to the carrying capacity, or minimize the extinction risk for a population with limited range. An f value of 0.1 should thus allow a population to develop as under a "no harvest" scenario, with minimal delay. Applying the Dillingham and Flechter (2008) criteria to the Helgoland population, a precautionary f value of 0.1 appears appropriate and advisable, given that recent declines may indicate that the population is above the carrying capacity of the surrounding habitat. Furthermore, the colony is a relatively isolated population and represents the only German breeding site, such that disappearance of the Helgoland colony would lead to the extinction of kittiwakes in Germany and a considerable range reduction of the species in the North Sea region. PBR results for the Helgoland kittiwake population (breeding adults and non-breeders associated with the colony) are presented in Table 8. In addition to the results based on an f value of 0.1, considered appropriate for this assessment, PBR values for $f = 0.2$ and $f = 0.3$ are also presented. A value of $f = 0.3$ is recommended as suitable for healthy protected populations (Busch and Garthe, 2016).

Comparing year-round collision estimates ("Year-round collision estimate" section) and PBR values for different recovery factors indicate that the Helgoland kittiwake population is likely to be under considerable unsustainable pressure from collisions with OWFs. The colony-attendance season collision estimates (58 individuals) account for more than half the annual PBR value based on the precautionary but advisable recovery factor of $f = 0.1$ (103 individuals). The total of non-breeding season collisions estimated to occur at UK North Sea OWFs plus the colony-attendance season estimate clearly exceeds the value that the local population is likely to be able to sustain (155 individuals). Extrapolating the average collision risk modelled for UK North Sea OWFs to OWF projects across the entire North Sea further increases the potential year-round collision estimate. The extrapolated collision estimate of 352 annual victims at a North Sea scale not only exceeds the less precautionary PBR value based on a recovery factor of $f = 0.2$ (206 individuals), but also considerably exceeds the PBR value based on a recovery factor of $f = 0.3$, and is more than three times higher than the PBR value at $f = 0.1$, which is considered sustainable from a conservative conservation perspective.

Discussion

This assessment illustrates how year-round impact assessments should be structured to allow the potential impacts on mobile species moving through different habitats over the course of a year to be assessed comprehensively.

Notably, cumulative, year-round collision estimates should not be considered as absolute figures, and the fact that the calculations within CRMs are based on several assumptions [e.g. avoidance rate, percentage of flights at collision risk height, assumptions on operational time of OWFs, etc., as discussed in detail by Green *et al.* (2016)], each with the potential to influence the model output, must be taken into account. The updated Band model (Masden, 2015) at least allows for uncertainty in terms of the mean collision estimates for the OWFs in the Helgoland Cluster.

However, incomplete knowledge of the behavioural parameters required by the model should not prevent the performance of assessments based on the best information currently available, combined with careful and conservative interpretation of the results obtained. It should be noted that the Band CRM represents the current standard tool used to model bird collisions with offshore wind turbines in the United Kingdom (Masden and Cook, 2016). This model has also been accepted by a steering group comprising representatives of OWF developers, regulators, and environmental advisory bodies in the context of the SOSS for the UK offshore wind industry (Band, 2012). These model results are regularly used to inform consenting decisions regarding OWFs in several European countries.

Table 8. Overview of PBR values based on different f values using the mean kittiwake population for the period 2010–2012 associated with the Helgoland colony as reference population.

Recovery factor (*f* value)	PBR value (annual number of collision victims the population can sustain) (individuals)	Relative increase in mortality
0.1	103	5.38%
0.2	206	10.75%
0.3	309	16.13%

The PBR input parameters generating the results presented in this table are summarized in Table 3.

This assessment aimed to provide an informed evaluation of the degree of threat to the Helgoland kittiwake population from collisions with OWFs. The results indicate that the study population is likely to be under considerable and potentially unsustainable pressure as a result of collisions with OWF turbines. The year-round assessment considered numerous OWFs that are not yet operational, and which are currently only at the consent or planning stage. However, although it is likely that at least some of these projects may not be realized, any cumulative year-round assessment should still consider such conceivable projects at the early planning stages (Brabant *et al.*, 2015).

In the case of the Helgoland kittiwake population, colony-attendance season collisions alone (all involving operational OWFs) have the potential to put considerable pressure on the local population, especially considering that collisions with offshore wind turbines represent only one source of additional mortality caused by anthropogenic activities. Moreover, this example illustrates the added value of cumulative year-round assessments. A spatially and temporally restricted impact assessment could indicate that collisions within the Helgoland OWF Cluster alone would be unlikely to have significant negative effects on the local kittiwake population, based on the argument that the modelled collision value remains below the $f = 0.1$ PBR value, especially when apportioning non-breeding season collisions according to the relative contribution of Helgoland kittiwakes to the overall North Sea non-breeding season population. However, the inclusion of potential non-breeding season impacts across the wintering range leads to a very different interpretation of the endangerment status.

Looking at the seasonal collision estimates separately, figures for the non-breeding season appear remarkably high, considering that the colony-attendance season, as defined in this assessment, is longer (7 months) than the non-breeding season (5 months). There are two possible reasons for this: first, the OWFs of the Helgoland Cluster are located towards the outer limits of the kittiwake foraging range, which is otherwise free of consented and planned OWF projects, and secondly, the non-breeding season collision estimate for UK North Sea OWFs and the extrapolated figure for the entire North Sea include extensive foreseeable but not yet implemented OWF development plans at the North Sea scale. These factors are likely to account for the relatively large collision estimate for the non-breeding season.

The fact that non-breeding season collisions could only be roughly extrapolated based on available CRM results for UK OWF projects indicates that data collected in the context of EIAs (e.g. collision risk estimates) should not only be collected and analysed in accordance with defined methodological standards [e.g. compulsory use of the Band model (Band, 2012; Masden, 2015)], but should also be reported in a consistent format suitable for feeding into a common database to support cumulative impact assessments. Such procedures would be beneficial for year-round impact assessments, especially when implemented on an international, as well as a national scale (Busch *et al.*, 2012). Methodological advances such as the Band model update by Masden (2015) should be implemented as they become available, though holistic assessments will also continue to depend on the inclusion of data from other, often older, assessments, potentially derived using superseded assumptions.

Although not considered in detail in the context of this study, it should be noted that impacts from sources other than OWFs should be considered within truly cumulative assessment approaches, as far as possible. Such assessments, which consider pressures imposed by different drivers on a receptor species, are sometimes referred to as in-combination assessments (Drewitt and Langston, 2006; Therivel, 2009), to distinguish them from cumulative assessments, which are sometimes interpreted in a sectoral manner (i.e. considering impacts of various developments within a certain sector, such as offshore wind energy).

Conclusions

The primary aim of this study was to highlight the need to perform year-round impact assessments that take account of the annual cycles of mobile, migratory species. In addition to providing conceptual thoughts on the matter, this example of the Helgoland kittiwake population illustrates the type of information needed and indicates the suitable tools for implementing such assessments.

In conclusion, year-round impact assessments are needed to allow informed decision making and to generate a realistic understanding of potentially adverse impacts of specific developments on migratory bird populations. This case study indicates that the Helgoland kittiwake population may be under considerable pressure, especially after accounting for potential impacts during the period when the birds are away from their breeding site. Impact assessments should thus give greater consideration to impacts occurring relatively further afield. This case study thus suggests that the planned offshore wind capacities at a North Sea scale conflict with the conservation of the Helgoland kittiwake population, and potentially with several other seabird populations in the North Sea region. The assessment also indicates that data availability may represent a limiting factor in terms of performing cumulative year-round impact assessments. Even in the case of kittiwakes, for which a comparably comprehensive database is available, assessments need to make various assumptions that introduce uncertainty, though these can at least partly be made transparent by employing methodological advances such as the Band model update (Masden, 2015). This study identified a clear need for international assessment standards and more field data, such as tracking studies, that include individuals across their entire species ranges (e.g. Frederiksen *et al.*, 2012).

At the same time, the empirical evidence for coastal and onshore bird collisions with wind turbines highlights the need to conduct thorough investigations and assessments of likely collisions at OWFs, even though such assessments may rely largely on modelling and simulating of the likely effects.

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