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TEESSIDE A & B

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Draft Environmental Statement
Chapter 11 Appendix B
Seabird Displacement Review

EXTENT OF DISPLACEMENT, AND MORTALITY IMPLICATIONS OF DISPLACEMENT OF SEABIRDS BY OFFSHORE WIND FARMS

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EXECUTIVE SUMMARY

Predictions based on species' ecology and conservation importance suggest that divers are the marine birds most at risk from displacement by offshore wind farms (OWFs). Sea ducks, auks, and gannets might be affected, while terns, gulls and petrels are unlikely to be affected.

Displacement of foraging seabirds by OWFs cannot readily be assessed from flight line data alone as only a very small proportion of flying seabirds land in any particular location. There is not yet very much empirical data on displacement of foraging seabirds by OWFs, and so assessment of the amount of displacement arising from developments is somewhat speculative. Available pre- and post-construction data from OWFs indicate variability among sites, but indicate that:

- Divers are displaced;
- Auks may be displaced to some extent by some OWFs, but displacement of auks is partial and apparently negligible at some OWFs;
- Gannets may be displaced, but responses seem to vary among sites; and,
- Gulls, terns and cormorants are generally not displaced and aggregate at some OWFs or show little or no change in distribution.

How displacement responses may alter over the longer term (e.g. through habituation) is uncertain.

There is strong evidence that seabird populations are limited primarily by food abundance. Food abundance affects breeding success and survival of seabirds. Adult auks generally have very high survival rates, even over a wide range of fish stock size, but 'wrecks' of auks can occur when food abundance falls to extremely low levels locally or regionally. Young birds tend to be particularly vulnerable to wrecks, which mostly occur in autumn and winter.

Displacement by OWFs is unlikely to reduce survival of breeding seabirds because they tend to buffer their survival by abandoning breeding when conditions are unfavourable. Displacement may affect breeding success, especially if OWFs are located close to seabird colonies and so displace birds from foraging areas near the colony. Displacement by OWFs far from colonies (at maximum foraging range for the species) may have relatively little impact on breeding success since individual seabirds repeatedly commuting unusually long distances to find food tend to be unsuccessful in their breeding attempts because they are unable to provision chicks as frequently as necessary.

Displacement of nonbreeding seabirds by OWFs may affect survival rates if birds are displaced from high quality foraging habitat and populations are at carrying capacity meaning there is no unoccupied high quality habitat. Displacement of nonbreeders would be unlikely to affect survival rates if birds are displaced from poor quality habitat. So assessment of relative habitat quality is key to assessing whether displacement of seabirds might have population-level impacts.

There are inconsistencies between different SNCBs and regulators in current approaches to assessment of displacement impacts, suggesting a need for agreement on best practice based on evidence (some of which is currently confidential).

Agent-based modelling of red-throated diver population numbers in relation to OWF displacement in the Baltic Sea and Danish waters indicates the potential for a small cumulative impact of displacement. This strategic modelling approach could be extended to divers, auks and gannets in the context of OWFs in the North Sea or other regions.

1. INTRODUCTION

There is concern that offshore wind farms may affect seabird populations as a consequence of displacement of birds from foraging habitat within which wind farms may be constructed (Natural England and Joint Nature Conservation Committee 2012). Firstly, there is considerable uncertainty regarding the likely extent to which seabirds may be displaced by offshore wind farms; this is likely to vary among species, and may be site and state-dependent. Secondly, there is considerable uncertainty as to whether displacement will lead to consequences for survival or productivity of seabirds. The following review considers readily-available evidence that helps to inform whether displacement is likely, and whether such displacement is likely to affect seabird breeding success or survival rates. Emphasis is placed on the latter question. Particular consideration is given to common guillemots and razorbills in the context of developments on the Dogger Bank.

2. EVIDENCE OF DISPLACEMENT OF SEABIRDS BY OFFSHORE WIND FARMS

Based on the known ecology, behaviour and life history of seabird species, Furness et al. (2013) predicted that displacement by offshore wind farms would be likely to affect populations of divers more than any other kinds of seabirds, might affect sea ducks and auks, and was unlikely to affect populations of gulls, terns or petrels. Those predictions were intended to help to set priorities in the absence of a large body of empirical evidence from post-construction monitoring at offshore wind farms, and only until such evidence has accumulated.

Few offshore wind farms have been in existence for long enough to provide the opportunity for empirical analysis of seabird displacement. In some studies, the fact that flying birds show macro-avoidance (changing their flight trajectory to avoid a wind farm) has been interpreted as evidence of displacement. That interpretation is inappropriate. The fact that birds avoid flying through a wind farm does not necessarily lead to displacement if birds are willing to either swim into the area, or to land there and remain on the water. In general, only a very small proportion of flying seabirds will land on the water at any particular location, so the probability of landing determines numbers on the water more than whether or not the flight direction may be altered by presence of turbines. In the following sub-sections, evidence for change in the relative density of seabirds within particular offshore wind farms and in control areas is reviewed. This review is incomplete, since several data sets exist that are not in the public domain or are difficult to access. Given the short time-scale available for this review the following sections present only the evidence from widely available published studies. It should be possible to add to these examples with data from some further sites, which could increase the confidence that can be placed on conclusions (e.g. with further columns of data in Table 1 in Section 2.6). However, some data have been excluded from this assessment where data quality compromises conclusions. For example, Rothery et al. (2009) counted numbers of marine birds flying past Blyth Wind Farm and compared numbers seen pre- and post-construction, but they did not count numbers in a control area so were unable to distinguish between changes due to the wind farm and other trends. Their results suggested a decrease in numbers of cormorants, apparent increases in Sandwich tern and great-black-backed gull numbers, and no change in numbers of herring gulls, black-headed gulls, kittiwakes, eiders, common scoters, and gannets passing the wind farm. However, without data from control areas outwith the wind farm it is unclear whether any of these changes were due to the wind farm.

2.1 Horns Rev Offshore Wind Farm

Post-construction studies showed almost complete absence of red-throated divers and common scoters within the Horns Rev Offshore Wind Farm in the years immediately following construction

(2002-2006) but most species showed no significant change (gannet, herring gull, kittiwake, Arctic tern, common tern, common guillemot, razorbill) (Petersen et al. 2006; Fox et al. 2006), or occurred in too small numbers to permit statistical analysis (Petersen et al. 2006). Leopold et al. (2011) cite studies at Horns Rev as showing displacement of common guillemots, but that assertion is not entirely correct. Fox et al. (2006), Petersen et al. (2006) and other reports from ornithological work at Horns Rev clearly state that there was no statistically significant change in common guillemot density within the wind farm post-construction, but that flying common guillemots did tend to show macro-avoidance, altering flight paths to avoid flying through the wind farm. So while there is an effect on flying common guillemots, empirical data from surveys shows no statistically significant impact on common guillemot density on the sea. There was, however, a statistically insignificant reduction in relative abundance of guillemots within the wind farm area, suggesting that there might possibly be a small displacement effect, not statistically significant due to high variability in the data (Petersen et al. 2006). Petersen et al. (2006) suggest that little gulls may have been attracted by the wind farm; numbers were higher post-construction within the wind farm and the 2 km buffer, but the increase was only statistically significant for the buffer area.

Although the initial post-construction studies at Horns Rev suggested that common scoter avoided the wind farm (Petersen et al. 2006), high numbers of common scoters appeared within the wind farm in 2006-07. While this change in behaviour might have been due to habituation, it appears to have arisen as a result of large changes in the distribution of preferred prey (molluscs) rather than a response to the presence of wind turbines (Leonhard et al. 2013). There was a high overlap between mollusc distribution and common scoter distribution, with no evidence of displacement by the wind farm, leading to the conclusion that the dramatic shift in common scoter distribution between 2002 and 2006 that had been attributed to displacement, was in fact a result of changes in the abundance and distribution of molluscs (Leonhard et al. 2013). However, red-throated divers continued to avoid the wind farm, apparently showing a strong and persistent behavioural response to the presence of turbines, but were the only birds at this site to show significant displacement (Leonhard et al. 2013).

2.2 Nysted Offshore Wind Farm

At Nysted Offshore Wind Farm, Petersen et al. (2006) and Fox et al. (2006) reported significant reductions in long-tailed duck densities within the wind farm relative to numbers there pre-construction, although numbers involved were small in relation to the flyway population. This conclusion was slightly complicated by the fact that the pre-construction distribution of long-tailed ducks indicated quite a strong preference for the area that was to become the wind farm. Post-construction, long-tailed duck numbers within the wind farm remained higher than outside the wind farm, but this preference was less pronounced, indicating some displacement. Most other species showed no significant change (cormorant, mute swan, goldeneye, common eider, herring gull and great black-backed gull) or occurred in too small numbers to permit statistical analysis (Petersen et al. 2006; Fox et al. 2006). Red-breasted mergansers '*showed indications of an increased preference of the wind farm site and its 2 and 4 km zones after the erection of the wind farm. Increased fish availability in the area in the post-construction phase could possibly be an explanation for this increase*' (Petersen et al. 2006).

Modelling of the distribution of the long-tailed ducks at Nysted (using the same data) showed that the density of birds within the wind farm was significantly lower than in control areas (Petersen et al. 2011), in agreement with the interpretation by Petersen et al. (2006) and Fox et al. (2006).

2.3 Kentish Flats Offshore Wind Farm

Kentish Flats monitoring reports indicated that there had been a statistically significant decrease in diver numbers within the wind farm site, most markedly within 500m of turbines. Analysis of data from 2009-10 confirmed a reduction in diver numbers within the wind farm site and 500m buffer (Percival 2010). There was some suggestion that the magnitude of the displacement may be decreasing through time; divers may be habituating to the presence of the wind turbines. The 2009-10 data showed more diver records from within the wind farm site than there had been in 2008-09 (Percival 2010). Rexstad and Buckland (2012) carried out further, more sophisticated and complex, analysis of post-construction data from Kentish Flats Offshore Wind Farm and concluded that on the basis of their methodology '*locations where the upper confidence interval for the difference in bird abundance was negative would be indicative of a decrease in bird abundance following operation of the Kentish Flats wind farm. No species had areas within the wind farm footprint where these upper confidence intervals were negative, red-throated divers did display some locations in the northeastern portion of the study area where they were estimated to be in lower abundance 2005-2010 than they were in 2001-2004*.'

2.4 Dutch Offshore Wind Farms

At the Egmond aan Zee Offshore Wind Farm and at the neighbouring Princess Amalia Windpark, numbers of most species of seabirds were too small to allow robust statistical analysis of displacement or attraction. Considering only the surveys where numbers were sufficient to permit analysis, cormorants showed significant attraction to the wind farms in 10 out of 14 surveys, increasing considerably in density within the wind farm areas post-construction after adopting the habit of roosting on wind farm structures (Leopold et al. 2011). Significant displacement was found in divers (3 out of 8 surveys), great-crested grebes (1 out of 4 surveys), gannets (2 out of 10 surveys), little gulls (1 out of 7 surveys), common guillemots (2 out of 11 surveys) and razorbills (1 out of 6 surveys). However, for all of these species, statistically significant displacement occurred in only a minority of surveys (Leopold et al. 2011). When it did occur, displacement was incomplete with a reduction in common guillemot density typically of about 50%, and no greater within the wind farms than the magnitude of displacement caused by anchored shipping in an adjacent area (Leopold et al. 2011). There was a suggestion that displacement was higher in common guillemots in the Princess Amalia Windpark than in Egmond aan Zee Offshore Wind Farm, which related to a higher density of turbines in the former (Leopold et al. 2011). In the context of gulls, Leopold et al. (2011) concluded that for common gull, lesser and greater black-backed gull, herring gull and kittiwake, there was hardly any effect of the wind farm on their distribution, which was predominantly related to trawl fishing activity levels in the area. Also working at Egmond aan Zee Offshore Wind Farm, Krijgsveld et al. (2011) concluded that '*Seabirds such as gannets, scoters, alcids and divers showed the highest levels of avoidance, while gulls (various species) and especially cormorants did not avoid the wind farm and most likely were attracted to it*'. For the same sites Lindeboom et al. (2011) concluded '*Gulls, cormorants and terns did not avoid the farm and used it for foraging. But gannets, scoters, auks, guillemots and divers showed strong avoidance behaviour in their flight pattern in the vicinity of the farm'* though the strong avoidance behaviour in their flight pattern reported by Lindeboom et al. (2011) clearly did not translate into strong displacement of foraging (birds on the sea surface) given the analyses reported above (Leopold et al. 2011). Thus Lindeboom et al. (2011) also concluded '*Ship-based bird counts in and around the wind farm indicated avoidance behaviour of common scoters and northern gannets, no marked avoidance by divers, guillemots, razorbills and most gulls and attraction for cormorants*'. However, some aspects of this conclusion seem to be slightly inconsistent with the analysis of the same data presented by Leopold et al. (2011).

2.5 Belgian Offshore Wind Farms

At Thorntonbank Offshore Wind Farm, post-construction monitoring in 2009-11 following a BACI design showed that densities of little gulls, great black-backed gulls, kittiwakes, Sandwich terns and common terns increased significantly within the wind farm, whereas densities of fulmars, gannets, lesser black-backed gulls, herring gulls, common gulls, common guillemots and razorbills showed no significant change (Vanermen et al. 2012).

At Blighbank offshore wind farm, post-construction monitoring in 2010-11 following a BACI design showed that densities of common gulls increased significantly within the wind farm, densities of common guillemots and gannets decreased significantly (by 70% and 30% respectively), and densities of fulmars, great skuas, little gulls, lesser black-backed gulls, herring gulls, kittiwakes and razorbills showed no significant change (Vanermen et al. 2012).

Vanermen et al. (2012) concluded '*During recent surveys in 2012, good numbers of auks and even Harbour porpoises were encountered inside the wind farm. From an ecological point of view, the presence of auks is very interesting, and we wonder if these self-fishing species are already habituating to the presence of the turbines, and if they will profit from a (hypothetical) increase in food availability*'. The possibility that birds will habituate to offshore wind farms is difficult to assess in view of the relatively short time-scale over which these have been present in the environment.

2.6 Summary of evidence regarding seabird displacement by Offshore Wind Farms

Published estimates of seabird displacement rates based on high quality before-after-control-impact (BACI) design studies or equivalent high quality modelling studies are summarised in Table 1. This table presents empirical data where the sample sizes are adequate to provide statistically robust evidence, and classifies evidence into 'traffic-light' categories as: Red = strongly displaced, or displaced in 80-100% of data sets; Orange = mildly but significantly displaced, or displaced significantly in 40-70% of data sets; Yellow = slight evidence of displacement, or displaced significantly in 10-30% of data sets; Green = Not displaced; Blue = Significantly attracted into wind farm area. The evidence summarised in Table 1 indicates that divers appear to be consistently displaced by offshore wind farms. Evidence for displacement of other species of marine birds appears less clearcut.

Common guillemots were displaced by Blighbank OWF, were displaced only in a minority of surveys at the two Dutch OWFs, but were not significantly displaced by Horns Rev OWF (although the data suggest that slight displacement was probably occurring) or Thorntonbank OWF. Razorbills were displaced in one out of six surveys at the Dutch OWFs, but not at Horns Rev, Thorntonbank OWF or Blighbank OWF. Long-tailed ducks were displaced by Nysted OWF, but apparent displacement of common scoters by Horns Rev OWF was found to be redistribution relating to prey availability and not a response to the wind farm. Gannets were slightly displaced at the Dutch OWFs and Blighbank OWF, but not at Horns Rev OWF or Thorntonbank OWF.

Differences in behavioural responses among wind farms and among surveys at particular sites, may relate to differences in the quality of foraging habitat for particular species at particular wind farms, or seasonal differences in the importance of particular habitats. So variation among sites is not unexpected. However, overall, the empirical data appear to indicate that at worst, a low level of displacement by offshore wind farms occurs for a small number of marine bird species. Particular concern about displacement of divers by offshore wind farms does appear appropriate, as predicted from different empirical data by Furness et al. (2013).

Given the particular concern about relatively large numbers of common guillemots and razorbills at Creyke Beck A and B OWFs (Dogger Bank), it is noteworthy that the empirical evidence indicates that these species are rarely displaced by offshore wind farms (Table 1).

Table 1. Classification of evidence of the impact (displacement, no effect, or attraction) of foraging seabirds by offshore wind farms. Red = strongly displaced, or displaced in 80-100% of data sets; Orange = mildly but significantly displaced, or displaced significantly in 40-70% of data sets; Yellow = slight evidence of displacement, or displaced significantly in 10-30% of data sets; Green = Not displaced; Blue = Significantly attracted into wind farm area. - = not enough data for this species at this site to carry out an analysis. Values in cells indicate the proportion of relevant results reported.

Species / group	Horns Rev	Nysted	Kentish Flats	Egmond aan Zee & Princess Amalia	Thornton Bank	Bligh Bank
Divers	🟥	-	🟧	🟧 3/8	-	-
Common guillemots	🟨	-	-	🟨 2/11	🟩	🟧 70%
Razorbills	🟩	-	-	🟨 1/6	🟩	🟩
Long-tailed duck	-	🟥	-	-	-	-
Grebes	-	-	-	🟨 1/4	-	-
Gannet	🟩	-	-	🟨 2/10	🟩	🟨 30%
Little gull	🟦	-	-	🟨 1/7	🟦	🟩
Common scoter	🟩	-	-	-	-	-
Kittiwake	🟩	-	-	🟩	🟦	🟩
Herring gull	🟩	🟩	-	🟩	🟩	🟩
Terns	🟩	-	-	🟩	🟦	-
Great black-backed gull	-	🟩	-	🟩	🟦	-
Lesser black-backed gull	-	-	-	🟩	🟩	🟩
Common gull	-	-	-	🟩	🟩	🟦
Cormorant	-	🟦	-	🟦 10/14	-	-

- = numbers too small for a sound statistical assessment to be made at this site

3. ECOLOGY OF COMMON GUILLEMOTS AND RAZORBILLS

Common guillemots and razorbills catch prey by 'flying' underwater, chasing small fish. They are capable of diving to considerable depths, common guillemots regularly feed at 60 m depth and have been recorded diving below 100 m. So in the North Sea, these seabirds are able to dive to the sea floor without difficulty. Razorbills tend to feed on slightly smaller fish than preferred by common guillemots. The latter species takes sandeels or other small pelagic fish that typically weigh up to about 20g, the ability to take larger fish being constrained by the birds' gape size. In the North Sea, both species feed mainly on sandeels in summer and feed their young primarily on sandeels. In southern parts of the North Sea, sprats may be important secondary prey, but there are no sprat stocks in the far north of the North Sea (Orkney, Shetland, off Caithness) where the largest guillemot and razorbill numbers breed. Breeding common guillemots and razorbills fly fast and can forage tens of kilometres from the breeding site. However, when they are unable to find food close to their colony and are forced to commute large distances to search for fish, their energy expenditure increases and their time budget becomes constrained so their rate of prey delivery to chicks declines. This leads to reduced chick growth rates, lower chick fledging mass, and eventually to chick death (either due to adult absence from the chick allowing predators to take chicks, or due to chicks starving to death at the nest site). Deployment of data loggers on breeding auks has shown some exceptionally long foraging trips (over 100 km from the nest) in recent years associated with severely depleted stocks of sandeels in the northern North Sea, but most common guillemots and razorbills feed primarily within 50 km of the colony, and breeding success is likely to be close to zero if adults are having to forage at much greater distances.

Sandeels feed on zooplankton during the spring and summer plankton bloom. They grow rapidly and are short-lived fish but are very abundant and widely distributed in the North Sea, supporting the biggest commercial fishery in the North Sea. Sandeels store lipid, and when they have achieved a high lipid content they survive overwinter on their accumulated energy store. From about August to March, sandeels tend to remain buried in the sand, and so are largely unavailable to surface-feeding seabirds. However, guillemots are known to dig sandeels out of the sand and eat them even in winter, and razorbills almost certainly do the same. In winter these auks will feed on other small pelagic fish such as sprats and young herring, and seasonal movements of auks often relate to the locations where there are large concentrations of these species which are more readily available in winter. Sprats tend to occur in estuarine habitat, and concentrations that attract auks in winter can be found in the Moray Firth, Firth of Forth area, off east England, in the German Bight and in the Skagerrak and Kattegat. In winter, a variety of other small fish may be eaten, including gobies and young gadoids (such as young saithe). In addition to small fish, some zooplankton may possibly be eaten by razorbills in winter, but common guillemots are unable to survive on a diet of zooplankton and are dependent on having access to small pelagic fish.

When chicks fledge in July, the male accompanies the chick which swims away from the breeding colony to feeding areas. This can involve long movements, with guillemots and razorbills from Shetland swimming to the Moray Firth, or to Danish waters. Auks become flightless when moulting their flight feathers, which makes them more efficient at flying underwater (by reducing surface area of the wing). Some 'wrecks' of auks involving deaths of hundreds or thousands of birds, have been associated with this vulnerable period of moult, and may occur if birds moult in areas where for some reason the prey fish on which they depend are absent. More often, wrecks involve primarily young birds, whose inexperience can lead them to starve if prey density is unusually low, and occasionally large numbers of young guillemots come ashore in late autumn starving. Mortality of young auks is probably quite high in most winters because they are inexperienced, whereas mortality of adult auks is normally extremely low (only around 5% per annum in most years). Most mortality of adult auks seems to occur in winter, but it is unclear how much the mortality rate is determined by events during breeding and how much by winter conditions. In very rare years when the abundance of small pelagic fish is exceptionally low, large wrecks of adult and juvenile auks can occur. The most spectacular of these occurred in the Barents Sea in 1986-87, when over 1 million common guillemots died because the stock of their key prey fish, the capelin, had fallen to less than 1% of its normal abundance due to fisheries impacts and indirect ecosystem interactions between cod, herring and capelin (Sakshaug et al. 2009). Auks can be affected by pollution, drowning in set fishing nets, and other factors, but the key influence on their mortality is food abundance, affecting either local areas or whole regions on the rare occasions when pelagic fish abundance falls to extremely low levels locally or regionally.

4. ECOLOGICAL THEORY - EFFECTS OF DISPLACEMENT ON BREEDING SEABIRDS

Ashmole (1963) developed a theory that tropical seabird populations are limited by competition for food within the foraging range of breeders from their breeding colony. This theory was based on the observation that seabirds can disperse over large areas outside the breeding season (while 'nonbreeders'), but are limited to relatively small areas occupied by large numbers of birds while breeding. So competition for food within the waters around breeding colonies limits population size. Subsequently, this theory was extended to temperate seabirds by Furness and Birkhead (1984) who showed that colony sizes of seabirds around the British Isles supported Ashmole's theory, and also subsequently to polar seabirds by Ainley et al. (2004) and Elliott et al. (2009). A wide range of studies of seabirds around the world have provided further support for this theory, and it is now well established both theoretically (Cairns 1989) and empirically (Birt et al. 1987; Lewis et al. 2001; Ainley et al. 2004; Elliott et al. 2009; Wakefield et al. 2013). Thus the evidence indicates that seabird population sizes are primarily determined by food abundance in the foraging area around colonies during the breeding season.

4.1 The role of food supply for breeding seabirds

There is a broad consensus that seabird numbers are particularly affected by food abundance, and that this factor (in some cases modulated by fisheries or by climate change) is the single most important influence on seabird population sizes at a regional level. Small, surface-feeding seabirds with short foraging ranges and a lack of alternative foods are especially vulnerable to such impacts (Furness and Tasker 2000). Changes in breeding numbers of seabirds resulting from changes in fish abundance can be dramatic. For example, common tern breeding numbers in the Firth of Forth were reduced to about half when sprat abundance fell (Jennings et al. 2012), Arctic tern and Arctic skua breeding numbers in Shetland fell by at least 50% after the decline of the Shetland sandeel stock (Forrester et al. 2007), common guillemot breeding numbers in the Barents Sea fell by over 90% when the capelin stock collapsed (Sakshaug et al. 2009). In contrast, swift tern and African penguin numbers increased in part of the Benguela ecosystem when the sardine stock redistributed into the vicinity of these particular colonies, while breeding numbers in areas where fish abundance declined fell dramatically (Cury et al. 2011).

Mitchell et al. (2004) concluded that seabird demography in Britain and Ireland is '*strongly affected by the availability of food*'. Many seabirds around the British Isles feed primarily on small schooling pelagic fish. These fish are important food because they tend to be abundant, available in the upper layers of the sea, have a high energy density, and are relatively small so are easy for seabirds to catch and swallow. In many different parts of the world and for many different kinds of seabirds, breeding success shows a strong sigmoidal correlation with the abundance of their preferred prey fish (Cury et al. 2011). Cury et al. (2011) identified a threshold of one third of the long-term maximum prey biomass of forage fish abundance. Below this stock biomass level, many seabird species suffer from reduced and more variable productivity which is likely to lead to population decline if sustained. When food fish abundance is above one-third of the long-term maximum, seabird breeding success tends to remain consistently high unless affected by weather or predators. It is difficult to assess from empirical data how many years of poor food supply are required to lead to seabird population decline, as there can be a considerable lag in this effect, since seabirds only recruit into the breeding population when several years old. In addition, it is difficult to detect small changes in breeding numbers, so a decline may occur without being detected for some years. Breeding numbers can also be buffered by a pool of nonbreeders and a decline in breeding numbers may only become evident after that pool has been depleted (Klomp and Furness 1992). This may take many years, and possibly one or two decades.

Several species' breeding success in Shetland, including that of the black-legged kittiwake (Votier et al. 2008), Arctic tern and Arctic skua, shows strong correlation with sandeel stock biomass (Furness, 2002, 2007). On the Isle of May, there was a positive correlation between seabird breeding productivity and the size of sandeel prey (both in terms of individual fish size and stock biomass) (seen for Atlantic puffin, shag, common guillemot, razorbill and kittiwake; Frederiksen et al. 2006). Another study on the Isle of May kittiwake colony in relation to the opening and closure of the Danish sandeel fishery operating locally showed that breeding productivity was significantly reduced while the fishery was active, reducing the availability of the kittwake's preferred prey (Frederiksen et al. 2008). In the Firth of Forth, common tern breeding numbers at individual colonies show rather different trends over recent decades, but the regional population size correlates with the abundance of sprats (their main breeding season food) in the area (Jennings et al. 2012). The variable dynamics of individual colonies seems to be driven by predation impacts and presence of gulls, with terns moving between colonies in response. Overall breeding numbers in the region varied much less than numbers at individual colonies. On the east coast of England and Scotland, large kittiwake colony "clusters" were associated with aggregations of sandeels whereas on the west coast, these predator-prey dynamics were not as apparent. The study showed that regional variation in prey abundance has a stronger impact on kittiwake populations than local prey depletion (Frederiksen et al. 2005, Fauchald et al. 2011).

Arctic skua breeding success in Shetland has shown a strong correlation with the biomass of the Shetland sandeel stock, but in addition the breeding numbers in Shetland have declined substantially over the last 20 years. A study was carried out to find out the source of this population decline and it was found that breeding pairs supplemented with food had a higher nest attendance rate than those without supplement. Not only did food availability have a strong impact on breeding success, but it also affected adult survival. Birds given supplementary food were more likely to return to breed the next year than were unfed controls (Davis et al. 2005). This supports the frequent suggestion that improved food supply will increase overwinter survival of seabirds through improvement to body condition. However, demonstrating this effect is difficult and has been beyond the scope of most studies on seabird ecology.

The breeding success of common guillemots, razorbills and shags in the North Sea has shown a positive correlation to sandeel growth rates (Burthe et al. 2012). In Shetland and on the Isle of May, a positive correlation between 0-group sandeel abundance and adult survival of kittiwakes is apparent (Oro and Furness, 2002, Wanless et al. 2007). Food abundance can affect a wide range of demographic parameters and even such biometrics as egg size. Decreases in puffin egg size at colonies in Norway and Scotland have been related to effects of reduced food fish abundance (Barrett et al. 2012).

Food availability also affects the foraging ranges of seabirds, with decreased levels of food increasing the distance that birds will travel to feed. A study of breeding northern gannet colonies around the

UK concluded that there was a positive correlation between population size and mean foraging trip duration (Lewis et al. 2001). Larger populations of seabirds increase competition for food and hence at larger colonies, birds will have to travel further to obtain food, depleting energy stores and potentially leaving nests unattended during the breeding season. This increase in foraging range is considered the likely mechanism limiting colony size, and can be brought about either by increases in seabird numbers increasing competition for food, or by depletion of food fish stocks (e.g. by fisheries) reducing food supply for seabirds.

4.2 Impacts of displacement of breeding seabirds

Displacement of foraging breeding birds could affect breeding success and possibly also survival rates, if breeding birds are required to travel further in search of food (Masden et al. 2010). In this case, impacts would be likely to be greatest when offshore wind farms are close to a colony of seabirds with limited maximum foraging range. In this situation, the wind farm area could represent an important fraction of the available foraging habitat for the breeding birds constrained as central-place foragers to remain close to their breeding site (Masden et al. 2010). Attempts have been made to model this interaction. Speakman et al. (2009) assessed energy costs to breeding seabirds of the extra flight required to fly around, rather than through, offshore wind farms placed between their breeding site and their foraging area. This work concluded that in most plausible scenarios, costs added by wind farms were negligible in relation to the overall daily energy budget of breeding seabirds. The same conclusion was reached by Masden et al. (2010).

McDonald et al. (2012) developed a model for the common guillemot breeding at the Isle of May, east Scotland. Their model demonstrated that displacement of foraging seabirds from an offshore renewables development could result in changes to their time/energy budgets, with potential consequences for breeding performance and/or survival. However, they were unable to link the estimated additional costs in time and energy to individual fitness, either in terms of breeding success or survival. Such relationships are very unlikely to be linear, and would be difficult to measure empirically. But ecological theory predicts that costs would impact on breeding success at higher thresholds than would lead to impacts on survival (Cairns 1987). Breeding adult seabirds are expected to buffer their own survival by abandoning breeding efforts when conditions are unfavourable, since fitness is primarily determined by survival rate in long-lived animals (Cairns 1987).

Langton (2013) presented in her PhD thesis work assessing the relationship between displacement of breeding common guillemots from foraging habitat and consequences for adult foraging rate, chick growth, breeding success, breeding adult body condition and survival. The modelling approach she used requires knowledge of the functional response in relation to food (in that case sandeel) abundance. Langton (2013) fitted this sigmoid curve based on only two data points on foraging success at a colony in Shetland in relation to estimates of (dramatically different) sandeel abundance in two consecutive years. This functional relationship was therefore based on extremely limited data. However, the model indicated that displacement would reduce chick growth so could affect chick survival, but that there would be little or no impact on adult survival. One clear conclusion, therefore, was that common guillemot survival is likely to be highly robust to displacement by offshore wind farms within their foraging range (and was not affected by displacement in the modelling) whereas breeding success would be more likely to be reduced if birds were unable to use suitable foraging habitat close to the colony. Displacement from habitat towards the extreme of their foraging range, however, was found to have very little impact on breeding success, because birds having to commute to these extreme distances were already failing to rear chicks due to the costs of long flight distances in terms both of energy cost and time constraints (Langton 2013).

This evidence strongly suggests that displacement by offshore wind farms could affect seabird breeding success, but is less likely to affect survival rates of breeding seabirds, because birds failing to rear chicks would be expected to abandon breeding before energy or time costs impacted significantly on their own survival.

5. ECOLOGICAL THEORY - EFFECTS OF DISPLACEMENT ON NON-BREEDING SEABIRDS

Nonbreeders can be defined as immature birds, or birds that have previously bred but are taking a year off from breeding, or birds that are 'breeders' but are outside the breeding season, as migrants or wintering birds. Seabirds typically start to breed when 4-8 years old, and so a typical seabird population may contain as many nonbreeders as breeders during the breeding season, while all birds are classified as 'nonbreeders' during the migration and wintering seasons. Nevertheless, seabird population sizes are typically expressed as breeding pairs rather than in terms of total population numbers, as the latter are very much more difficult to census. Nonbreeders may show very different 'at-sea' distributions to breeders, as they do not necessarily need to remain in the vicinity of breeding colonies. Some nonbreeders (especially young immature birds) spend the summer in the wintering area rather than returning to the general breeding area, presumably because there is less competition for food when remaining in an area that has been vacated by breeding adults. Similarly, some nonbreeders visit areas away from breeding colonies, probably for the same reason. It is likely, although difficult to demonstrate, that many of the seabirds in central areas of the North Sea during the breeding season may be nonbreeders feeding on resources that are too distant from coastal breeding colonies for breeding adults to utilise while they are central-place foragers based at the colony.

5.1 The role of food supply for nonbreeding seabirds

The impact of food availability on adult survival is potentially the most influential factor affecting seabird population trend (Mitchell et al. 2004). In waters around the British Isles, sandeels, sprats, and juvenile herring, are preferred food for most seabirds in summer (Mitchell et al. 2004). In winter, a few deep-diving seabirds (such as common guillemots and shags) continue to feed on sandeels but because sandeels spend most of the winter buried in the sea bed, they are unavailable to surface-feeding or shallow-diving seabird species; seabird diets in winter tend to be more diverse than in summer, with less focus on small pelagic fish. In northern waters such as around Shetland, there are no sprats and no juvenile herring, so seabirds depend strongly on sandeels in summer and tend to move away from the area to spend the winter elsewhere. In coastal areas further south there may be sprats and young herring as well as sandeels, providing seabirds with a more stable prey base, and after breeding, some seabirds aggregate in areas where high densities of sprats or young herring are present, such as the Skagerrak and Kattegat off Denmark (Skov et al. 2000). Others migrate further afield, to feed on small fish such as pilchards and sardines in sub-tropical waters. Some species show changes in migrations and winter distributions over decades, responding to changes in food distribution. For example, common guillemots from British colonies show changes in distribution over decades that relate to long-term changes in abundance of sprats and young herring in the North Sea and in Danish waters (Lyngs and Kampp 1996).

Scottish breeding great skuas were shown by deployment of data loggers to now mainly winter off northwest Africa, much further south than previously thought. This migration further south could be linked to an increase in fishery discards and increasing pelagic fish stocks in the area, making west African waters more profitable for foraging than the waters off Iberia and France where this species used to winter (Magnusdottir et al. 2012).

A similar increase in southerly migration by gannets has been identified, and interpreted as a response to reduced discarding in the North Sea and increased discarding by new trawl fisheries and increased pelagic fish stocks off west Africa (Kubetzki et al. 2009). Comparison of flight time data from geolocation loggers on wintering northern gannets suggested that food availability to gannets at this time is much more varied in the north of the range (Bay of Biscay, Celtic sea) but more consistent and more abundant off Western Africa (Garthe et al. 2012).

Harris et al. (2010) used geolocators to identify wintering areas of breeding adult puffins from the Isle of May. Mortality rates of puffins at this colony had increased and the authors concluded that increased mortality related to reduced food availability in the wintering areas used by these birds in the North Sea, and changes in their migration behaviour. This indicates a complex relationship between food, survival and productivity. Birds unable to find sufficient food during winter may not be in adequate body condition in spring to sustain successful breeding, so that food shortage may have complex 'carry-over' effects on productivity as well as migration behaviour and overwinter survival.

The survival of Scottish adult common guillemots has been correlated for colonies which share wintering areas, suggesting that some environmental factor present at these shared wintering sites is affecting adult guillemot survival (Reynolds et al. 2011). Perhaps the most likely common factor affecting adult survival is food availability. However, other factors can also be important. A study on the effect of oil spills on adult common guillemots showed that major oil pollution incidences doubled their winter mortality rate (Votier at al. 2005, 2008). Nevertheless, it is thought that after an oil spill, when it affects adult seabird survival, population declines may occur the following year but that after the decline, populations generally recover again (Piatt and Roseneau, 1999), while in some cases, such as the Braer oil spill in Shetland, impacts on breeding seabird numbers may be too small to detect even at the oil spill site (Heubeck 1997), suggesting that populations are indeed mainly limited by food supply rather than by other factors.

5.2 Impacts of displacement of nonbreeding seabirds

Displacement of nonbreeders would have no impact on survival rates or body condition if the seabird population(s) were well below carrying capacity, since birds could use alternative habitat (Newton 1998). However, if seabird numbers were at, or close to, carrying capacity, then displacement could lead to birds being unable to find alternative foraging habitat that was not already fully occupied, in which case an impact on survival rates would be likely. Birds tend to occupy high quality habitat first, then overspill into poor quality habitat when optimal habitat is fully occupied (Newton 1998). Carrying capacity will be reached when a large enough proportion of the population is in poor quality habitat and so net population growth falls to zero (Newton 1998).

Displacement from high quality habitat would lead to redistribution of birds into poor quality habitat and could have consequences for survival prospects or body condition of the individual birds concerned.

Displacement from areas of poor quality habitat, however, can be anticipated to have less impact than displacement from high quality habitat, since populations at carrying capacity will fully occupy optimal habitat, but only occupy some of the poor quality habitat, according to the balance of profitability of different habitats which will determine net population growth rate of zero as the sum of (positive) growth generated in good quality habitat and (negative) growth generated in poor quality habitat (Newton 1998). Displacement from one area of poor quality habitat to other (unoccupied) areas of poor quality habitat may have little or no impact on population size.

So this clearly raises two key questions: are seabird populations at carrying capacity, and would displacement of nonbreeders by offshore wind farms represent loss of high quality habitat, or of poor quality habitat?

Carrying capacity is likely to be set by food resources (Newton 1998). But there is a paradox that although small forage fish populations (stocks) such as sandeels fluctuate in abundance over a wide range in a short time scale (a few years), seabirds tend to be long-lived (adult seabirds may typically live up to 40 years). So clearly seabirds are able to survive in years when food stocks are relatively low. Survival rates of seabirds appear to remain high at levels of food supply that start to cause declines in productivity (Cairns 1987; Cury et al. 2011). This suggests that a small loss of foraging habitat would normally be within the range of natural variation to which seabird populations have evolved and adapted.

Fisheries stock assessment data indicate the extremely high year to year variation in abundance of small pelagic fish such as sandeel and sprat. ICES separate the North Sea into a number of distinct stocks of sandeels with independent dynamics and low connectivity through larval drift, including area 1 (Dogger Bank), area 2 (German Bight, Danish and Dutch coastal waters) and area 3 (Norwegian waters) (ICES 2010). In the Dogger Bank region of the North Sea from 1983 to 2011, the sandeel spawning stock biomass varied between 100,000 tonnes and 1,100,000 tonnes (an 11-fold range), while recruitment of sandeels varied between 5 billion and 950 billion fish (a 190-fold range) (ICES 2010). In the Norwegian sandeel stock, spawning stock biomass varied between 40,000 t and 420,000 t (a 10-fold range), while numbers of recruits varied from 10 billion to 900 billion (a 90-fold range). In the stock distributed in coastal waters from Denmark to the Netherlands, spawning stock biomass varied from 40,000 t to 290,000 t (a 7-fold range), and recruits varied from 2 billion to 240 billion (a 120-fold range) (ICES 2010). The Shetland sandeel total stock biomass varied from 162,000 tonnes in 1981 to 15,000 tonnes in 1990 (more than a 10-fold range). Recruitment to that stock varied from 15 billion fish in 1990 to 123 billion fish in 1991, showing the dramatic differences that can occur even from year to year.

During all of these fluctuations in sandeel abundance, populations of common guillemots and razorbills in the North Sea remained fairly steady (Mitchell et al. 2004, Forrester et al. 2007). This is true both of breeding numbers on North Sea coasts, and apparently of wintering numbers in the North Sea although data on the latter are less reliable. The implication of this is that common guillemots and razorbills can survive well even when sandeel abundance is a small fraction of the long-term average abundance, suggesting that these populations are not at carrying capacity at least in most normal years. Similar data for capelin stock biomass in the Barents Sea also demonstrates that over a wide range of capelin stock biomass (spawning stock biomass varying from 1 million to over 7 million tonnes) the common guillemot population remained moderately stable; however in this case a dramatic collapse of capelin stock in 1986 to below 50,000 tonnes led to mortality of about 95% of the common guillemots (about 1 million birds) in the Barents Sea in winter 1986-87 (Sakshaug et al. 2009). In this case it would appear that the population of common guillemots was below carrying capacity in most years, but that in 1986 the extreme decrease in capelin stock biomass reduced carrying capacity dramatically.

However, occasional ‘wrecks’ of seabirds occur (e.g. Piatt and van Pelt 1997), and offshore wind farms placed in areas of high quality habitat may increase frequency or intensity of ‘wrecks’ if high quality foraging habitat is lost and seabird numbers are at carrying capacity.

It is also possible that offshore wind farms may provide marine protected areas where small forage fish can increase in abundance as a consequence of exclusion of fisheries and so could potentially convert low quality habitat into high quality habitat, but the relevance of such indirect effects is highly uncertain (Fox et al. 2006; ICES 2011; Busch et al. 2012).

6. METHODS USED IN PREVIOUS OWF ASSESSMENTS

Given that divers are the marine birds thought most likely to be displaced by offshore wind farms, and are birds of high conservation concern, Topping & Petersen (2011) developed an Agent-based model to assess the population-level consequences of displacement of red-throated divers by offshore wind farms in the Baltic Sea and Danish waters. They used the model to explore three scenarios. The first considered existing offshore wind farms in Danish waters, and modelling indicated no impact on red-throated diver numbers. The second scenario considered existing and planned future offshore wind farms in Danish waters and modelling predicted a 0.1% reduction in diver numbers resulting from displacement from these sites, while the third scenario was the presence of all existing and proposed offshore wind farms in the Baltic Sea and Danish waters. That scenario (with a total area of 15,000 km^2 classified as offshore wind farms) predicted a 1.7% reduction in red-throated diver numbers in this flyway. This modelling approach makes numerous assumptions, and is based on rather limited data on red-throated diver behaviour, but provides a very useful approach to estimating cumulative impact of displacement. The results suggest that even for this species, considered to be one of the marine birds most likely to be at risk of displacement impacts, a relatively small population-level impact was predicted due to the construction of offshore wind farms, at least for the Baltic Sea and Danish waters. No extension of this model to consider development scenarios in the North Sea, or to other seabird species, has yet been published, but this Cumulative Impact Assessment by Agent-based modelling appears to provide a useful approach that could be used strategically for North Sea auks and gannets as well as divers.

The Beatrice OWF Environmental Statement developed a mechanistic model to explore how seabirds may be displaced by offshore wind farms (Trinder et al. 2012). The model used predictions of radial turbine avoidance distance to estimate the total area from which seabirds could be displaced. The predictions from the displacement model were coupled with simple population models to estimate the magnitude of any impact during the breeding season. As recommended by the regulator (Marine Scotland), the impact of displacement of breeding birds was considered to be a reduction in reproductive output, with no impact on survival (an approach that is consistent with the modelling by Langton 2013). Displacement of seabirds from areas within a certain specified distance of each wind turbine provided estimates of numbers of seabirds that would be displaced, and a population model was run with these numbers entered as birds failing to breed successfully in order to assess the impact of displacement on productivity. The predicted impacts were considered to be small and spread across large populations and so of negligible concern (Trinder et al. 2012). A number of precautionary assumptions were made in this assessment, including that only one bird from any given pair is using the development site, and that displacement from the development site of one member of the pair is sufficient to cause a failed breeding attempt (Trinder et al. 2012). This approach developed through dialogue with Marine Scotland and SNH (Trinder et al. 2012) is rather different from the approach suggested by Natural England and JNCC (NE & JNCC 2012).

7. THE SPECIFIC CASE OF CREYKE BECK A & B

The area occupied by Creyke Beck A and B Offshore Wind Farms represents ca. 0.1% of the total area of the North Sea (this total is around 750,000 km^2). Given the distance of this site from major seabird colonies, it is likely that for most seabird species a high proportion of the birds present in the breeding season are nonbreeders. The European Seabirds-at-Sea (ESAS) database indicates that outside the breeding season, and especially in winter, common guillemots and razorbills are distributed across much of this total area, though with somewhat higher concentrations along the east coast of Britain from the Moray Firth to the Dogger Bank, the German Bight, and in the Skagerrak and Kattegat (see for example maps in Stone et al. 1995). Concentrations in the Moray Firth have been associated with concentrations of sprats, while those in Skagerrak and Kattegat have been associated with concentrations of young herring and sprats. Elsewhere, areas with high densities of sandeels are also likely to represent high quality foraging habitat, since auks can feed on sandeels at the sea bed even during winter. Compared to natural fluctuations of fish abundance which can typically vary by a factor of as much as ten-fold from year to year, the loss of 0.1% of foraging habitat appears trivial. If this habitat is poor quality then this conclusion is even stronger.

Areas in which the proportion of 'coarse sands' is high are more likely to constitute suitable sandeel habitats than those characterised by a high proportion of 'silts and fine sands' (Greenstreet et al. 2010). In the majority of stations sampled within Creyke Beck A and B Offshore Wind Farms the percentage contribution of 'silts and fine sands' was comparatively higher than the proportion of 'coarse sands' (Environmental Statement Chapter 13, para 4.7.12, and Figure 1). On this basis, Creyke Beck A and B are not areas that would normally be expected to hold high densities of sandeels.

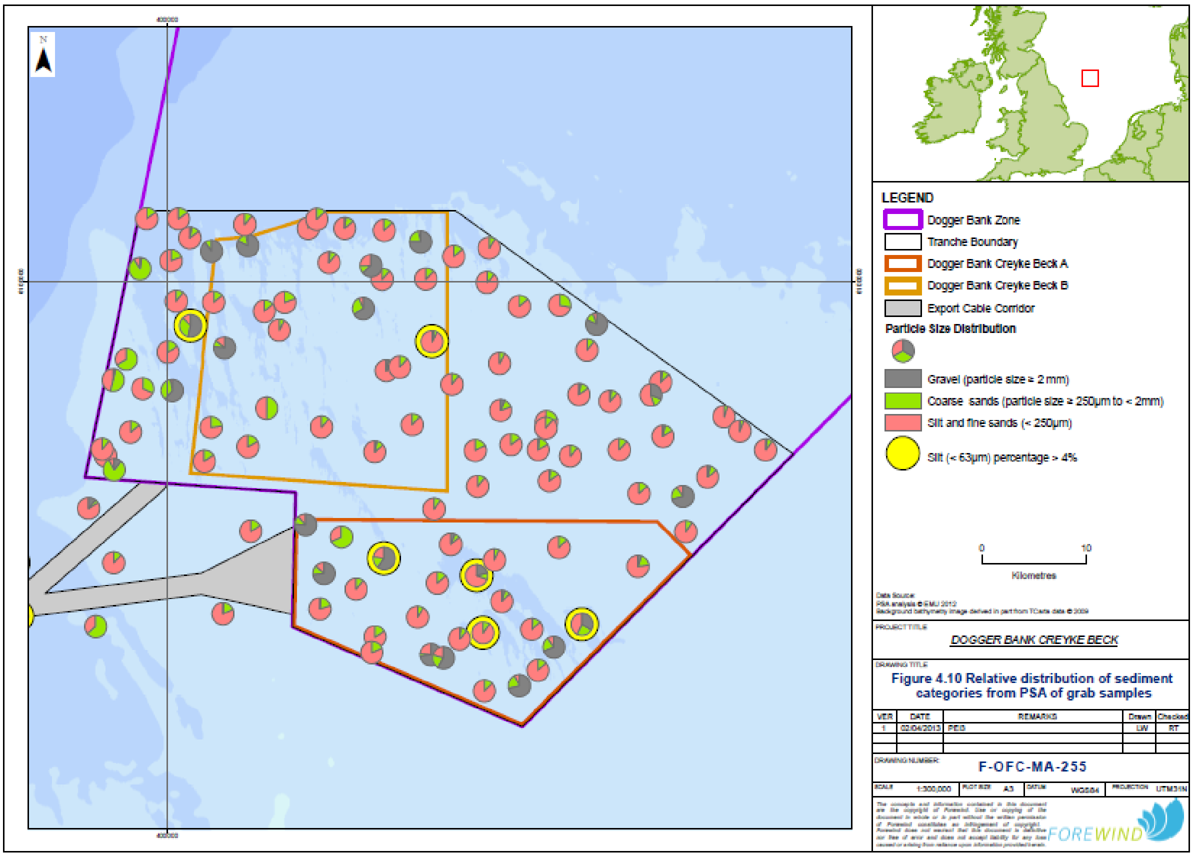

Figure 1. Sediment type within the development areas, showing the high proportion of silt and fine sand which is relatively unsuitable for sandeels, and low proportion of the coarse sand preferred by sandeels. Figure taken from Environmental Statement Chapter 13,Figure 4.10.

Data on the distribution of Danish sandeel industrial fishing effort support this. The fishing effort has been low in the areas of Creyke A and B, but high in areas with suitable sandeel sediment elsewhere on the Dogger Bank (Figure 2).

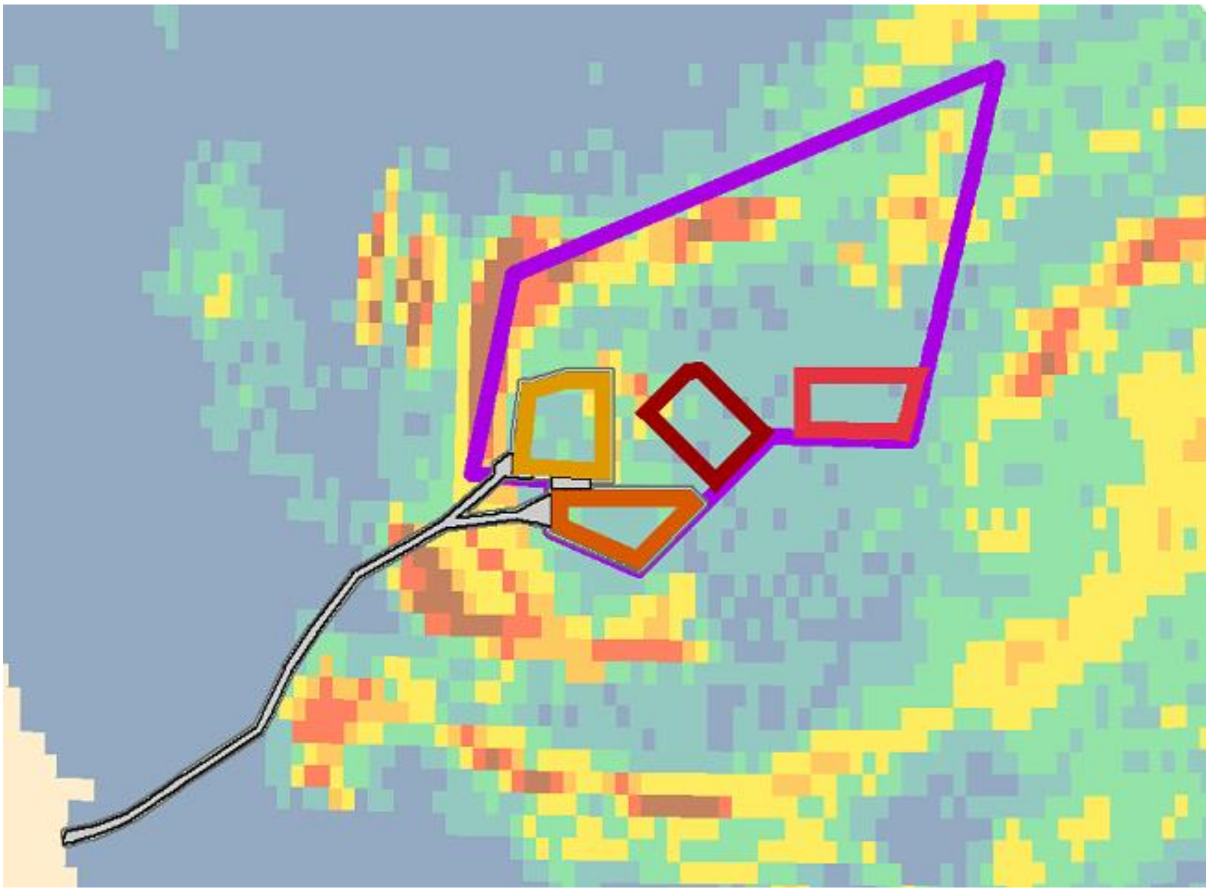

Figure 2. Danish sandeel industrial fishery VMS data by statistical rectangles (brown pixels = high effort, yellow = moderate effort, blue = low effort) showing the low average sandeel fishing effort, and therefore low catches taken in the development areas Creyke Beck A (brown triangle), Creyke Beck B (tan rectangle), Teeside A (pink rectangle) and Teeside B (maroon rectangle). Areas with low sandeel fishing effort (and therefore low catches) are likely to be less suitable foraging habitat for guillemots and razorbills than areas with high sandeel catches (reflecting high abundance of sandeels leading to high fishing effort).

It is therefore reasonable to conclude that Creyke A and B are very unlikely to represent high quality foraging habitat for common guillemots and razorbills, although the relatively shallow water depth there may be more suitable than in some parts of the North Sea. This implies that displacement of nonbreeding common guillemots and razorbills by Creyke Beck A & B Offshore Wind Farms will have a negligible impact on the survival rates of these birds, which can be expected to redistribute across other areas of the North Sea, where there is a huge total area of suitable habitat likely to be of similar (moderate) quality to that present within Creyke Beck A & B. Displacement of breeding common guillemots or razorbills from Creyke Beck A & B, if it does occur, would be unlikely to increase mortality of these populations, since life history theory predicts that breeding failure would be expected to occur before adults compromised their chances of survival. Displacement of breeding common guillemots or razorbills might increase breeding failure of these birds, although in view of the very long distance of breeding colonies from Creyke Beck A & B Offshore Wind Farms, it seems unlikely that common guillemots or razorbills would commute to these wind farm areas to feed, and that any breeding birds travelling so far while breeding would already be likely to experience breeding failure due to the time constraint of such long foraging journeys (Langton 2013).

For some other species, displacement represents an even smaller part of the nonbreeding distribution. For example, gannets from British colonies winter over the southern North Sea, Celtic Sea, Bay of Biscay, Iberian continental shelf, western Mediterranean, and continental shelf of west Africa (Fort et al. 2012). This represents a foraging area for gannets in winter considerably in excess of 3 million km^2. Displacement of gannets by Creyke Beck A & B Offshore Wind Farms, if it does occur, would represent habitat loss of less than 0.03% for these nonbreeding gannets. Atlantic puffins, although closely related to common guillemots and razorbills, feed on a more varied diet in winter, with zooplankton included as well as small fish, and feed mainly in the upper layers of the sea rather than diving deep. As a result, their habitat preference will not be as strongly related to distribution of sandeels. However, as with gannets, Atlantic puffins appear to have a very large potential distribution in winter. While puffins from North Sea breeding colonies may remain within the North Sea in winter, some move into the North Atlantic, where they may be distributed from inshore waters of the UK across to Newfoundland, and as far south as the Azores and into the Mediterranean Sea (Forrester et al. 2007; Harris et al. 2010). Forrester et al. (2007) describe puffin winter distribution as '*at low density over vast areas of northern seas*'. Displacement of wintering puffins from Creyke Beck A & B Offshore Wind Farms, if it does occur, would represent habitat loss of less than 0.01% of their winter distribution.

In exceptional years when the abundance of key prey (sandeels, sprats, young herring) are at exceptionally low levels for the North Sea, increased mortality rates of common guillemots and razorbills can be expected as a consequence of scarcity of food, and offshore wind farms could then exacerbate that mortality if they cause displacement of auks from foraging habitat. While the amount of habitat from which birds may be displaced by a single development is trivial in the context of the North Sea as a whole, cumulative effects of developments may be significant if many are in high quality habitat. However it is notable that where modelling of such cumulative impacts has been conducted, for what is considered to be one of the most sensitive species (red-throated diver) (Topping & Petersen 2011), the effects were predicted to be small.

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