Mainstream Neart na Gaoithe

Offshore Wind Farm

Ornithology Technical Report

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Contents

1 Introduction

Mainstream Renewable Power was awarded the wind farm development rights for an area of seabed lying approximately 15 km off Fife Ness, off the east coast of Scotland in February 2009. The development is known as Neart na Gaoithe.

Mainstream Renewable Power commissioned Natural Research (Projects) Ltd, Cork Ecology and Bureau Waardenburg to undertake studies of birds to inform an assessment of the effects due to the proposal. Baseline seabird surveys commenced in November 2009. This Technical Report presents details of the study methodology, together with baseline results from Years 1 and 2 of the seabird and marine mammal survey work. In addition, an environmental impact assessment, based on results from Years 1 and 2 is also presented.

This Technical Report (Appendix 12.1) is in support of the Neart na Gaoithe Environmental Statement. Appendix 12.2 presents detailed information about Distance sampling methods and collision risk modelling techniques used in the impact assessment for the Neart na Gaoithe Environmental Statement.

The offshore site is taken to be the proposed wind farm footprint and forms the core part of the Survey Area covered by the baseline characterisation surveys (Appendix 12.1). The Survey Area consists of the offshore site buffered to 8 km. This was surveyed monthly using the ESAS boat-based survey method (Camphuysen *et al*., 2004) from November 2009 to October 2011.

This chapter describes the methods used to establish the bird interest of the offshore site and surrounding buffer together with the process used to determine Nature Conservation Importance of the species and populations present. The ways in which birds might be affected by the offshore site are explained and the magnitude of the probable impacts of the scheme considered. Finally the significance of any identified impacts is assessed.

The aims of this chapter are to support the application for the Marine Licence for the proposal. It does this by presenting reasoned arguments supported by evidence to predict how the offshore site will most likely affect bird populations of different species using or passing through the site. The chapter's aims include consideration of whether predicted likely impacts should be judged to be significant or not according to the Electricity Works (Environmental Impact Assessment) Scotland Regulations (2008). Within the context of EIA Regulations, an ecologically significant impact is '*an impact that has a negative or positive effect on the integrity of a site or ecosystem and/or conservation objectives for habitats or species populations within a given geographical area.*' (IEEM 2010). Information to determine whether the offshore site is likely to have a significant effect on a European designated site, and thereby require Appropriate Assessment to ascertain that it will not adversely affect the integrity of such a site, is presented in Chapter 11 Nature Conservation. It should be noted that the meaning and use of the term 'significant' differs between its usage in the context of the EIA Regulations and Habitat Regulations (IEEM 2010).

For the purposes of predicting the impact of effects, analyses are restricted to the area plausibly affected. In the case of direct habitat loss and collision mortality this is taken to be the offshore site only. In the case of indirect habitat loss (displacement, disturbance and barrier effect) this is taken to be the offshore site and a surrounding buffer area. In assessing the impact of predicted effects and judging their significance a wide range of contextual information is used. In particular, the national datasets on seabird colony counts (SMP, 2012) and the published results of ESAS results for the North Sea (Stone *et al.*, 1995, Skov *et al.*, 1995). Nevertheless, these ESAS publications are based on data collected approximately 20 years ago and are sometimes incomplete in their coverage. Such limitations are noted and taken into consideration.

1.1 Effects Assessed

Through scoping (MRP 2009) the main effects on birds from the offshore site were considered likely to arise from:

- Construction activities;
- Operational activities, including turbine function and maintenance;
- The contribution of the offshore site towards cumulative impacts generated by other developments in the context of the elements above;
- Decommissioning activities.

The following types of potential impacts resulting from the offshore site on birds have been considered:

- Direct habitat loss due to land-take by turbine bases and ancillary structures;
- Indirect habitat loss due to displacement of birds from potential marine foraging areas as a result of construction and maintenance activities (e.g., disturbance from vessels), or due to the presence of operation turbines;
- The extent to which the offshore site acts as a barrier to the free movement of birds (in flight or swimming) through the area.
- Collision with rotating turbine blades, (i.e., killing or injury of birds);
- Indirect effects on birds through changes to their prey (e.g., small fish);
- The beneficial contribution made by the Project towards countering climate change. Uncertainties regarding climate change predictions mean that it is not possible at present to carry out a quantitative assessment of these effects on birds. However, climate change is widely perceived to be the single most important long-term threat to the global environment, particularly to biodiversity and birds. Thus, the continued rise in mean global temperatures is predicted to affect the size, distribution, survival and breeding productivity of many British bird species (Leech and Crick, 2007). For example, Zockler & Lysenko (2000) predicted a reduction in the breeding range of Arctic species of between 5 % and 93 % depending on the species. It has been estimated that 84 % of migratory species face some threat from climate change (Robinson et al., 2005).

2 Guidance, Legislation and Policy Context

2.1 Guidance

The following guidance has been consulted:

- Band, W., M. 2011. Using a collision risk model to assess bird collision risks for offshore windfarms. Final version, September 2011. SOSS, The Crown Estate, 37pp. http://www.bto.org/science/wetland-and-marine/soss/projects.
- IEEM 2010. Guidelines for ecological impact assessment in Britain and Ireland. Marine and Coastal.
- King *et al*., 2009. Developing Guidance on Ornithological Cumulative Impact Assessment for Offshore Wind Farm Developers, Cowrie;
- Maclean *et al.,* 2009. A Review of Assessment Methodologies for Offshore Windfarms, Cowrie;
- SNH, 2005a. Scottish Natural Heritage Guidance: Survey Methods for Use in Assessing the Impacts of Onshore Windfarms on Bird Communities;
- SNH, 2005b. SNH Guidance: Cumulative Effect of Windfarms;
- SNH, 2006. SNH Guidance: Assessing the Significance of Impacts from Onshore Windfarms on Birds outwith Designated Areas;
- SNH, 2010. SNH Guidance: Use of Avoidance Rates in the SNH Wind Farm Collision Risk Model.

2.2 Legislation

The following legislation was taken into account during this assessment:

- Electricity Works (Environmental Impact Assessment) (Scotland) Regulations 2008;
- The Council Directive on the Conservation of Wild Birds 2009/147/EC (EU Birds Directive);
- The Wildlife and Countryside Act 1981 (as amended) (WCA);
- The Conservation (Natural Habitats & c.) Regulations 1994 (as amended); ('The Habitats Regulations');
- The Nature Conservation (Scotland) Act 2004 (amended).

2.3 Planning Policy and EIA Context

The Planning Policy Context is presented in Chapter 6.

The evaluation approach is set in the context of:

- The statutory requirements of the Electricity Works (Environmental Impact Assessment) Scotland Regulations 2008, which define the information to be supplied within an ES;

- Scottish Planning Policy (The Scottish Government 2010) which includes guidance on how planning applications are to be considered; and
- PAN 58, Environmental Impact Assessment (Scottish Executive 1999) which includes general guidance on EIA.

Of particular pertinence to the current assessment is the requirement set out within the Electricity Works (Environmental Impact Assessment) Scotland Regulations 2008 – (Regulation 4(1)) to report:

- A description of the likely significant effects of the development on the environment;
- The main alternatives studied; and
- An indication of difficulties encountered.

Whilst considering a range of potential outcomes that could arise from implementation of the development, the assessment reports the effects that were considered to be most likely. It is these likely effects that the applicant is obliged to report, and that Scottish Ministers are obliged to consider – (Regulation 3(1), 4(1)).

The underlying approach comprises:

- Gathering and characterising baseline data;
- Characterising impacts that are predicted to occur as a result of the development;
- Evaluating the significance of the predicted impacts on the species' population at an appropriate geographical scale;
- Where significant effects are likely, to propose mitigation measures; and
- Re-evaluating the significance of effects after taking mitigation into account to determine likely residual effects.

In accordance with Regulation 4(1) (Schedule 4 Part I, 6), elements of uncertainty encountered in making the Environmental Assessment are identified along with the measures taken to reduce the level of uncertainty, assumptions made, and a commentary as to the likely extent that such uncertainties could affect assessment conclusions.

The level of certainty of predicted impacts varies depending upon a range of parameters and assumptions. With regard to the details of the offshore site itself (its size, turbine specifications, duration, construction and maintenance schedule etc.) the worst-case is assumed for any particular element within the range described by the Rochdale Envelope. With regard to possible impacts (e.g. displacement and collision mortality) a worst-case is not necessarily assumed where there is evidence from operational wind farms indicating that this is unlikely. For some elements, it is relatively straightforward to assess the effects because the nature of the change is predictable and the likely response by birds is well understood leading to a high degree of certainty. However, other impacts are less straightforward to assess because there is greater uncertainty, either in context data, the likely response of birds or the receptor populations sensitivity. The approach taken is to base assessments on assumptions and scenarios that are considered to be the most likely but factoring in appropriate caution where there is uncertainty. Where there is substantial uncertainty a worst case scenario may be used but care is taken to temper this to prevent predictions becoming unrealistic. This is particularly the case where the magnitude of an impact is derived by multiple calculations involving a number of factors and there is a danger of worst case assumptions being compounded.

2.4 Designated Sites

The offshore site is not statutorily designated at international or national level for ornithological interests e.g., as part of the Natura 2000 site network. Nevertheless, as shown below, individuals from a number of SPA populations are likely to regularly use or pass through the proposed offshore site and therefore could potentially be impacted. Given the different Regulations governing assessment of potential impacts of proposed developments on Natura 2000 sites, these are considered separately in Chapter 11 (Nature Conservation).

The offshore site is within the typical foraging range of several seabird species breeding at Special Protection Areas (SPAs) in eastern Scotland and north-east England. Three SPAs are of particular interest as there is potential for relatively large numbers of birds from these SPAs to be regularly using the offshore site in the breeding season. These are Forth Islands SPA (this includes the Isle of May, Bass Rock and Craigleith) situated approximately 16 to 30 km (depending on the colony concerned) to the south west, St Abb's Head to Fast Castle SPA situated approximately 31 km to the south and Fowlsheugh SPA situated approximately 62 km to the north.

For seabird species with particularly large foraging ranges (in particular fulmar) there is also potential for breeding birds from more distant SPA colonies to be affected. There is also potential for birds from SPA populations to be impacted during the non-breeding period if they overwinter in the vicinity of the offshore site or pass through on migration. In these cases, the birds affected could potentially be birds that breed at SPAs even further afield, for example, Arctic tern and Arctic skua breeding at SPAs in Shetland. There is also a theoretical collision risk to some land bird species (e.g., wildfowl and wader species) that are qualifying interests at certain SPAs designated for their importance for over-wintering or passage aggregations if they pass through the offshore site during migration.

Although impacts on designated sites (including SPAs) should be considered as part of the EIA Regulations, this can be subsumed under the 'higher authority' of the Habitat Regulations for SPAs. Therefore, the assessment of predicted impacts arising from the development on SPAs is deferred to Chapter 11 (Nature Conservation) and dealt with within the information presented for Habitat Regulations Assessment.

2.5 Data Sources

The following data sources have been consulted:

- SNH SiteLink web pages (online information on designated sites);
- UK Biodiversity Action Plan (BAP) (www.ukbap.org.uk);
- Birds of Conservation Concern (BoCC) 'Red list' (Eaton et al, 2009);
- SMP online seabird colony database
- JNCC online SPA site information

- Seabird data from Regional Seas 1 and 2 from ESAS database
- Published papers and unpublished reports providing information on bird status, ecology and response to wind farms and other developments.
- Neart na Gaoithe baseline survey results

3 Methods

3.1 Baseline Surveys

The methods used for the two years of baseline seabird and marine mammal surveys followed standard COWRIE approved survey methodology (Camphuysen *et al.* 2004). Seabirds and marine mammals were recorded using an adaptation of the standard Joint Nature Conservation Committee (JNCC) Seabirds at Sea survey method, which uses line transect methodology (see Webb & Durinck 1992 for further details).

A series of transects running in a north-west to south-easterly direction across the Neart na Gaoithe study area and spaced 2 km apart were surveyed each month (Figure 1). Birds were counted ahead of the ship and out to one side of the survey vessel in a 90° arc, with a 300 m transect width, using two surveyors, as per Camphuysen *et al.,* (2004). Three ESAS accredited surveyors were on board for the majority of surveys, apart from between November and March of Year 1, when only two ESAS surveyors were on board. At any one time, one surveyor was acting as the primary observer, with a second acting as scribe and secondary observer, while the third surveyor was on a break.

Binoculars were used to confirm identifications as well as to scan ahead for species such as red-throated divers, which are easily disturbed and take flight at some distance from the approaching vessel. Birds on the water were assigned to distance bands (A = <50 m, B = 51-100 m, C = 101-200 m, D = 201–300 m, E =>300 m), according to their perpendicular distance from the ship's track. A snapshot method was used for flying birds, which takes the ship's speed into account and prevents overestimation of seabird densities. In addition, the estimated height of flying birds was also recorded, to the nearest 5 m. The count interval for surveys was 1 minute intervals, and synchronised GPS recorders were used to record the vessel position every minute. Any marine mammals and uncommon bird species seen on the 'non-survey' side of the vessel were also recorded. All terrestrial bird species seen were also recorded.

Marine mammals (seals and cetaceans) were recorded concurrently with the seabird surveys. Sightings were recorded using the same methodology as for birds on the water. Species, number of animals, direction of travel and behaviour were recorded. In addition, the angle of the sighting was estimated using an angle board and the radial distance was estimated either using a range finder or a visual estimate in metres, if no horizon was visible. All marine mammals and other marine species such as basking sharks were noted during surveys, regardless of the distance from the vessel.

Surveys were conducted on the *M.V. Fleur de Lys*, which has a custom-built surveyor platform with an eye-height of greater than 5 m, as recommended for ESAS surveys (Webb & Durinck 1992, Camphuysen *et al.* 2004). In Years 1 and 2, surveys were conducted by Simon Pinder, Ailsa Reid, Richard Schofield, Caroline Weir, Stuart Murray, Digger Jackson, Ewan Wakefield, Andy Sims, John Clarkson, Tim Sykes, Rachel Coombes and Jon Ford.

Environmental conditions such as wind direction and force, sea state, swell height and visibility were recorded every 15 minutes throughout survey days. Surveys were carried out in good weather where possible, to maximise detection rates of birds and marine mammals on the water. Surveys were halted if the sea state exceeded sea state 4, as recommended in Camphuysen (2004).

Within the Neart na Gaoithe study area, there are two components; the offshore site and the surrounding Buffer area, which extends out to 8 km (Figure 3.1).

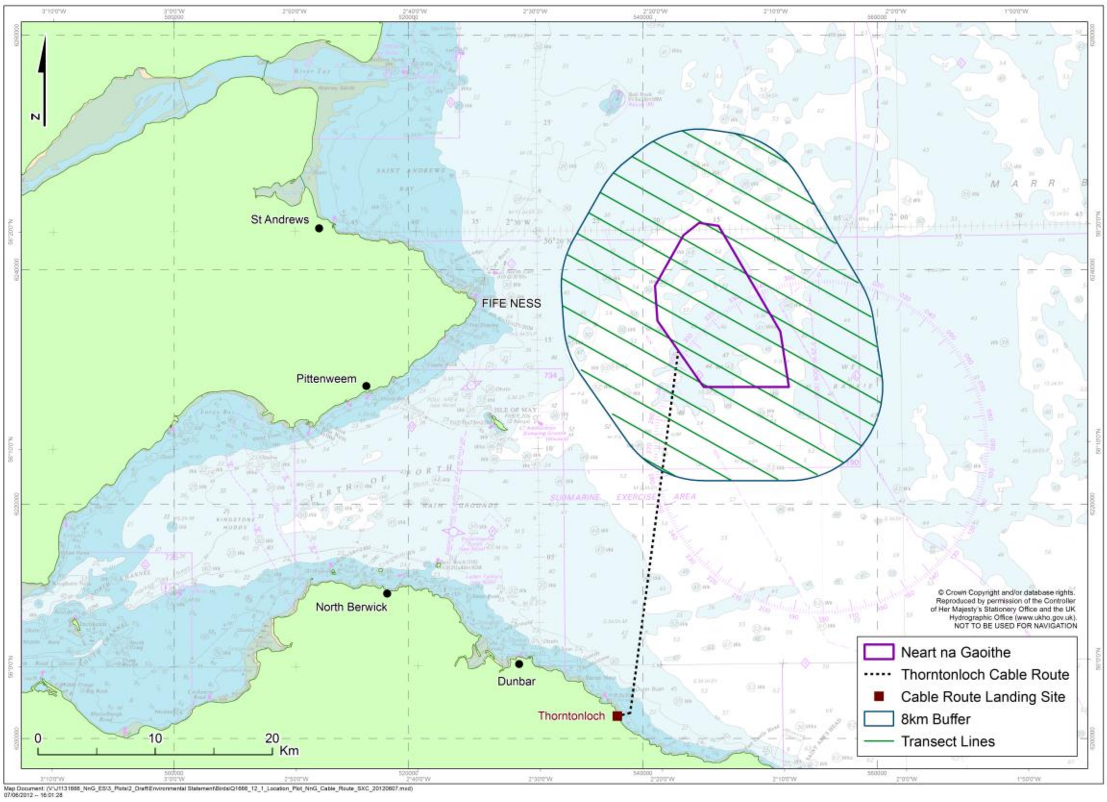


Figure 3.1 Neart na Gaoithe Study Area

3.2 Data analysis methods

All data were entered onto a Paradox database using the JNCC Seabirds at Sea Team data-entry program, then printed and manually checked for any errors before the analysis of the data was conducted.

This data formed the basis for estimating population sizes and densities of seabirds in the study area. These estimates were derived by applying Distance sampling techniques using Distance 6.0 software. Further details on this technique and associated corrections in relation to the baseline survey data are discussed in Appendix 12.2.

In addition, collision risk modelling was also carried out to inform this assessment. Details of this work are also presented in Appendix 12.2.

3.3 Impact Assessment Methods

3.3.1 The Approach to Impact Assessment

Impact is defined as change in the assemblage of bird species present as a result of the proposed Development and can be adverse, neutral or favourable. Change can occur either during or beyond the life of the proposed Development. Where the response of a population has varying degrees of likelihood, the probability of these differing outcomes is considered.

Effects are judged in terms of magnitude in space and time (Regini, 2000). There are five levels of spatial effect (Table 3.1) and four levels of temporal effect (Table 3.2).

Table 3.1 Scales of Spatial Magnitude

Significance of Impact	Description
Very High	Total/near total loss of a bird population or productivity due to mortality or displacement or disturbance. Guide: >80% of population affected, >80% change in mortality or productivity rate.
High	Major reduction in the status or productivity of a bird population due to mortality or displacement or disturbance. Guide: 21-80% of population affected, 21-80% change in mortality or productivity rate.
Moderate	Partial reduction in the status or productivity of a bird population due to mortality or displacement or disturbance. Guide: 6-20% of population affected, 6-20% change in mortality or productivity rate.
Low	Small but discernible reduction in the status or productivity of a bird population due to mortality or displacement or disturbance. Guide: 1-5% of population affected, 1-5% change in mortality or productivity rate
Negligible	Very slight reduction in the status or productivity of a bird population due to mortality or displacement or disturbance. Reduction barely discernible, approximating to the “no change” situation. Guide: <1% population affected, <1% change in mortality or productivity rate

The duration of an impact is defined as the time over which the impact is expected to last prior to recovery or replacement of the resource or feature and is defined with respect to ecological characteristics relevant to the species under consideration (IEEM 2010) (Table 3.2).

Table 3.2 Scales of Temporal Magnitude

Significance of Impact	Description
Permanent	More than approximately 30 years. Substantially greater than the life span of the longest lived individuals and corresponding to many generation times.
Long term	Approximately 10 - 30 years. This duration broadly corresponds to the maximum longevity of individual adult seabirds, waders and wildfowl and typically would represent several generation times.
Medium term	Approximately 3 - 10 years. This duration broadly corresponds to age of first breeding for seabirds, waders and wildfowl and typically would represent approximately one generation time
Short term	Up to approximately 3 years. This duration is substantially less than the average generation time for most seabirds, waders and wildfowl.

The potential nature conservation importance of an avian receptor (i.e. a potentially affected bird population) is determined within a defined geographical context (SNH 2006).

In the case of non-designated sites, magnitude is assessed in respect of an appropriate ecological unit. International, national and regional importance are used as frames of reference, following best practice guidance (IEEM 2010), and adapted to meet local circumstances. Given SNH (2006) advice, the top three geographical tiers (international, national and regional) are the most important within the context of the wind farm developments. The classification is hierarchical; therefore, species that would qualify under more than one category are defined according to the highest class.

For breeding populations there are generally very good estimates of international and national population sizes (e.g., Mitchell *et al*. 2004). However, there is no accepted or officially endorsed division of the UK coastal waters into regions for the purpose of defining regional seabird populations. This matter has been discussed between FATOWDG and SNH. SNH have advised that '*regional populations should be defined according to species ecology'*. For the purposes of this assessment, it is taken to mean that the size of a region defined for a breeding species should be broadly proportional to the size of that species' breeding season foraging range. Therefore, the defined regions for species with particularly large foraging ranges should be larger than those with smaller foraging ranges. The definition of regions used for assessment purposes below attempts to heed SNH's advice whilst at the same time be in keeping with the approximate geographic scale of the normal use of the term.

With the exception of fulmar and gannet, the regional breeding populations of seabird species is defined as that comprised by all birds breeding between Peterhead in northeast Scotland to Blyth in Northumberland. Any division of the UK coast into seabird regions is essentially arbitrary as the species concerned have very wide breeding ranges. The choice of Peterhead to Blyth was based on the desire to use a broadly natural and discrete geographical division, and to place boundaries at where there are relatively large gaps in the distribution of large colonies. For example, south of Northumberland there are no large seabird colonies for a distance approximately of 150 km, until the large colony at Bempton in Yorkshire. North of Peterhead there is a gap of approximately 40 km with no seabird colonies, before the large colonies at Troup, Pennan and Lion Heads. The colonies at Troup and Pennan and Lion Head are clearly part of the Moray Firth, which is an obvious discrete very large geographical area with many associated breeding colonies and which therefore merits treatment as a region in its own right. For fulmar, a species with extremely large foraging range the regional population is defined to include all areas within the mean maximum foraging distance of 400 km from the offshore site (Fair Isle to Bempton Cliffs) (Thaxter *et al.,* 2012).

The size and definition of seabird populations outside the breeding season is less straightforward as at these times many species have wide ranging nomadic lives offshore. For most species, information on the numbers occurring on passage or during the non-breeding period is relatively poor. Following advice from SNH to define seabird regional populations on the basis of species ecology, the appropriate reference populations for non-breeding seabird populations have been taken to be the subdivisions of the North Sea population as considered appropriate for that species depending on its ecology and observed spatial patterns of distribution and density. The distribution maps and population estimates consulted were derived mainly from JNCC ESAS survey data e.g., Stone *et al.* (1995) and Skov *et al. (*1995). For most species, one or more of the sub-areas, and corresponding estimated population size, presented in Skov *et al. (*1995) were the basis of the definition of non-breeding period regions. The details of the definition of the non-breeding-period regions are presented in the accounts of individual species.

Where the available data allow, the conservation status of each potentially affected species is evaluated for the appropriate 'population'. For these purposes conservation status is taken to mean the sum of the influences acting on a population which may affect its long term distribution and abundance. Conservation status is considered to be favourable where:

- a species appears to be maintaining itself on a long term basis as a viable component of its habitats;
- the natural range of the species is not being reduced, nor is likely to be reduced for the foreseeable future; and
- there is (and will probably continue to be) sufficient habitat to maintain the species population on a long term basis.

3.3.2 Sensitivity

The sensitivity of the receptor population to the effect under consideration is taken into consideration during assessments (Table 3.3). Sensitivity may depend on the time of year that an effect occurs. For example, a species is likely to be more sensitive to displacement or barrier effects when under high time/energy stress such as when breeding. Seabirds that are temporally flightless whilst undergoing their annual wing moult may be more sensitive to disturbance from vessels.

Table 3.3 Criteria for assessment of sensitivity of bird populations

Receptor population Sensitivity	Definition
High	No capacity to accommodate the proposed form of change.
Medium	Low capacity to accommodate the proposed form of change.
Low	Some capacity to accommodate the proposed form of change.
Negligible	Receptor is likely to have tolerance to accommodate the proposed change.

3.3.3 Evaluation of Nature Conservation Importance

The Nature Conservation Importance of the bird species potentially affected by development is defined according to the highest category of qualification in Table 3.4.

Table 3.4 Determining Factors for Nature Conservation Importance

Importance	Definition
Very high	Species regularly present in internationally important numbers (>1% international population).
High	Species listed in Annex 1 of the EU Birds Directive. Breeding species listed on Schedule 1 of the WCA. Species present in nationally important numbers (>1% national population). Regular occurrence of >1% of an internationally designated population (i.e., from a SPA or Ramsar site).
Moderate	Other species on the Birds of Conservation Concern (BoCC) 'Red' list (Eaton *et al*., 2009). UK Biodiversity Action Plan species. Species on IUCN threatened species list. Regularly occurring migratory species, which are either rare or vulnerable, or warrant special consideration on account of the proximity of migration routes, or breeding, moulting, wintering or staging areas in relation to the proposed development

	Species present in regionally important numbers (>1% regional population).
Low	All other species not covered above.

3.3.4 Determining Significance under EIA

As this is a Section 36 project, the evaluation follows the process set out in the Electricity Works (Environmental Impact Assessment) (Scotland) Regulations 2008 (the 'EIA Regulations': SERAD, 2000a).

Where there is a potential impact on a bird population that forms part of the qualifying interest of an internationally (i.e., a SPA or Ramsar site) or nationally designated site (i.e., a SSSI) impacts are judged against whether the proposal could significantly affect the site population and its distribution.

In the case of a bird population that is not protected by an international or national designation then judgement is made against a more general expectation that the Development would not have a significant adverse impact on the overall population, range or distribution; and that it would not interfere significantly with the flight paths of migratory birds. In assessing the impacts, consideration is given to the international or national or regional population of the species as appropriate. Trivial or inconsequential impacts are excluded.

The assessment determines the potential impacts of the proposal and the likelihood of their occurrence. In judging whether a potential impact is significant or not, several factors are taken into account:

- the Nature Conservation Importance of the species involved;
- the magnitude of the likely impact;
- duration and reversibility of impact; and
- the sensitivity of the receptor population to the impact.

The significance of potential impacts is determined by integrating the assessments of Nature Conservation Importance, and magnitude and duration of impacts and the sensitivity of a population in a reasoned way. Inclusion of population sensitivity in the making of professional judgements on significance means that the population status and trend of the potentially affected species is taken into account. If a potential impact is determined to be significant, measures to avoid, reduce or remedy the impact are suggested wherever possible.

The criteria for determining the significance of impacts on birds are provided in Table 3.5. Impacts considered to be major or moderate significance are deemed to be significant in terms of the EIA Regulations.

Detectable changes in international, national or regional populations of high or moderate Nature Conservation Importance are considered to be fundamental effects and therefore significant under the EIA Regulations. Non-significant effects included all those which were likely to result in non-detectable changes in regionally or nationally important bird populations.

Table 3.5 Significance Criteria

Significance of Impact	Description
Major	Detectable changes in a receptor population that will have severe impacts on its conservation status.
Moderate	Detectable changes in a receptor population that will likely affect its conservation status.
Minor	Small or barely detectable changes that will unlikely affect the conservation status of a receptor population.
None	No or non-detectable changes in the conservation status of a receptor population.

Evaluation of effects on Natura 2000 network populations (e.g., SPA populations) also needs to take account of whether or not the conservation status of a species is favourable (in terms of the robustness of its population and the adequacy of its supporting habitats), and whether the proposal would add substantially to the difficulty of taking action to reverse any decline and enable the species to achieve Favourable Conservation Status (FCS).

It is recognised that the term 'Favourable Conservation Status' as articulated within the Habitats Directive is not used in the Birds Directive, conservation status is favourable where:

- Population dynamics indicate that the species is maintaining itself on a long-term basis as a viable component of its habitat;
- The natural range of the species is not being reduced, nor is it likely to be reduced in the foreseeable future; and
- There is (and will continue to be) a sufficiently large habitat area to maintain its populations on a long-term basis.

According to SNH (2006), an impact should be judged as of concern where it would affect the Favourable Conservation Status of a species, or stop a recovering species from reaching Favourable Conservation Status, at international, national or regional population levels.

3.3.5 Overall Significance

Where the proposal is predicted to have multiple impacts on a population, the significance of the various effects acting together on the receptor population is also considered.

The approach used in assessment of effects takes into consideration the likely response of the birds and thereby attempts to be a realistic, albeit cautious, prediction rather than a worst case prediction. For example, far field displacement is factored into the assessment of collision risk. An advantage of this realistic-but-cautious approach is that it allows the overall impact of the proposal on a population to be estimated by summing the individual impacts of each effect. This would not be the case where worst-case scenarios are used as some effects act in mutually exclusive ways. For example, displacement and collision mortality, if all flying birds are displaced (the worst-case scenario treatment of displacement) it follows that none can be killed by collision.

3.4 Assessment calculations

3.4.1 Construction and Decommissioning effects

As well as considering the operational phase, the detailed assessments also predict the magnitude and significance of effects on bird populations during the construction and decommissioning phases of the proposed wind farm. Such effects on birds are generally of lesser magnitude and shorter duration.

The magnitude of collision mortality during construction and decommissioning phases is assumed to be negligible, as any turbines present will not be rotating. Therefore this risk is not assessed.

The response of birds to non-operational turbines, at various stages of completion is unknown. It is likely that in general they will show a smaller, possibility much smaller, displacement response than they do to operational turbines.

3.4.2 Operational Effects

Scoping identified three potential adverse effects likely to be important when turbines are operational: displacement, barrier effect, and collision mortality. A summary of these effects, where known, is included in the species accounts.

In addition a number of other potential effects were identified as likely to be of less importance, or in some cases beneficial. These were direct habitat loss, contamination, and indirect effects through changes to the marine ecosystem, e.g., beneficial effect on prey fish populations. These are considered in a generic way and do not form part of the individual species assessments.

All operational effects are likely to continue throughout the operational period of the wind farm. This is planned to be 25 years and so all effects are judged to be long-term effects. It is possible that displacement and barrier effects could diminish with time if birds habituate to the wind farm infrastructure. It cannot be assumed that habituation will occur, however it has been demonstrated for some species at operational offshore wind farms elsewhere (Zucco *et al.,* 2006).

All effects of the wind farm on birds would cease following decommissioning and are therefore considered to be reversible.

3.4.3 Definition of Assessment Periods

An obvious feature of the results of the baseline survey work is the strong seasonality in the numbers of a species present in the Survey Area. This reflects the timing of the breeding season and the movement of birds between areas where they breed, moult and spend the winter. The assessment of each species reflects this seasonality and balances the desire to keep the process relatively simple by using fewer periods and the desire to make best use of the data and take account of species' biology by using a larger number of periods. All periods are based on whole months.

For the majority of species two periods were used; the 'breeding period' and the 'non-breeding period'. In these cases the breeding period was defined as when the majority of breeding adults are strongly in attendance at breeding colonies (based on dates in Cramp and Simmons 1983) and the non-breeding period the remainder of the year. For guillemot and razorbill a 'chicks-on-sea period' was also used corresponding to the time when adults are no longer attending colonies but some adult males are attending dependent young on the sea; this was defined as the months of July and August (based on Cramp and Simmons (1983), and the dates when dependent young were recorded on surveys). For some species (kittiwake, guillemot, razorbill, puffin), where appropriate, a late summer 'post-breeding period' was also used. This broadly corresponds to the time when adult guillemots and razorbill and kittiwake are undergoing wing moult (Ginn and Melville 1983) and when particularly high densities of all four species occurred in the study area. In the case of the three auk species, the post-breeding period was defined as the months of September and October. In the case of kittiwake it was defined as August and September.

For species that are only present for part of the year (e.g., sooty shearwater, Arctic tern, little gull, lesser black-backed gull and little auk) the assessment was restricted to the period of the year when they were present.

For the species assessments, the mean number of birds present in the period under consideration for both baseline survey years was used.

3.4.4 Estimation of Potential Collision Mortality

Birds flying through the proposed development could potentially collide with rotating turbine blades and be killed or injured. The collision risk posed by the proposed development was quantified by Collision Rate Modelling (CRM) to predict the annual mortality of each species (Band 2000, Chamberlain et al. 2007). Data on flying bird density and flying height was derived from the monthly baseline boat-based surveys, and values for typical flight speed and bird size was obtained from published sources. Full details of the methods used to estimate collisions are presented in Appendix 12.2.

For CRM, the analyses of commonly occurring species is based on the density of birds flying through the offshore site during baseline surveys. Birds outside the offshore site were not considered as these are at no risk of collision.

CRM calculations were made for four hypothetical development design scenarios encompassing the range of wind farm scales that may be proposed (see Appendix 12.2). Results for the least adverse (64 x 7 MW turbines) and most adverse (128 x 3.6 MW turbines) design scenarios are considered in the assessments.

The assessment of the significance of the predicted collision mortality is based on examining the magnitude of the additional mortality against the published estimates of baseline adult mortality rate for species. Adult mortality rates were obtained from the BTO BirdFacts website (BTO, 2012). For two species, little gull and great black-backed gull, there is no published estimate of adult mortality rate. Therefore, the mortality rates for closely related species were used as surrogates. The mortality rate for black-headed gull was used for little gull, and the mortality rate for herring gull was used for great black-backed gull. Closely related species with similar ecologies typically have similar mortality rates, however this is not always the case and so using the mortality rate of or a surrogate species inevitably introduces a degree of uncertainty that needs to be borne in mind in reaching conclusions.

There is a theoretical collision risk to SPA populations of some land bird species (e.g., waders and wildfowl species) that potentially migrate over the offshore site. The relatively few (or in some cases total lack of) records of these species from the baseline surveys may not give representative information on their occurrence in the offshore site as some species may migrate at night and/or in large flocks that could pass through on days when no survey work took place. Therefore, the potential for collision risk was examined using theoretical scenario that approximates to the worst case.

Collision risk modelling was therefore undertaken for 15 species of geese and waders based on an assumed population of 1,000 birds of each species passing through the offshore site twice per year, on spring and autumn passage, with all birds flying at rotor height.

The assumption underlying the scenario assessed that all birds in a population will pass through the wind farm twice per year is unrealistic as all species are likely to migrate on a relatively broad front and so only some are likely to pass through the wind farm. This means that the conclusions based on this scenario will be cautious.

3.4.5 Estimation of Potential Displacement

Displacement is defined as the potential for the wind farm and associated human activities to reduce or prevent birds, including flying birds, from using the offshore site and is therefore akin to habitat loss. The assessment of displacement of flying birds transiting around through the offshore site instead of through it is considered under barrier effects.

Displacement is assessed in terms of how potentially important the area under consideration (the offshore site and an appropriate buffer) is to the receptor population. In this case, as the area entirely consists of open sea away from the immediate vicinity of breeding colonies, its major use is likely to be as a place to forage. Therefore, the question of displacement is assessed in terms of the effective loss or reduction of the food resources the area provides for a receptor population. Conventionally, importance of a site is indicated by the proportion of the receptor population present at the site, or as in this case, what proportion of foraging birds are present. Outside the breeding season all individuals of the seabird populations of interest live at sea (with the exception of some gull species). Therefore, the proportion of individuals in the population present in an area to be assessed should give a reasonable and unbiased measure of its importance for foraging. On this basis, for seabird receptor populations in the non-breeding periods (i.e., when all individuals live at sea), the importance of the offshore site (and an appropriately sized buffer) for foraging was based on the proportion of the receptor population that was on average present during the period under consideration. In all these cases, populations in the non-breeding periods are the total number of all birds (i.e., adult plus immature birds). For example, if the receptor population size is 10,000 birds, and baseline surveys showed that in the non-breeding part of the year there are on average 500 individuals present in the offshore site buffered to 1 km, then it would be estimated that this area is likely to provide approximately 5% of the populations foraging needs at this time of year.

If the same method is used for seabirds during the breeding season the estimated importance of the offshore site (and an appropriately sized buffer) for foraging would be significantly biased low, and therefore give an inappropriate estimate for assessment purposes. This is because at this time of year, at any one time, a high proportion of individuals (typically 50% - 70%) are not at sea, but are attending their breeding colony. This has to be taken into account when estimating foraging importance of an area based on the numbers of birds present. To give a reasonable and unbiased measure of the importance of an area for foraging during the colony-attendance part of the year the numbers of birds present needs to be expressed as a proportion of the population that is at sea, i.e., the average number that at any one time are away from the colony foraging. This is easily calculated if the mean colony attendance rate by adults is known. Published estimates of colony attendance rates derived from tagging and colour-ring studies were used.

The following simple example shows how foraging importance was calculated for seabirds in the colony-attendance period, using the method described above. The example is based on a hypothetical receptor population of 10,000 breeding adults that is known (from studies on the species) to have a mean colony attendance rate of 60%. Therefore, for such a population, it is assumed that on average, at any one time in the colony-attendance period, there are 6,000 adults present at the colony and 4,000 away at sea foraging. If, say, baseline surveys observed on average 200 adults in the offshore site buffered to 1 km during the colony-attendance period, then it would be estimated that this area is likely to provide approximately 5% (200/4,000) of the population's foraging needs at this time of year.

Tagging data, when available, can give an independent measure of the importance of the offshore site to breeding seabirds, assuming that the tagged individuals are representative. This assumption is unlikely to be entirely met as the number of tagged individuals is always small relative to the number of individuals in a colony. Tagging data can give precise information on the proportion of time spent foraging in the offshore site relative to other areas and information on the proportion of time spent actively foraging compared to commuting time. A number of tagging studies have been undertaken in recent years for priority seabird species at some of the major colonies in the region, notably on gannet, kittiwake, guillemot, razorbill and puffin (e.g. Daunt *et al.,* 2011a & 2011b). In any case, colony attendance and the proportion of time flying will vary from year to year and site to site due to other factors, for example availability of food (e.g. Monaghan *et al.,* 1994) and weather (e.g. Birkhead and Taylor, 1977), and this limits the scope to refine measures of the importance of the offshore site for foraging.

For receptor populations that are populations of breeding seabirds the published population estimates are based on colony counts and refer to (or translate to) the number of breeding adults. Therefore, they do not include non-breeding individuals. However, because seabird species have delayed maturity, with individuals typically not breeding for the first time until several years old, immature birds may be potentially be approximately as numerous as breeding adults. Estimating breeding season foraging importance from the apparent proportion of receptor population present could lead to a bias if there are significant numbers of non-breeding (immature) birds present and mixed with breeding birds. In species where immature birds can, on the basis of plumage, be distinguished from adult birds (e.g., gannet and gull species) it is straightforward to take into account any immature birds seen in analyses. Indeed, the potential bias caused by the presence of immature birds for these species was accounted for in analyses of estimated importance of the offshore site for foraging by simply excluding the immature birds from the totals used in calculations.

For seabird species in which immature non-breeding birds cannot be distinguished from breeding adults during baseline surveys (e.g., fulmar and auk species) the potential for bias remains and no attempt was made to correct for this. Put simply, for these species it is assumed that all birds present in the breeding season in adult summer plumage were breeding birds. Although it is likely that some immature birds were present with breeding adults, including these in the analysis potentially causes the estimated importance of the offshore site for foraging to be bias high, and, as a consequence, leads to more cautious assessments. Although the actual magnitude of any such bias is unknown, evidence from ringing studies shows that a high proportion of immature fulmars, guillemots and razorbills spend the spring and early summer away from their natal area (Wernham *et al*., 2002), suggesting that for these species at least the size of any bias is likely to be small.

In the case of displacement the birds at potential risk are assumed to be the estimated mean number present in the offshore site buffered to 1 km.

3.4.6 Estimation of Potential Barrier Effect

The proposed development has the potential to act as a barrier to the free movement of birds, either flying or swimming, that under normal circumstances would choose to pass through the area occupied by the development. Such an effect has been observed for many seabird species at operational offshore wind farms, in particular by using radar to track flight routes (Pettersson 2005, Petersen *et al.,* 2006). A barrier effect causes displacement of birds, and to some extent this issue overlaps with the displacement of foraging birds from the offshore site discussed above. However, a barrier effect can potentially cause impacts further afield, and in any case is assessed here in terms of the effect it could have on the time and energy budget of foraging birds through causing them to make longer flights between breeding colonies and foraging locations. For these reasons it is considered separately.

There are two consequences of the barrier effect. First, it could reduce birds' access to areas containing resources they would otherwise exploit, for example to feeding grounds (assessment of this has already covered within displacement). Second, a barrier can cause birds to undertake detours to reach areas that they would otherwise travel directly to and from. Undertaking a detour affects time and energy budgets, and this could have a knock-on effect on their survival and breeding success if it occurs at times when birds are under stress, for example when provisioning young (Masden *et al.,* 2010).

The scale of the potential barrier that the proposed development would present is examined in terms of its size in relation to, and distance from the four closest large seabird breeding colonies (Table 3.6), namely Isle of May, Bass Rock, Craigleith and St Abb's Head. The potential for flights from colonies more than 50 km away to be detoured was not examined because for most species beyond 50 km, relatively few individuals are likely to be affected and for all species the size of any detour required to go around the wind farm would be small compared to the length of the foraging trip.

Table 3.6 The size of and distance to the potential barrier formed by the proposed wind farm, with respect to major seabird colonies, and the percentage of heading directions that are potentially affected

Colony	Distance from offshore site (km)	Barrier width (km)	Compass degrees available	Degrees affected by barrier	% blocked
Isle of May	16.0	17.9	132	43	33%
Craigleith	32.0	17.8	109	30	28%
Bass Rock	27.0	17.8	111	30	27%
St Abb's Head	33.0	11.6	210	19	9%

The size of the barrier presented to birds at each of these colonies is assumed to be the linear width of the barrier measured at right angles to a flight on a heading towards the centre of the proposed wind farm. The width of the barrier was assumed to be the width of the offshore site with a 1 km buffer either side, this buffer width being considered to be larger than the likely average far-field avoidance distance shown by birds that are affected and therefore likely to lead to cautious estimates. This choice of buffer size was informed by the typical closest approach distances observed for detouring birds from radar studies (e.g. Zucco *et al.,* 2006) and experience from observing flying seabirds avoiding other natural and man-made barriers. The proportion of flights potentially affected was estimated from the proportion of the compass sector (spread of directions) potentially available that would be blocked to birds from each colony wishing to undertake foraging trips further than the distance to the wind farm (Table 3.6).

The additional distance that birds affected by the barrier would need to fly from these colonies in order to access areas at a range of distances away was calculated for each colony. This was evaluated for hypothetical foraging locations immediately beyond the barrier (26 – 42 km depending on the colony) and for locations at 30, 40, 50, 60, 80, 90 and 100 km from each colony (Table 3.7). The size of detour a bird would be required to undertake depends on where along the front edge of barrier the bird initially approaches, e.g., a bird approaching the mid-point would be required to make a greater detour than one approaching near the end (Figure 3.2).

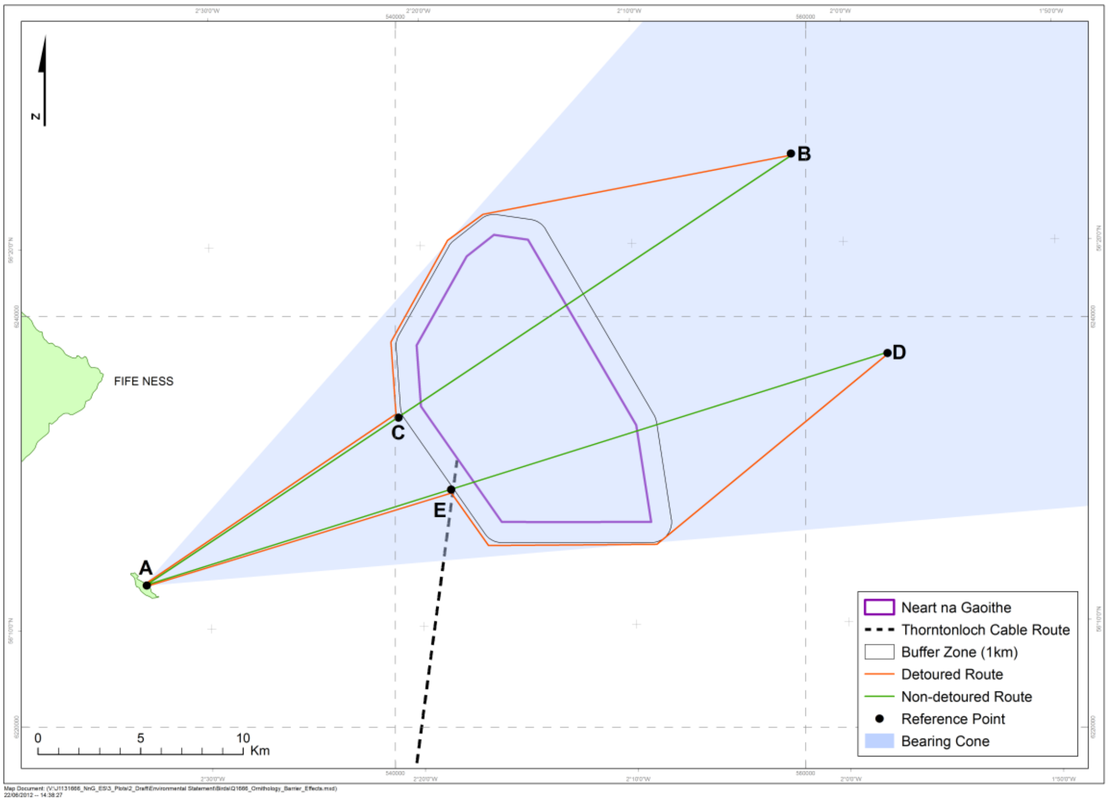


Figure 3.2 Schematic diagram showing how barrier effects were estimated

Figure 3.2 illustrates the theoretical situation for guillemots breeding on the Isle of May. Birds on foraging trip headings within the grey cone would be potentially affected by the wind farm acting as a barrier. In the absence of any barrier, the average bird in the northern part of the cone would fly directly from the colony (Point A) to feed at Point B, located 38 km away (the mean foraging distance) (Thaxter *et al.,* 2012). After the wind farm is constructed, the same average bird is assumed to perceive a barrier at Point C and respond by detouring around the northern perimeter of the wind farm to reach its intended destination at Point B. Similarly, the average bird affected in the southern half of the grey cone would be detoured at Point E around the southern perimeter of the wind farm to reach its intended destination at Point D.

The calculation was therefore based on the 'average detoured flight path', which was taken to be the path taken by a bird that encounters the barrier halfway between one of the ends and the centre of the front edge of barrier. Because the proposed development is an irregular shape the length of the 'average detoured flight path' was calculated for both the left and right halves of the barrier and the average of these two values taken. The size of detour is also affected by how close affected birds approach the wind farm before detouring and thereafter stay away from it; these were both assumed to be 1 km. The lengths of the 'average detoured flight path' were divided by the length of the corresponding direct flight path to give a measure of the detour expressed as a percentage of the direct route. It was assumed that the theoretical detour distance would be the same for both outward and return flights from the colony although it was only calculated for outward flights.

In assessing the likely effects of the proposed wind farm acting as a barrier for a particular species, the destination location beyond the barrier was assumed to lie at the mean foraging distance for that species from the colony (Thaxter *et al.,* 2012*)* where this did not correspond to one of the distances evaluated.

The extent to which flights by breeding seabirds will actually be affected by the proposed wind farm causing a barrier will depend on a combination of how birds perceive the development and where they choose to feed, and the extent to which there is spare capacity in the resources available in alternative areas. These are examined in the species accounts. It is worth pointing out three generalities:

- Foraging trips from colonies to locations that are less far away than the wind farm will not be affected;
- The birds that are potentially greatest affected are those that use feeding areas located a relatively short distance beyond the barrier; and,
- For birds foraging a long way beyond the barrier the additional distance or time of detoured flights is small compared to the length of the direct route.

The theoretical effects of wind farms forming barriers to breeding seabirds has been examined in detail for a range of species, including most of the species considered as priority to the current proposal (Masden *et al*., 2010). This study shows that there is potential for there to be significant effects for species with a high wing loading such as auks, especially puffin.

Table 3.7 Magnitude of barrier effect for 'average detoured flight path' from the four colonies examined for a range of destination distances from the colony

Colony	Direct distance (km)	Detoured distance (km)	% extra distance
Isle of May	25.6*	32.9	28.4%
	30	35.3	17.7%
	35	39.4	12.6%
	40	44.0	9.9%
	45	48.7	8.1%
	50	53.6	7.1%
	60	63.4	5.6%
	70	73.3	4.7%
	80	83.2	3.9%
	90	93.2	3.5%
	100	103.1	3.1%

* indicates the distance to the rear edge of barrier, i.e., the closest possible destination distance beyond the barrier.

Colony	Direct distance (km)	Detoured distance (km)	% extra distance
Craigleith	41*	46.3	12.1%
	45	48.9	8.7%
	50	53.2	6.4%
	60	63.3	5.4%
	70	72.6	3.7%
	80	82.6	3.2%
	90	92.6	2.8%
	100	102.6	2.6%

Colony	Direct distance (km)	Detoured distance (km)	% extra distance
Bass Rock	37.3*	42.1	12.7%
	40	43.5	8.7%
	45	47.7	5.9%
	50	52.4	4.8%
	60	62.2	3.6%
	70	72.0	2.9%
	80	82.0	2.5%
	90	92.0	2.2%
	100	102.0	2.0%

Colony	Direct distance (km)	Detoured distance (km)	% extra distance
St Abb's Head	45*	47.4	5.3%
	50	52.2	4.4%
	60	62.1	3.4%
	70	72.0	2.8%
	80	82.0	2.5%
	90	92.0	2.2%
	100	102.0	2.0%

3.4.7 Estimation of disturbance from vessels

The response of birds to boat traffic is relatively well understood and provided the amount of traffic and vessel speeds are moderate then the main seabird species that use the offshore site can be expected to show only small scale short term behavioural responses, similar to those observed on the site during baseline surveys in response to the approach of the survey vessel or a fishing boat. Therefore, the effects of additional boat activity caused by construction and decommissioning activities are likely to have a negligible and non-significant effect on all seabird species. A quantified assessment of the effects of construction and decommissioning boat activities on birds would require details of the number and size of vessels and when they would be present.

Disturbance from vessels has the potential to cause displacement of seabirds from foraging habitat and cause flying birds to detour their flight routes. Observations by NRP surveyors during baseline surveys of the offshore site and at many other sites in Scotland show that the response of flying seabirds to vessels are of very short term duration and spatially of small scale, amounting to a minor inconvenience at most. Therefore, the potential effect of vessel disturbance on flying birds is considered to be negligible and is not assessed. There is also no evidence in the published literature, or indeed suggestions from experienced observers, that vessels pose a significant collision hazard to flying seabirds. Consideration of vessel disturbance is therefore limited to its potential to displace birds from foraging habitat and disrupt their foraging behaviour. Although not limited to the construction and decommissioning phases, vessel disturbance is of particular relevance to these stages because of the relatively high vessel activity associated with the development that will occur at these times.

Displacement of birds from foraging habitat is assessed using the approach described earlier in which the importance of the area for foraging from which birds are displaced is estimated. In this case the area from which birds are displaced is assumed to be that defined by an appropriate buffer distance around a vessel or number of vessels (moving or stationary), rather than the offshore site as a whole. In doing this it becomes obvious that the potential area affected by vessel displacement at any one time will be relatively small compared to the size of the offshore site (Figure 3.3). It is also worth noting here that none of the regularly occurring common seabird species recorded in the offshore site are considered to have high susceptibility to disturbance (Langston 2010, Garthe and Hüppop 2004).

The displacement effects that might be caused by vessel disturbance were examined by simple modelling of a number of hypothetical scenarios. The model outputs are estimates of displacement in terms of foraging habitat loss from the offshore site (Figure 3.3). Separate models were undertaken for static vessels and vessels in transit. The basis of the models is the assumption that all individuals of a species are displaced from a disturbance zone around each vessel present. In the case of static vessels this zone was assumed to be a circle with a radius equal to one of the three buffer sizes (150 m, 300 m and 600 m). In the case of transiting vessels it was assumed to be a rectangle corresponding to a disturbance corridor. The width of this corridor was twice one of three specified buffer sizes (150 m, 300 m and 600 m). The length of the rectangle was the specified buffer distance plus a travel distance of 1,540 m. This is the distance travelled by the vessel (moving at 10 knots) in the assumed average time it takes for birds to recommence using an area after the vessel passes. The travel distance of 1,540 m is based on the assumption that the average vessel velocity is 10 knots and an assumed time for birds to resettle of five minutes. No published information on this subject could be found for the species of interest. Therefore, it was informed by the opinion of experienced NRP surveyors who have observed how the species of interest respond when disturbed by vessels. The choice of five minutes is a conservative interpretation of observing the response of seabirds (especially auk species) at the offshore site and in the Sound of Islay to disturbance by vessels. The buffers chosen for modelling were 150 m, 300 m and 600 m, these broadly corresponding to the expected typical upper response distances for species which might be categorised as having low, moderate and high susceptibility to vessel disturbance respectively. The reason for modelling three different buffer-size values was to illustrate how differences in susceptibility (e.g. between species) affects the amount of displacement that might occur. All the regularly occurring seabird species that forage in the offshore site are considered to have relatively low susceptibility to disturbance i.e., the predicted amount of displacement by the models would be the green line scenarios (150 m disturbance buffer) illustrated in Figure 3.3.

The potential amount of displacement that could result at a given time is the sum of the disturbance from static and transiting vessels.

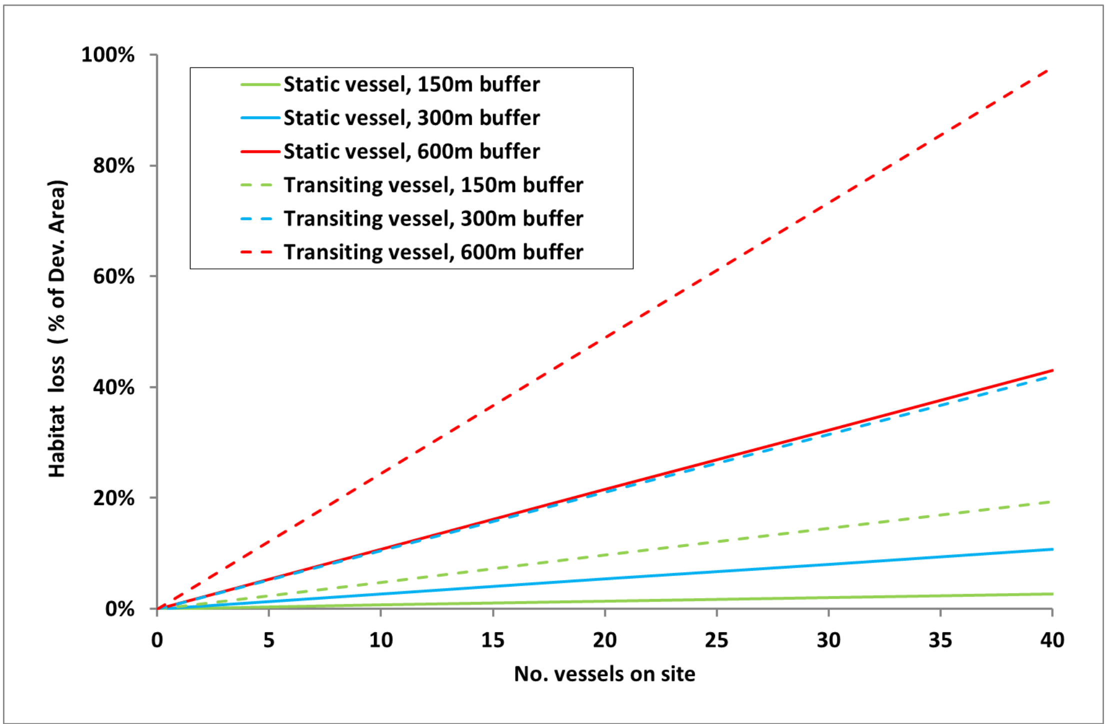


Figure 3.3 The hypothetical loss of seabird foraging habitat from the offshore site caused by disturbance from vessels

Vessel disturbance can also disrupt the normal behaviour of foraging seabirds, for example through causing birds to flush and relocate. This dynamic 'active' element to this disturbance is something that is specific to vessel disturbance (and the human activities on board, such as creating loud noises), especially transiting vessels, and is not relevant to the disturbance caused by, for example, wind turbines. The effects of disruption to behaviour could be assessed in terms of impacts to birds' time and energy budgets. However, given the low susceptibility of the seabird species that regularly forage in the offshore site and the very small proportion of the receptor populations that are expected to be affected by vessel disturbance at any one time (Figure 3.3), it is clear that the numbers of potentially affected and frequency that individuals would experience such active disturbance are both so low that it is not plausible that it could significantly affect populations. Therefore, it was concluded that attempting to quantify the effect on time and energy budgets was not merited.

3.4.8 Connectivity to Designated Sites

The proposed development is not located within areas of designated European importance for birds (i.e. SPAs). However, breeding seabirds forming part of the qualifying interest of several SPAs potentially forage in the offshore site during the breeding season on a regular basis. In addition, birds breeding at SPAs outwith the region may visit or fly through the offshore site at other times of year, either during on migration or during the moulting or wintering periods. Therefore, the proposed development has the potential to affect the qualifying bird populations at many SPAs.

To assess how displacement and collision effects could impact on an SPA population a measure of the extent of connectivity between birds from the SPA and the offshore site is required so that a reasonable estimate can be made of the importance of the offshore site to birds from the SPA. The quality of the available information on connectivity varies between species and between SPAs. Tagging studies of birds at the SPAs under consideration provide high quality information for some species, and this is used where possible. Where such high quality data are lacking for a particular SPA, then a generic approach is adopted, although this is informed by tagging studies from other sites.

The first question to address in determining the importance of the offshore site is what proportion of the birds present in the offshore site (and appropriate buffer) are likely to be from a particular SPA population. Once this can be evaluated then the numbers using the offshore site that are considered likely to be from a particular SPA population can be evaluated in terms of the proportion of that population they represent. Of particular relevance is the importance of the offshore site in terms of a foraging location. The numbers of birds using the offshore site (and buffer) during baseline surveys is used as a natural assay of its importance.

For assessment purposes, connectivity is quantified in terms of the proportion of birds that are on average present in the offshore site that are assumed to be from the SPA population under consideration. This is calculated according to the most appropriate of several generic scenarios examined (see below). The assumptions used in the scenarios (see below) are chosen so that the assumed level of connectivity is likely to be overestimated and therefore lead to cautious assessments. Calculating the strength of connectivity according to one of the scenarios has the advantage that connectivity is expressed as a quantitative measure and therefore enables quantitative impact assessments to be made; this is considered preferable to qualitative assessments.

For the Forth Islands SPA (the closest SPA to the offshore site) results for more than one scenario are presented for the breeding season as in this case there is uncertainty as to which scenario is the most appropriate (Scenario 1 or Scenario 2) for this period of the year.

Connectivity values estimated by the below scenarios are presented as a percentage value given to two significant figures to reduce the potential for rounding errors when the values are used in assessment calculations.

Four connectivity scenarios are considered:

Connectivity Scenario 1

This scenario is used for estimating the extent of connectivity during the colony attendance part of the breeding season. It is only used for the closest SPA, the Forth Islands SPA. Under this scenario it is assumed that all adults present in the offshore site are from the SPA population.

Connectivity Scenario 2

This scenario is also used for estimating connectivity in the colony-attendance period. The birds present in the offshore site are assumed to originate from all breeding colonies that are within 110% of the mean maximum foraging range (MMFR) distance away, and occur in proportion to the inverse of this distance and weighted by colony size. This scenario is used for estimating breeding season impacts. The reason why 110% MMFR is used rather than 100% is to incorporate some additional caution by not screening out SPAs for a species that are slightly further away than the MMFR distance. In practice this only effects whether or not two species are screened in at one SPA (Fowlsheugh SPA). Using the 110% threshold means that these species are included and therefore the assessment process is more cautious.

Connectivity Scenario 3

This scenario is also used for estimating the connectivity for seabird species in the non-breeding period(s) of the year. At this time of year, especially in the winter months, for most seabird species the birds present in the offshore site are likely to consist of a mixture of individuals from the regional breeding population and birds from breeding populations further afield, e.g., more northern breeding colonies including those in other countries such as Norway and Iceland. To estimate the degree of connectivity, information is required on the likely ratio of regional breeding to non-regional breeding birds present. Information is also required on the proportion of the regional breeding component present that is likely to be from the SPA in question. Unfortunately information on both these things is generally poor though reasonable approximations can be made for most species. Data from ringing studies and abundance surveys can broadly indicate the likely linkage at a broad scale to different breeding areas. Birds from breeding colonies within the region are assumed to occur in proportion to the size of colonies in the region.

The following hypothetical example illustrates how Scenario 3 is calculated. Ringing and abundance survey data may indicate that some of the regional breeding population of a species commonly winter outside the region and that birds from other breeding grounds are also likely to be commonly present. On this basis it might be assumed that 50% of the birds present in the non-breeding period are from regional breeding colonies. If the SPA in question has 30% of the regional breeding population of the species, then the estimated connectivity to that SPA in the non-breeding period would be 15% (i.e. 50% x 30%).

Connectivity Scenario 4

This scenario is used for migrant land bird species that might fly through the offshore site on migration. It is assumed that all individuals from a population under consideration fly through the wind farm twice per year. This is a worst case scenario. This scenario is extremely precautionary.

3.4.9 Estimating Likely Realised Effects

The methods described above in all cases consider the potential for receptor populations to be affected by the various different effects. Essentially this addresses the question of what proportion of the population under consideration is potentially exposed to the risk of the effect. Experience from operational offshore wind farms shows that in many cases the potential risks are not fully realised due to the behavioural responses exhibited by birds. For example, estimates of potential collision risk assume that flight activity by a species when the wind farm is operational will be the same as observed during the baseline surveys. However, if a species was to show complete (or partial) behavioural displacement from the wind farm area then none (or a proportion only) of the potential collision risk would be realised. Similarly, the predictions of potential displacement and barrier effects assume that a species shows complete displacement or the barrier effect is absolute. If however some or all individuals do not show a displacement response or do not perceive the wind farm as a barrier to their free movement then the potential for these effects will not be fully realised. It should be noted that displacement and collision risk act antagonistically, i.e., birds that are displaced from the wind farm are no longer at risk of collision with turbines.

The method used to estimate how much of the potential risk of an effect occurring is likely to translate into a realised effect is presented using evidence for that species from operational wind farms. Where evidence is lacking for a species, evidence from closely related species is also considered although obviously this has to be treated more cautiously. The information available from other wind farms is not always consistent or fully comparable for one reason or another, or in some cases is missing altogether. Therefore in reaching conclusions on the likely realised effects caution is exercised, in particular where there is uncertainty the more cautious interpretation is used. However, in keeping with SNH Guidance (PAN 58) it is important for conclusions to be based on biologically credible and likely scenarios and judgements; does a worse-case scenario pass such tests. It is also important for conclusions to be based on reasoned argument based on evidence and for uncertainties to be highlighted, and where appropriate to also present alternatives.

3.1 Cumulative Impact Assessment Approach

Cumulative impacts refer to assessing the predicted effects of other offshore wind farms in the region alongside those predicted for the Neart na Gaoithe Development. The cumulative impact region considered extends from Peterhead to Blyth. The offshore wind proposals in the Moray Firth are considered to be in a different region and are not therefore considered. To a very large extent, especially in the breeding season, the Moray Firth proposals would affect different populations of seabirds, although there are potential impacts on fulmars from colonies in the Moray Firth during the breeding season and other breeding species from these colonies during the non-breeding season.

There are currently no operational offshore wind farms in the cumulative impact region considered. There are two proposed offshore wind farms close to Neart na Gaoithe, in the wider Firth of Forth area. These are the Inch Cape Offshore Wind Farm (1,000 MW), which is similar in scale to the Neart na Gaoithe proposal, and the Firth of Forth Round 3 Zone wind farm which is much larger in its proposed scale (approximately 3,700 MW). In addition, an application has been submitted for the proposed Aberdeen Offshore Wind Farm in Aberdeen Bay, approximately 100 km to the north of Neart na Gaoithe. This proposal is relatively small in scale (100 MW, 11 turbines). Information from the initial submitted Environmental Statement, which was based on 18 months of data and is currently being revised, states that the main species of concern were divers and scoter species. Neither species group were found in significant numbers on baseline surveys for Neart na Gaoithe. Relatively low numbers of auks were predicted to be displaced by Aberdeen Offshore Wind Farm, including from colonies further north and therefore beyond the maximum foraging range to affect Neart na Gaoithe Development (AOWFL, 2011). Overall, based on the scale of the proposed project (11 turbines) and the initial findings in the Environmental Statement, it was concluded that impacts from Aberdeen Offshore Wind Farm would not significantly add to the cumulative effects from Neart na Gaoithe, Inch Cape and the Firth of Forth Round 3 Zone developments. Therefore the Aberdeen Wind Farm was not considered further in this Cumulative Impact Assessment.

For the Inch Cape and Firth of Forth Round 3 Zone proposals the cumulative impact assessment involved first predicting the individual impacts for each effect that might arise from these developments. This was done using information from the first year reports of baseline bird surveys for these projects (RPS, 2012, SWEL, 2011). The information in these reports is provisional and in some cases does not go into the degree of detail ideally required. The year-1 baseline survey reports also cover only a single year of baseline bird studies. For all these reasons the estimation of predicted impacts arising from these projects presented are approximate and provisional. They have been produced in good faith to give the best indication with the available data of likely cumulative impacts of offshore wind farms proposed in the region on bird populations. When the baseline studies for Inch Cape and Firth of Forth Round 3 Zone are complete and detailed impact assessment have been carried out by the ornithological consultants supporting these projects, final impact predictions for these proposals will become available.

Impacts on birds that might arise from these developments have been provisionally calculated using data provided in the year one reports on baseline bird surveys for these projects (RPS, 2012, SWEL, 2011). Both these reports provided information on the numbers of seabirds of each species recorded during the first year of baseline surveys and present preliminary results from collision rate modelling for selected species. However, information on seasonal changes in abundance and on the proportion of immature birds present is lacking for some species. In undertaking assessments of the available results for these projects various assumptions were required to overcome these information gaps.

In both cases the results of collision rate modelling do not account for the proportion of immature birds present. If this is not taken into account the predicted effect of collision mortality on the baseline adult mortality rate will be biased high. This shortcoming was overcome by assuming that the proportion of immature birds present at these sites in each season was the same as that observed during the baseline surveys at Neart na Gaoithe. In the case of the results on collision risk for the Firth of Forth Round 3 Zone development, only a single collision value is given for the whole year. To assess the impact of collision on receptor populations it was assumed that proportion of collisions occurring in the breeding season and non-breeding period respectively was the same as predicted for Neart na Gaoithe.

Where possible, displacement effects for the Inch Cape and the Firth of Forth Round 3 Zone developments were predicted using the same methods used for Neart na Gaoithe. The same seasonal divisions for a species were adopted as defined for Neart na Gaoithe. Data on the mean density of birds present each season was derived and the numbers of birds predicted to be affected was calculated by multiplying the density by the area of the proposed wind farm development buffered to 1 km. In the case of sooty shearwater, little gull, great black-backed gull and lesser black-backed gull, the Firth of Forth Round 3 Zone Year 1 report only specifies the maximum numbers present on any one survey visit. Therefore, the predicted impacts arising from this development on these species is based on the peak density recorded in a season rather than the mean density. It was assumed in these cases that the peak density occurred in the same season of the year as the peak density for the species at Neart na Gaoithe.

For the Phase 2 and Phase 3 sites of the Firth of Forth Round 3 Zone the area proposed to be covered by wind farm development is not specified in the Year 1 report. Therefore the area affected by these phases was estimated from the number of turbines proposed there and assuming the same turbine density specified for the Phase 1 site.

4 Baseline Description

4.1 Survey effort

In Year 1, surveys were conducted over 32 days between November 2009 and October 2010, with a total of 3,734.6 km surveyed. In Year 2, surveys were conducted over 28 days between December 2010 and October 2011, with a total of 3,429.6 km surveyed.

Complete coverage of both the offshore site and buffer area was achieved in all months in Year 1. In Year 2, there was no survey coverage in November due to bad weather, however full coverage was achieved in all other months (Table 4.1).

Table 4.1 Survey effort in the Neart Na Gaoithe Study Area in Years 1 and 2

Month	Offshore Site Km travelled		Buffer Area Km travelled		Proportion target coverage [1]	
	Year 1	Year 2	Year 1	Year 2	Year 1	Year 2
November	54.4	0	257.1	0	99.4%	0%
December	54.7	54.9	254.7	246.5	98.7%	96.3%
January	54.0	53.5	256.5	256.9	98.6%	98.5%
February	53.9	55.0	259.7	257.0	99.6%	100.1%
March	56.7	58.7	258.3	259.0	100%	101.7%
April	51.9	55.0	258.3	256.0	99.5%	99.8%
May	51.0	55.2	259.6	259.4	99.5%	100.4%
June	55.1	56.6	256.8	256.0	99.2%	99.9%
July	52.4	55.7	256.2	257.4	99.7%	100.1%
August	48.2	52.8	263.6	258.3	100.1%	99.7%
September	50.5	53.4	260.0	261.3	99.6%	100.8%
October	48.7	52.2	262.3	258.8	100.1%	100.9%
Total	**631.5**	**603.0**	**3,103.1**	**2,826.6**	**99.5%**	**91.6%**

1 Although full coverage was achieved in all months except in November of Year 2, there was slight variation in monthly effort, compared to the absolute length of transects, due to slight variations in the vessel trackline.

To improve data quality, Camphuysen *et al.,* (2004) recommend that seabird data collected in sea states greater than 4 are not used in subsequent analyses. Consequently, surveys were normally suspended when sea state increased above 4. Overall, the majority of all data (98.7%) were collected in Sea States 0 to 4, with only 1.3% conducted in Sea State 5 (Figure 4.1 and Figure 4.2). This data was excluded from further analyses.

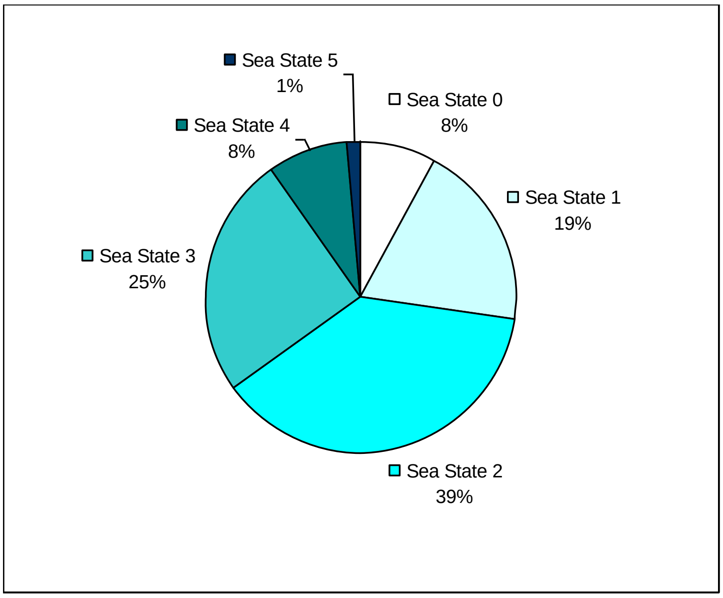


Figure 4.1 Survey effort in the Neart na Gaoithe Study Area in relation to sea state during Year 1

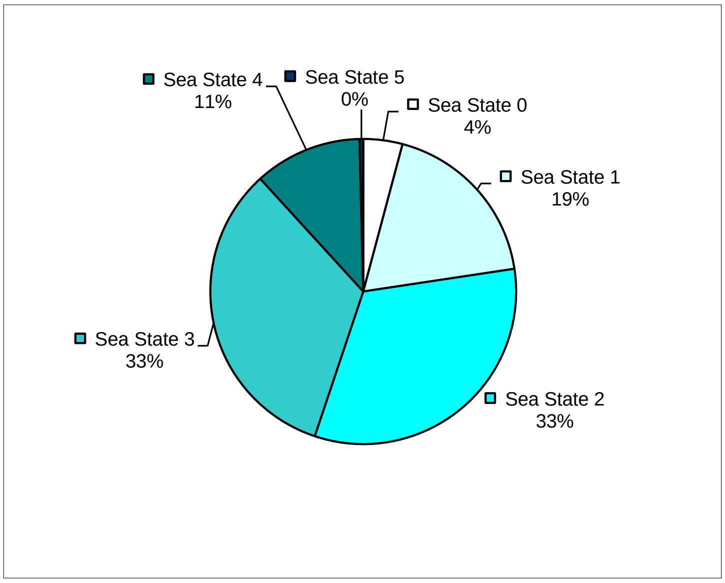


Figure 4.2 Survey effort in the Neart na Gaoithe Study Area in relation to sea state during Year 2

The monthly breakdown of survey effort in relation to sea state for Years 1 and 2 as well as survey routes for individual months is presented in Annex 1.

4.2 Raw numbers of seabirds in the Neart Na Gaoithe Study Area in Years 1 and 2

A total of 29 seabird species were identified on surveys in the Neart na Gaoithe study area in Year 1 (November 2009 to October 2010). In Year 2, 26 seabird species were recorded in the Neart na Gaoithe study area (November 2010 to October 2011) (Table 4.2).

Within the Neart na Gaoithe offshore site, 22 species were recorded in Year 1. The three most frequently recorded species in the offshore site in Year 1 were gannet, puffin and guillemot, which together accounted for 62.3% of all birds recorded. In Year 2, 16 species were recorded in the offshore site, with gannet, guillemot and puffin again the three most frequently recorded species, although the ranking was slightly different. These three species accounted for 77.1% of all birds recorded.

25 species were recorded in the buffer area in both Years 1 and 2 (Table 4.2). In Year 1 gannet, puffin and guillemot accounted for 64.7% of all birds recorded. In Year 2, gannet, guillemot and puffin accounted for 71.2% of all birds recorded.

Monthly summary tables are presented in Annex 1.

Table 4.2 Comparison of seabird numbers in the Offshore Site and Buffer Area in Years 1 and 2 (Raw numbers, all sea states)

Species	Year 1			Year 2		
	Offshore Site	Buffer Area	Total	Offshore Site	Buffer Area	Total
Red-throated diver	0	5	5	0	0	0
Fulmar	112	580	692	189	927	1,116
Sooty shearwater	84	143	227	4	175	179
Manx shearwater	16	56	72	27	259	286
Balearic shearwater	1	0	1	0	0	0
Storm petrel	0	1	1	0	0	0
Gannet	1,649	11,372	13,021	3,122	16,294	19,416
Cormorant	0	1	1	0	3	3
Shag	0	11	11	0	6	6
Eider	9	11	20	0	2	2
Common scoter	5	0	5	0	2	2
Red-necked phalarope	0	0	0	0	1	1
Grey phalarope	1	0	1	0	2	2
Pomarine skua	0	6	6	0	0	0
Arctic skua	0	6	6	0	18	18
Great skua	1	23	24	0	16	16
Little gull	32	266	298	6	214	220
Sabine's gull	1	0	1	0	1	1
Black-headed gull	0	27	27	0	11	11
Common gull	6	72	78	12	40	52
Lesser black-backed gull	10	56	66	11	184	195
Herring gull	50	1,673	1,723	58	1,375	1,433
Great black-backed gull	25	503	528	20	414	434

Species	Year 1			Year 2		
	Offshore Site	Buffer Area	Total	Offshore Site	Buffer Area	Total
Large gull species	4	158	162	1	347	348
Kittiwake	801	3,154	3,955	719	3,404	4,123
Small gull species	0	0	0	0	1	1
Common tern	3	10	13	13	37	50
Arctic tern	205	672	877	37	292	329
Common/Arctic tern	1	75	76	28	167	195
Unidentified tern species	0	34	34	0	0	0
Guillemot	1,252	6,646	7,898	1,544	10,186	11,730
Razorbill	596	3,384	3,980	350	2,781	3,131
Little auk	26	109	135	16	97	113
Puffin	1,306	9,893	11,199	1,110	5,512	6,622
Puffin/little auk	0	3	3	0	0	0
Guillemot/razorbill	368	2,955	3,323	168	1,364	1,532
Unidentified auk species	155	1,193	1,348	56	771	827
Total numbers	**6,719**	**43,098**	**49,817**	**7,491**	**44,903**	**52,394**

4.3 Flight height of birds

Information on the height of flying birds in Years 1 and 2 combined (November 2009 to October 2011) is summarised in Table 4.3. Species where fewer than 20 individuals were recorded are excluded from this, but the information is presented in the individual species accounts. Overall, 94.4% of all flying birds on baseline surveys were recorded flying below 22.5 m in height, i.e. below the wind turbine rotor swept zone. No birds were recorded flying above an estimated height of 120 m on baseline surveys.

For fulmar, sooty shearwater, Manx shearwater, guillemot, razorbill and puffin, all or nearly all birds were recorded flying at less than 22.5 m in height.

For other species, a greater proportion of birds were recorded flying above 22.5 m i.e. in the wind turbine rotor swept zone, for example 6.0% of kittiwakes (n=4,914), 6.5% of gannets (n= 28,828), 19.3% of great black-backed gulls (n=440) and 30.0% of herring gulls (n=1,253) were recorded flying above 22.5 m.

Two species of geese were recorded on baseline surveys in the Neart na Gaoithe study area, with 42.6% of pink-footed goose recorded flying above 22.5 m (n=549), while 100% of barnacle goose sightings were recorded below 22.5 m (n=900).

In Years 1 and 2, meadow pipit was the only species of land bird for which more than 20 individuals were recorded, with 1.6% recorded flying above 22.5 m (n=64). The majority of all other passerine species combined (97.2%, n=36) and wader species combined (93.9%, n=33) were recorded flying below 22.5 m in height.

Table 4.3 Flight heights of birds in the Neart na Gaoithe Study Area in Years 1 and 2 (November 2009 to October 2011) [1]

Species	Height bands in metres					Total in flight	% above 22.5m
	0 – 7.5	7.5 – 12.5	12.5 –17.5	17.5 – 22.5	Above 22.5		
Fulmar	1,464	23	0	0	2	1,489	0.1
Sooty shearwater	93	0	0	0	0	93	0
Manx shearwater	153	1	0	0	0	154	0
Gannet	22,678	1,651	499	2,132	1,864	28,824	6.5
Pink-footed goose	301	0	0	14	234	549	42.6
Barnacle goose	900	0	0	0	0	900	0
Wigeon	0	0	0	20	1	21	4.8
Eider	12	8	2	0	0	22	0
Golden plover	14	0	0	0	10	24	41.7
Unidentified waders	20	2	2	7	2	33	6.1
Arctic skua	10	3	1	5	1	20	5.0
Great skua	24	7	3	1	1	36	2.8
Little gull	163	21	3	1	83	271	30.6
Black-headed gull	4	13	6	5	10	38	26.3
Common gull	26	19	8	22	29	104	27.9
Lesser black-backed gull	115	33	10	28	30	216	13.9
Herring gull	426	193	76	200	383	1,278	30.0

Species	Height bands in metres					Total in flight	% above 22.5m
	0 – 7.5	7.5 – 12.5	12.5 –17.5	17.5 – 22.5	Above 22.5		
Great black-backed gull	191	69	23	72	85	440	19.3
Large gull species	2	33	3	13	37	88	42.0
Kittiwake	2,549	1,191	308	571	295	4,914	6.0
Common tern	29	5	1	0	0	35	0
Arctic tern	799	100	17	44	4	964	0.4
Common/Arctic tern	134	62	14	0	0	210	0
Unidentified tern species	34	0	0	0	0	34	0
Guillemot	4,981	72	1	5	2	5,061	< 0.1
Razorbill	1,698	15	6	7	0	1,726	0
Little auk	68	1	0	0	0	69	0
Puffin	5,696	69	7	5	2	5,779	< 0.1
Guillemot/razorbill	1,175	13	1	0	0	1,189	0
Unidentified auk species	151	0	0	0	0	151	0
Meadow pipit	22	25	9	2	1	59	1.7
Unidentified passerines	18	9	8	1	1	37	2.7
Total numbers	**43,950**	**3,638**	**1,008**	**3,155**	**3,077**	**54,828**	**5.6**

1 Where fewer than 20 individuals of a species were recorded in flight, the species is not shown

Table 4.4 Summary of seabird sensitivities to offshore wind farms

Species [1]	Collision	Displace-ment	Barrier effects	Habitat/ Prey interactions	Overall Risk
Red-throated diver	Low risk	High risk	Moderate risk	Moderate risk	High risk
Fulmar	Low risk	Low risk	Low risk	Moderate risk	Low risk
Sooty shearwater	Low risk	Low risk			?
Manx shearwater	Low risk	Low risk		Moderate risk	High risk
Balearic shearwater	Low risk	Low risk		Moderate risk	Moderate risk ?
Storm petrel	Low risk	Low risk		Moderate risk	Moderate risk
Gannet	Moderate risk	Low risk	Low risk	Low risk	High risk
Cormorant	Moderate risk	Low risk	Moderate risk	Moderate risk	Moderate risk
Shag	Low risk	Moderate risk	Moderate risk	Moderate risk	Moderate risk
Pink-footed goose	Moderate risk	Moderate risk	Low risk		High risk
Barnacle goose	Moderate risk	Moderate risk	Low risk		High risk
Eider	Low risk	Low risk	Moderate risk	Moderate risk	Moderate risk
Common scoter	Low risk	Moderate risk	Moderate risk	Moderate risk	Moderate risk
Pomarine skua	Moderate risk	Low risk	Low risk	Low risk	Moderate risk ?
Long-tailed skua	Moderate risk	Low risk	Low risk	Low risk	Moderate risk ?
Arctic skua	Moderate risk	Low risk	Low risk	Low risk	Moderate risk
Great skua	Moderate risk	Low risk	Low risk	Low risk	High risk
Little gull	Low risk	Low risk	Low risk	Low risk	?
Black-headed gull	Low risk	Low risk	Low risk	Low risk	Low risk
Common gull	Low risk	Low risk	Low risk	Low risk	Moderate risk
Lesser black-backed gull	Moderate risk	Low risk	Low risk	Low risk	High risk
Herring gull	Moderate risk	Low risk	Low risk	Low risk	Moderate risk
Great black-backed gull	Moderate risk	Low risk	Low risk	Low risk	Moderate risk
Kittiwake	Moderate risk	Low risk	Low risk	Low risk	Low risk
Common tern	Moderate risk	Low risk	Low risk	Moderate risk	Moderate risk
Arctic tern	Moderate risk	Low risk	Low risk	Moderate risk	Moderate risk
Guillemot	Low risk	Moderate risk	Moderate risk	Moderate risk	Moderate risk
Razorbill	Low risk	Moderate risk	Moderate risk	Moderate risk	Moderate risk
Little auk	Low risk	Moderate risk	Moderate risk	Moderate risk	Moderate risk ?
Puffin	Low risk	Moderate risk	Moderate risk	Moderate risk	Moderate risk

Key

Low risk

Moderate risk

High risk

Source: Langston 2010

4.4 Species Accounts

The following species accounts present a summary of the baseline surveys for each species, together with information on the species status and sensitivity, as well as an assessment of impacts.

Thirteen of the seabird species qualifying for assessment were considered to be higher priority on account of the high numbers present at certain times, the likely high connectivity to SPAs (nine species only) and their potential sensitivity to potential effects. The higher priority species are: fulmar, sooty shearwater, gannet, little gull, lesser black-backed gull, herring gull, great black-backed gull, kittiwake, Arctic tern, guillemot, razorbill, puffin and little auk. The possible effects of the Development on populations of these species are assessed in detail in the accounts that follow.

A further 14 seabird species seen during baseline surveys also qualified for assessment. In all cases these occurred only sporadically and in low or very low numbers, and in the case of red-throated diver, storm petrel, cormorant, shag, red-necked phalarope, pomarine skua and Arctic skua were only recorded in the 8 km buffer area around the offshore site. For this reason they are considered to be of lower priority. The assessment for these species is correspondingly less detailed.

4.4.1 Red-throated diver *Gavia stellata*

4.4.1.1 Status

Wintering red-throated divers show a preference for sheltered shallow waters and sandy bays along North Sea coasts. Numbers may fluctuate widely in response to weather and other factors affecting the prey supply of sandeels, crustaceans and small fish (Lack 1986). The wintering population around Britain has been estimated at 17,000 individuals (O'Brien *et al.,* 2008). Red-throated divers winter in small numbers off the coast of Fife, (Dean *et al.,* 2004, Barton and Pollock 2004, Söhle *et al.,* 2007), with higher numbers in the Firth of Forth, although numbers there are no longer nationally important (Calbrade *et al.,* 2010).

An estimated 1,255 pairs breed in the UK, with the majority of pairs found in the north and west of Scotland (RSPB, 2012).

4.4.1.2 Neart na Gaoithe Study Area

A total of five red-throated divers were recorded on surveys in Year 1, although no birds were recorded in the offshore site (Table 4.2). Singles were recorded in April and October, with three birds seen in June. Four birds occurred in the north of the buffer area, with one in the south-east of the buffer area (Figure 4.3).

No red-throated divers were recorded in the Neart na Gaoithe study area in Year 2.

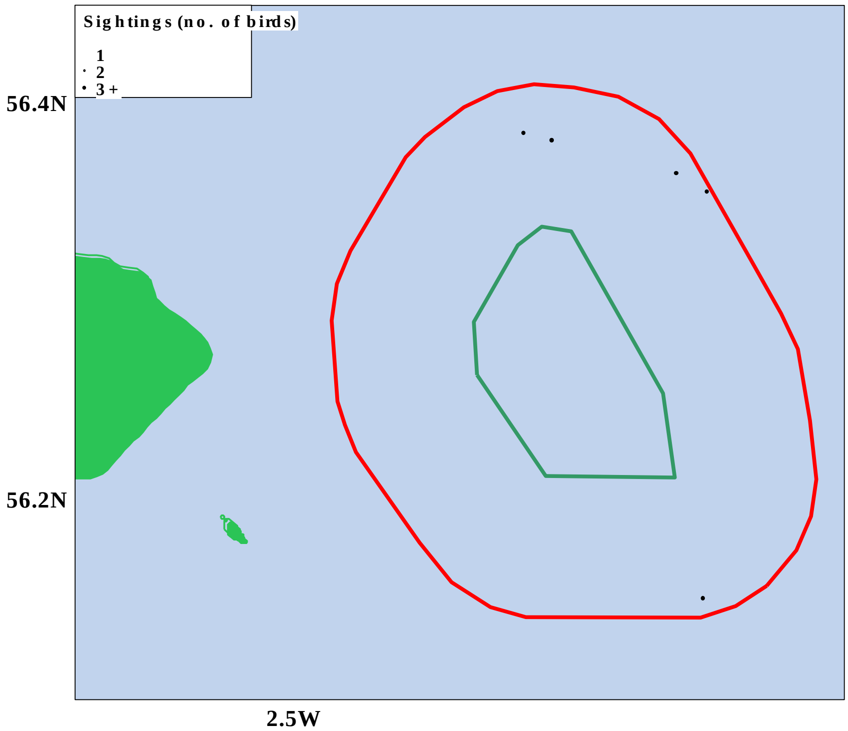


Figure 4.3 Red-throated diver sightings in Year 1

All five birds in Year 1 were recorded flying below the wind turbine rotor swept zone, with two below 7.5 m, two flying between 7.5 m and 12.5 m, and one flying between 12.5 and 17.5 m.

Species sensitivity

Red-throated diver is listed on Annex I of the EU Birds Directive (2009/147/EEC), and the species is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al.,* 2009).

A recent review assessed red-throated divers as being at high risk of displacement from wind farms (Langston 2010), and there is published evidence from individual offshore wind farm development site studies to support this (e.g. Peterson 2005, Barton *et al.,* 2008). Red-throated diver was also assessed as being at moderate risk of barrier effects and habitat loss, and at low risk of collision with turbines. Overall, the species was assessed as being at high risk from offshore wind developments (Langston 2010) (Table 4.4).

Red-throated diver is listed as a qualifying interest species in the non-breeding season for one SPA on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development; Firth of Forth SPA. This SPA held 88 birds or 1.8% of the UK non-breeding population, and 0.1% of the biogeographic population at the time of designation (JNCC, 2012). A similar figure (80 birds) was recorded as the most recent available five-year mean for wintering red-throated divers in the Forth Estuary, between 2004 and 2009 (Calbrade *et al.*, 2010). This figure is below the 1% threshold of national importance (170 birds) (Holt *et al.,* 2011).

Assessment

No red-throated divers were recorded in the offshore site on baseline surveys. A total of five red-throated divers were recorded in the Buffer area in Year 1, while the species was not recorded in the study area in Year 2.

The small number of birds wintering off the coast of Fife and nationally important numbers wintering in the Forth Estuary (Barton & Pollock 2004; Dean *et al.,* 2004; Söhle *et al.,* 2007; Calbrade *et al.,* 2010) were not recorded within the Neart na Gaoithe study area. The results of baseline surveys suggest that red-throated divers occur occasionally and in very small numbers in the Neart na Gaoithe offshore site and are therefore unlikely to be at risk from the Development.

4.4.2 Northern Fulmar *Fulmarus glacialis*

4.4.2.1 Status

Fulmar numbers and distribution around the UK have increased considerably since the mid-19th century (Pennington *et al.,* 2004). The species is now one of the commonest seabirds in Britain, with an estimated breeding population of 499,081 pairs (Mitchell *et al.,* 2004). The largest breeding colonies are located off the north and west coasts of Scotland. Birds are often present at breeding cliffs outside the breeding season. Fulmars forage at sea, with offal and fish discards from trawlers now a major part of their diet (Forrester *et al.,* 2007).

4.4.2.2 Neart na Gaoithe Study Area

In Year 1, 692 fulmars were recorded on surveys in the Neart na Gaoithe study area, with the majority of birds (83.8%) recorded in the buffer area (Table 4.2). Numbers of fulmars recorded on surveys in Year 2 were higher, with a total of 1,116 birds recorded, the majority of which (83.1%) were in the buffer area.

During the breeding season (March to September), the peak estimated number of fulmars occurred in September of Year 1, with an estimated 57 birds in the offshore site, and 295 birds in the buffer area (Table 4.5) (Figure 4.4). In Year 2, peak estimated numbers of fulmars were also recorded in September, with an estimated 69 birds in the offshore site, and 890 birds in the buffer area.

In the non-breeding season (October to February), the peak estimated number of fulmars occurred in December of Year 1, with an estimated 70 birds in the offshore site, and 185 birds in the buffer area (Table 4.5) (Figure 4.4). In Year 2, peak estimated numbers of fulmars in the offshore site were recorded in December and January (77 birds), with an estimated 439 birds in the buffer area in January.

Table 4.5 Estimated numbers of fulmars in the offshore site and 8 km buffer in Years 1 and 2

Month	Estimated nos on water Offshore Site	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying Offshore Site	Estimated total Offshore Site	Estimated total 8 km buffer [1]	Estimated total
Yr1 Nov	0	0	0	14	14	110	124
Yr1 Dec	43	7	249	27	70	185	255
Yr1 Jan	0	0	0	48	48	120	168
Yr1 Feb	28	3	280	7	35	110	145
Yr1 Mar	32	13	81	13	45	218	263
Yr1 Apr	0	0	0	14	14	135	149
Yr1 May	0	0	0	0	0	52	52
Yr1 Jun	0	0	0	0	0	18	18
Yr1 Jul	0	0	0	0	0	27	27
Yr1 Aug	0	0	0	14	14	59	73
Yr1 Sep	50	21	123	7	57	295	352
Yr1 Oct	0	0	0	7	7	0	7
Yr2 Nov	-	-	-	-	-	-	-
Yr2 Dec	0	0	0	77	77	416	493
Yr2 Jan	9	4	22	68	77	439	516
Yr2 Feb	11	2	62	20	31	62	93
Yr2 Mar	0	0	0	13	13	147	160
Yr2 Apr	43	17	105	7	50	114	164
Yr2 May	0	0	0	13	13	111	124
Yr2 Jun	0	0	0	34	34	68	102
Yr2 Jul	21	3	165	13	34	212	246
Yr2 Aug	0	0	0	14	14	761	775
Yr2 Sep	42	23	75	27	69	890	959
Yr2 Oct	0	0	0	7	7	73	80

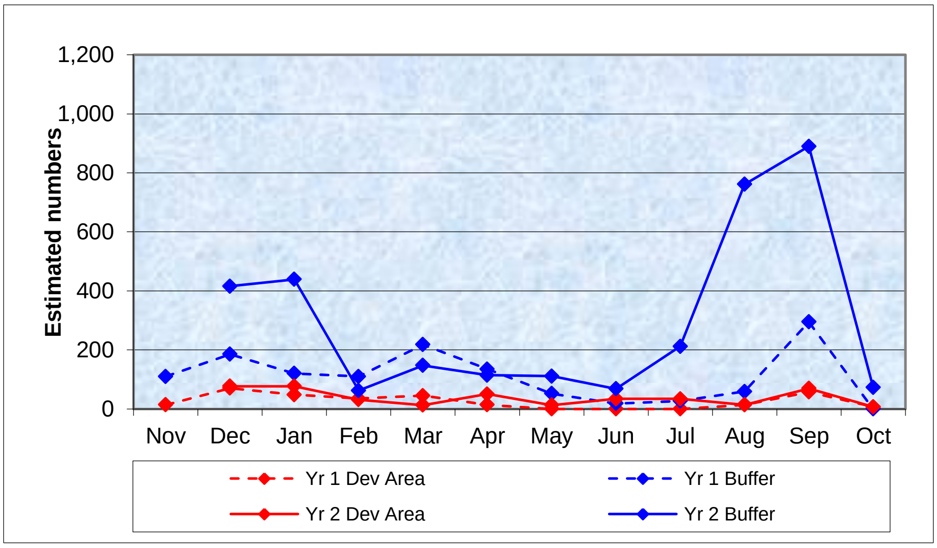


Figure 4.4 Monthly estimated numbers of fulmars in the Neart na Gaoithe Development & buffer areas in Years 1 and 2

Mean monthly fulmar density was generally low in the offshore site and the buffer area throughout Years 1 and 2 (Figure 4.5). ESAS density data from the surrounding ICES rectangles was similar, while ESAS density data for fulmar across Regional Sea 1 was higher, particularly in late autumn and early winter.

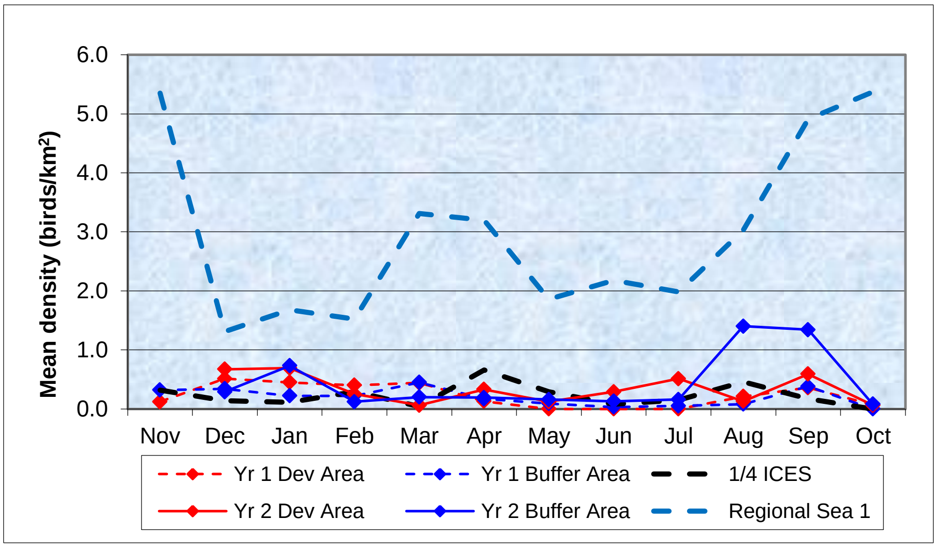


Figure 4.5 Comparison of fulmar monthly mean densities in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

Between November and March of Year 1, fulmars were widespread at low to moderate densities across most of the Neart na Gaoithe study area, although fewer birds were present in the west (Figure 4.6).

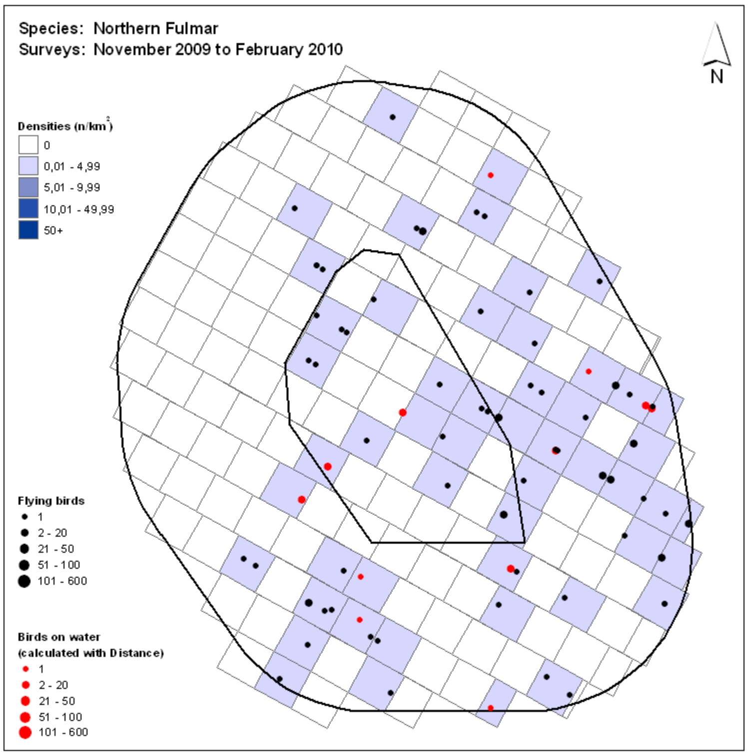


Figure 4.6 Fulmar density between November and February, Year 1

Between October and February of Year 2, fulmars were widespread at low to moderate densities across the Neart na Gaoithe study area (Figure 4.7). Highest densities were recorded to the south of the offshore site at this time.

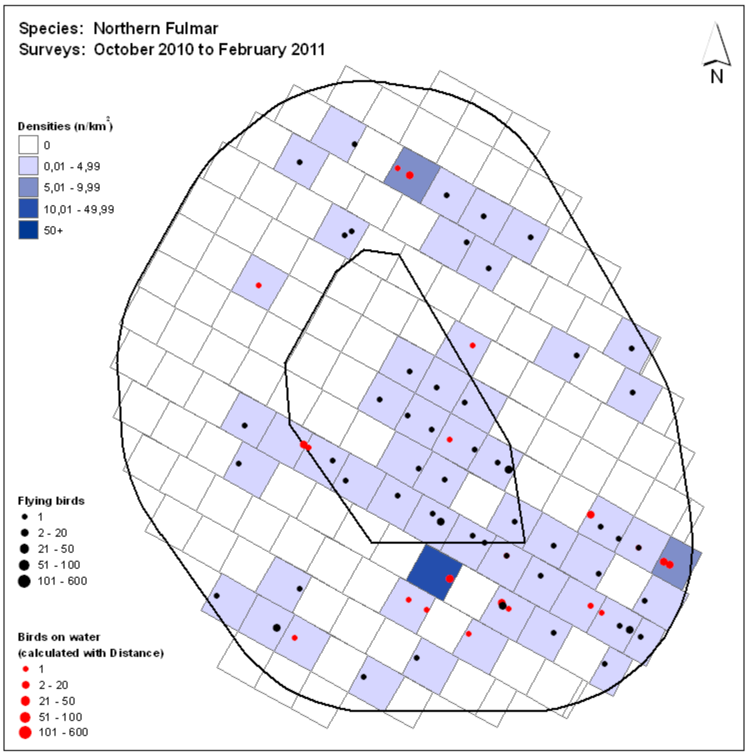


Figure 4.7 Fulmar density between October and February, Year 2

Fulmar density during the Year 1 breeding season (March to September) was low to moderate across the Neart na Gaoithe study area over the period. Few birds were recorded in the offshore site at this time (Figure 4.8).

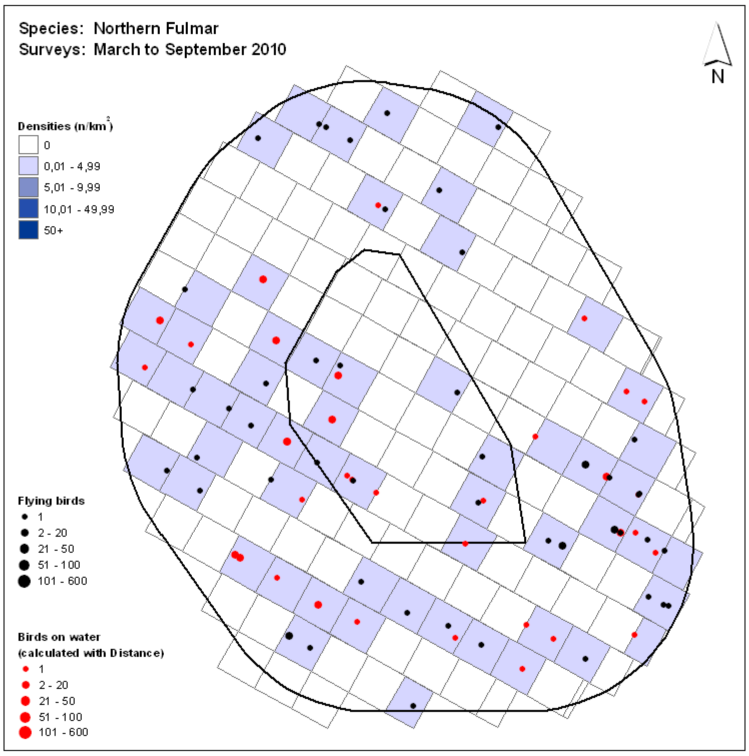


Figure 4.8 Fulmar density between March and September, Year 1

In Year 2, Fulmar density in the Neart na Gaoithe study area in the summer months was similar to Year 1, with low to moderate densities recorded, although birds were more widespread in the offshore site over the period, compared to Year 1 (Figure 4.9).

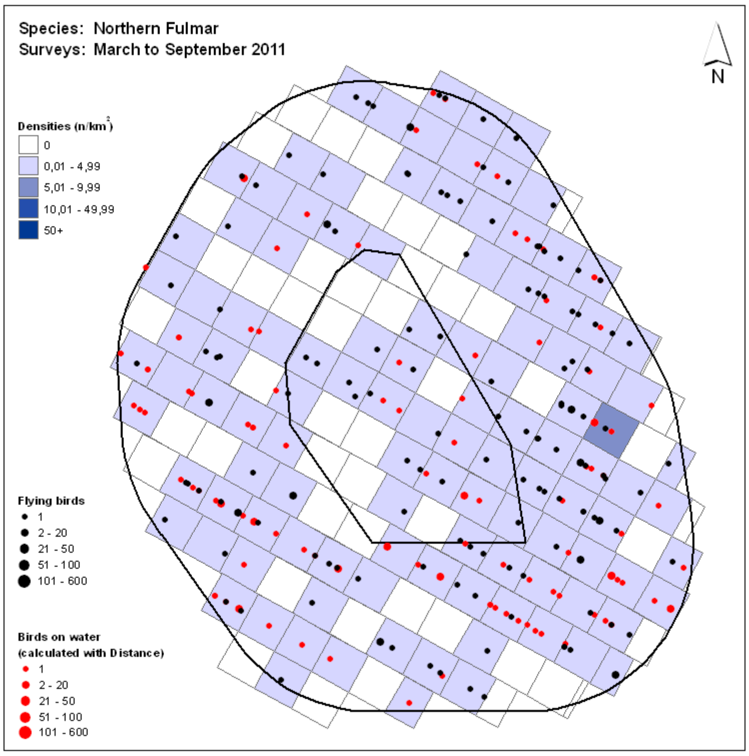
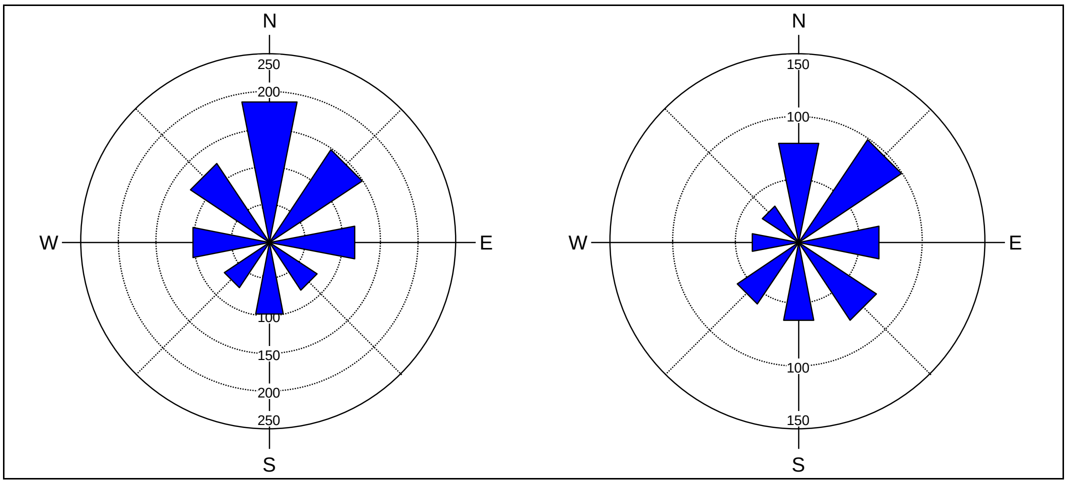


Figure 4.9 Fulmar density between March and September, Year 2

A total of 1,489 fulmars were recorded in flight in Years 1 and 2, with the majority of all birds (98.3 %) recorded flying below 7.5 m in height (Table 4.3). Just two birds (0.1 %) were recorded flying above 22.5 m, i.e. within the rotor swept zone of the turbines, at an estimated height of 30 m.

Flight direction was recorded for 927 fulmars in the breeding season (March to September), with direction recorded for 514 fulmars in the non-breeding season (October to February) (Figure 4.10). An additional 15 birds were recorded as circling (not shown).



March to September (n=927 birds) October to February (n=514 birds)
Numbers shown on figures are numbers of birds recorded

Figure 4.10 Flight direction of fulmars in the Neart na Gaoithe Study Area in Years 1 and 2

In the breeding season, there was a slight pattern of fulmars flying north (20.5 %) and north-east (16.0 %), with fewer birds recorded flying in other directions. In the non-breeding season, 19.3 % of birds were recorded flying north-east, with 15.8 % flying north.

Three types of foraging behaviour were recorded for fulmars in Years 1 and 2, with surface pecking the most frequently recorded behaviour, although the sample size was very small (Table 4.6).

Table 4.6 Fulmar foraging behaviour in the Neart na Gaoithe Study Area in Years 1 and 2

Behaviour	Number of birds
Dipping	2
Scavenging	4
Surface pecking	8

Species sensitivity

Fulmar is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al.*,2009).

A recent assessment rated fulmar as being at moderate risk of habitat loss or changes in prey distribution resulting from offshore wind farms. Risk of collision, displacement and barrier effects were rated as low. Overall, fulmar was assessed as being at low risk from offshore wind developments (Langston 2010) (Table 4.4).

Fulmar is listed as a qualifying interest species in the breeding season for 17 SPAs on the UK east coast between Hermaness (Shetland) and Spurn (Yorkshire) that could potentially be affected by the Neart na Gaoithe development (Table 4.7). These SPAs held 38.0% of the UK breeding population, and greater than 2.4% of the biogeographic population at the time of designation (JNCC, 2012). The distance from the offshore site to 13 of these SPAs is within the mean maximum foraging range of 400 km, while the distance from the development to the remaining four SPAs is within the maximum known foraging range of 580 km (Thaxter *et al.,* 2012). The five closest SPAs to the offshore site are shown in Figure 4.11.

Table 4.7 SPAs for breeding fulmar between Hermaness and Spurn

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count [2]	Year
Buchan Ness to Collieston Coast	**113**	**1,765**	**<0.1**	**0.3**	**1,389**	**2007**
Calf of Eday	**333**	**1,955**	**<0.1**	**0.4**	**940**	**2002**
Copinsay	**297**	**1,615**	**<0.1**	**0.3**	**1,366**	**2008**
East Caithness Cliffs	**260**	**15,000**	**0.2**	**2.8**	**16,164**	**1999**
Fair Isle	**356**	**43,320**	**0.6**	**8.0**	**27,896**	**2006**
Fetlar	*477*	*9,800*	*0.1*	*1.8*	*9,203*	*1999-2001*
Forth Islands	**16**	**1,600**	**<0.1**	**0.3**	**514**	**2005**
Foula	*424*	*46,800*	*0.6*	*8.7*	*21,106*	*2000*
Fowlsheugh	**62**	**1,170**	**<0.1**	**0.2**	**193**	**2009**
Hermaness, Saxa Vord &Valla Field	*510*	*14,890*	*0.2*	*2.8*	*11,144*	*1999*
Hoy	**301**	**35,000**	**0.5**	**6.5**	**35,858**	**1999-2001**
North Caithness Cliffs	**275**	**16,310**	**0.2**	**3.0**	**4,551**	**1999**
Noss	*428*	*5,870*	*<0.1*	*1.1*	*5,169*	*2002*
Rousay	**337**	**1,240**	**<0.1**	**0.2**	**1,622**	**1999-2001**
Sumburgh Head	**396**	**2,542**	**<0.1**	**0.5**	**1,487**	**1999**
Troup, Pennan & Lion Heads	**171**	**4,400**	**<0.1**	**0.8**	**2,900**	**2001**
West Westray	**342**	**1,400**	**<0.1**	**0.3**	**3,185**	**1999-2001**
Total	**-**	**204,677**	**>2.4 %**	**38.0 %**	**144,687**	**-**

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. Sites in italics lie within the maximum known foraging range of 580 km. Sites in bold lie within the mean maximum foraging range of 400 km (Thaxter *et al.*, 2012).

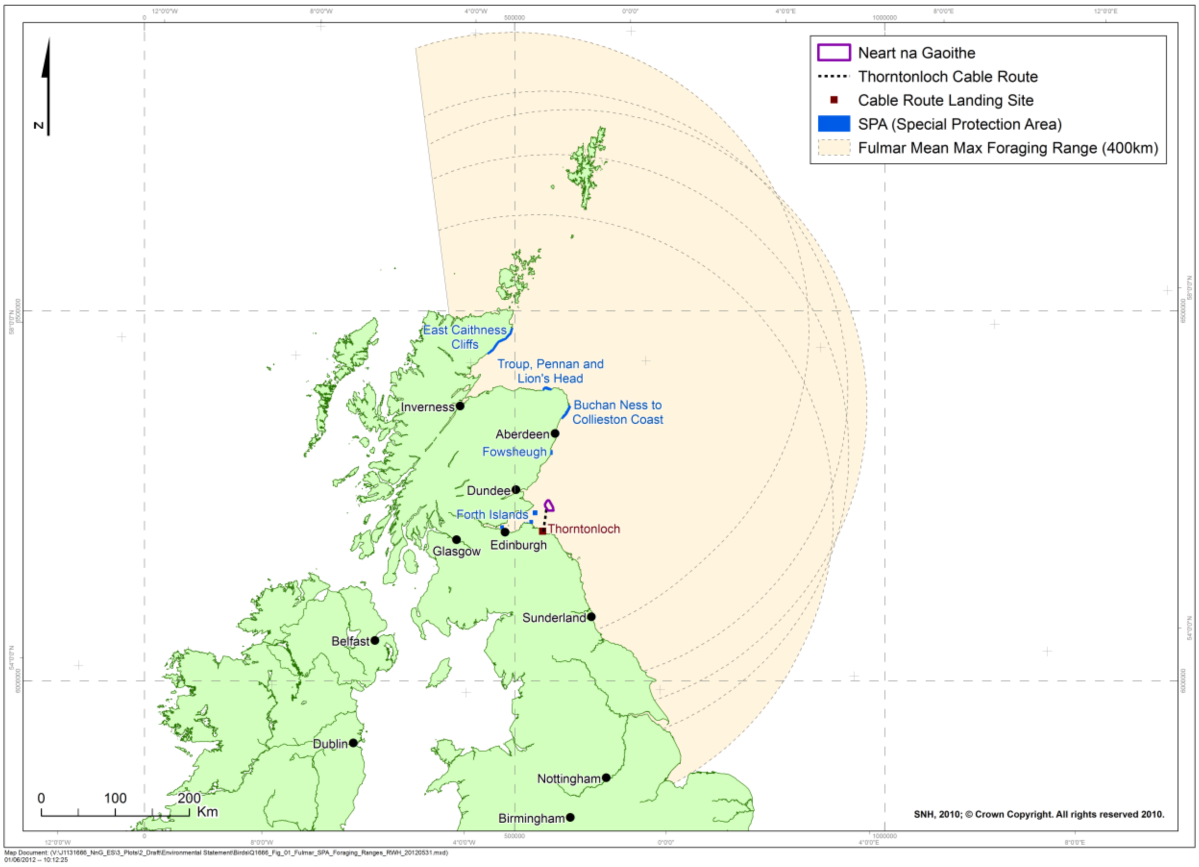


Figure 4.11 Fulmar foraging range from the five nearest SPAs in relation to the Development

4.4.2.3 Assessment

Definition of seasons used for fulmar

The annual cycle for fulmar is divided into two parts to reflect the biology of the species and the broad pattern of use of the offshore site.

The breeding season, the period when breeding adults are attending colonies, is defined as March to September. At this time the vast majority of birds present in the offshore site are likely to be from breeding colonies that are no further from the offshore site than the mean maximum foraging range.

The non-breeding period is defined as October to February and broadly corresponds to the period when fulmars are in their over-wintering area. In this period it is likely that a high proportion of individuals present in the offshore site are from breeding sites outwith the region, including birds from other countries (Wernham *et al.,* 2002).

Foraging range and seasonal movements of fulmar

Breeding fulmars travel large distances from breeding sites to forage, and the mean maximum foraging distance is 400 km (Thaxter *et al.,* 2012).

Baseline conditions for fulmar

In the breeding season (March to September) the mean estimated number of fulmars present (birds on the sea and in flight) in the offshore site was 27 birds, and the mean estimated number in the offshore site buffered to 1 km was 49 birds. On average 65% of birds present in the breeding season were on the sea and the remainder were in flight, this suggests that a high proportion of birds were using the area for foraging.

For the purposes of assessment it is assumed that all birds in the breeding season were breeding birds. This will tend to lead to cautious conclusions as it is likely than some immature birds were also present, either from the regional population or from populations with breeding grounds further north. For the purposes of assessment it was assumed that the average colony attendance rate by adults in the breeding season is 50%, and this figure is used in calculations of the estimated breeding season importance of the offshore site for foraging.

In the non-breeding period (October to February) the mean number of fulmars present in the offshore site was 37, and the mean number in the offshore site buffered to 1 km was 48. On average 47% of birds present in the non-breeding period were on the sea and the remainder were in flight, suggesting that a high proportion of birds were using the area for foraging.

Populations

The regional breeding population size is assumed to be 333,188 birds (i.e. 166,549 pairs). This figure is based on the Seabird 2000 counts for Orkney to Yorkshire adjusted to take account of recent changes where these are available (SMP 2012).

The regional population size in the non-breeding period is assumed to be approximately 400,000 birds. This figure is based on a cautious interpretation of the population estimates for Areas 7, 8, and 9 for November to February given in Skov *et al*. (1995).

Nature conservation importance

For EIA assessment purposes, the nature conservation importance (NCI) of fulmars using the offshore site is rated at moderate throughout the year. This species merits moderate NCI classification on the basis that a high proportion of birds present in the offshore site are likely to be from SPA colonies, in particular Forth Islands SPA and St Abb's Head to Fast Castle SPA. This species is not subject to special legislative protection, and is not listed on any conservation listings. The mean numbers present are well below 1% of the (at-sea) regional population at all times of year.

Offshore wind farm studies of fulmar

Fulmars were uncommon at almost all the operational wind farms that have been studied and therefore there is a paucity of information on how this species responds to offshore wind farms. At Egmond aan Zee, the Netherlands, the results for the single survey with a sufficiently large sample for analysis showed no clear influence of the wind farm on the distribution of fulmar. At Arklow Bank, Ireland, the number of fulmars significantly declined on the survey legs closest to the turbines, however there was no evidence that these declines were associated with proximity to the turbines (Barton *et al*., 2009). There is limited evidence of the extent to which wind farms present a barrier to fulmars. At Horns Rev, Denmark, a single fulmar approaching the wind farm was observed to change direction to apparently avoid flying through the turbines (Diersche and Garthe, 2006). However at Blyth Harbour, UK, anecdotal reports of fulmars passing through the wind farm, corroborated by one recorded collision at this site, suggest that here any barrier effect to fulmars was at most only partial (Zucco *et al.,* 2006).

EIA Construction Phase assessment for fulmar

The construction phase is of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

Fulmars are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during construction operations on the regional populations of fulmars in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

EIA Operational Phase assessment for fulmar

Potential for fulmar to be affected by displacement

Table 4.8 The mean estimated number of fulmar present during the breeding and non-breeding periods, and this value as a percentage of the (at-sea) receptor population potentially at risk of displacement

Receptor population (Source)	Population size (adults)	No. assumed at sea	Dev.Area		Dev.Area + 1km	
			Mean no. at risk	%	Mean no. at risk	%
National, breeding period (Seabird 2000)	993,333	496,667	27	<0.1	49	<0.1
North Sea, non-breeding period (Skov *et al*., 1995)	1,872,000	1,872,000	37	<0.1	48	<0.1
Regional, breeding period (Seabird 2000)	333,188	166,594	27	<0.1	49	<0.1
Regional, non-breeding (Skov *et al*., 1995)	400,000	400,000	37	<0.1	48	<0.1

For estimating the potential for displacement to affect foraging birds, the assessment was made on all birds (on the sea and in flight) recorded during baseline surveys. The value of the offshore site buffered to 1 km as a foraging site for breeding fulmars was estimated for each of the receptor populations in terms of the average proportion of at-sea adults (i.e. not attending a colony) that were present in the offshore site during the breeding season (Table 4.8). This is not a precise measure of foraging value but it is considered to be a reasonable measure in the absence of more detailed information on behaviour, and is likely to result in cautious assessment conclusions as it will tend to overestimate the importance of the area.

On this basis, it is estimated that the offshore site buffered to 1 km provides <0.1% of the foraging resources of the regional breeding population (Table 4.8). Put another way, at any one time in the breeding season, it is estimated that <0.1% of the breeding birds in the regional population that are at sea (i.e. away from their colony) occur in the offshore site and therefore are potentially at risk of being displaced by the wind farm.

The value of the offshore site (buffered to 1 km) for foraging fulmars in the autumn/winter period was estimated using the regional populations. The regional autumn/winter population for fulmar is poorly defined and quantified, however was crudely estimated at 400,000 birds based on information in Skov *et al*. (1995) (Table 4.8).

Likely impacts of displacement on fulmar populations

Given the paucity of information on the likely displacement response of fulmar to the wind farm, for the purposes of assessment it is assumed that fulmars will show complete displacement from the wind farm area buffered to 1 km. This assumption is close to a worst-case scenario and leads to cautious assessment conclusions because it is likely that not all individuals will be displaced. Furthermore some individuals present may not be foraging in the area but merely passing through the area.

Nevertheless, it is estimated that on average displacement could result in the effective loss of up to 0.06% of the foraging habitat of the regional breeding fulmar population. It is also estimated that displacement could result in the effective loss of only 0.01% of the foraging habitat of the regional non-breeding period fulmar population.

The likely impact of displacement from foraging areas in the breeding period on the regional breeding fulmar population is an effect of negligible magnitude, temporally long term and reversible. It is concluded that this effect is ***not significant*** under the EIA Regulations.

The likely impact of displacement from foraging areas in the non-breeding period on the regional non-breeding fulmar population is rated to be an effect of negligible magnitude, temporally long term and reversible. It is concluded that this effect is ***not significant*** under the EIA Regulations.

Barrier effects on regional fulmar population

There is the potential for the wind farm to act as a barrier to the foraging flights of breeding fulmar and cause them to detour around the wind farm. The paucity of information on the response of fulmars to wind farms means there is uncertainty over the extent to which this potential would be realised. The relatively low wing loading and efficient dynamic soaring flight of fulmars mean that this species is likely to have low sensitivity on energetic grounds to small increases in foraging flight distances. Langston (2010) categorised fulmar as having low sensitivity to barrier effects.

The proposed development buffered to 1 km would potentially form a barrier 17.9 km wide and located 16 km to the north-east of the Isle of May. This barrier would potentially block approximately 33% of the possible flight directions available to fulmars flying out to distances in excess of 16 km from the Isle of May (Table 3.6 and Table 3.7). Similarly, the proposed development buffered to 1 km would potentially form a barrier 17.8 km wide 27 km north-east of the Bass Rock and block approximately 27% of the possible flight directions available to fulmars flying out to distances in excess of 27 km (Table 3.6). For fulmars breeding at St. Abb's Head, the proposed development buffered to 1 km would potentially form a barrier 11.6 km wide 33 km to the north and potentially block approximately 9% of the possible flight directions available to fulmars flying out to distances in excess of 33 km (Table 3.6). In one respect these figures on the percentages of foraging trips likely to be affected are likely to be cautious (biased high) because the destinations of some of these flights will be closer than the barrier. However, it is also possible that fulmars preferentially select the directions affected by a barrier above the other directions available, in which case the assumed proportion of flights affected could be biased low. No attempt is made to correct for these potential biases; it is likely that they will cancel each other to some extent.

The potential effect the barrier would have on flight distances and times depends on how far the destination areas lie behind the barrier. The results from tagging studies on fulmars show that they forage over vast areas and commonly travel distances in excess of 300 km, and sometimes over twice this distance (Thaxter *et al.,* 2012). It is therefore reasonable to assume that likely destinations of fulmar foraging trips affected by the wind farm acting as a barrier would be at a wide range of distances beyond the offshore site, and commonly many tens of kilometres beyond.

The mean destination distance of fulmar foraging flights is 48 km (Thaxter *et al.,* 2012). Acknowledging there is uncertainty in how far on average the destination distance of affected flights are from the colony, for the purpose of assessment a cautious value of 50 km is assumed. This would mean that the flight routes of birds affected by a barrier effect would be increased by approximately 2.2% for the Isle of May (Table 3.7).

Assuming the destinations of affected flights are on average 50 km from the colonies, the mean increase in the length of barrier-affected flights is estimated at 7.1% (Table 3.7) for birds nesting on the Isle of May, 4.8% for birds breeding on Bass Rock and 4.4% for birds from St Abb's Head. The size of detours that fulmar experiencing a barrier effect would be required to make is small and only a small proportion (the affected colonies represent approximately 1.5% of the regional total) of the breeding fulmars in the region would potentially be affected. The effect on fulmars of the wind farm forming a barrier is categorised as negligible magnitude temporally long term and reversible. Bearing in mind that fulmars are considered to have low sensitivity to barrier effects (Langston 2010), it is concluded that the impact of the wind farm acting as a barrier during foraging trips of the regional breeding fulmar population is ***not significant*** under the terms of the EIA Regulations.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

Fulmars are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010) (Table 4.4). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional populations of fulmars in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

Collision mortality

CRM was not undertaken for fulmar because 99.9% of all birds seen in flight during the baseline surveys were below the proposed minimum rotor swept height of turbines. Therefore, it is not plausible that this species will experience significant mortality from collision with turbine rotors.

The potential effect of the collision mortality of fulmars on the baseline mortality rate is rated as negligible in magnitude (<1%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of collision mortality on fulmars is ***not significant*** under the terms of the EIA Regulations.

EIA Decommissioning Phase assessment for fulmar

The decommissioning phase is of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

Fulmars are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010) (Table 4.4). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during decommissioning operations on the regional population of fulmars in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

EIA summary of effects combined for fulmar

The adverse impacts of the effects assessed will act in a broadly additive manner. In combination it is judged that the magnitude of the three effects on the regional population of fulmars in the breeding period is negligible. Furthermore the population has low sensitivity to all effects. It is concluded that the overall impact on the regional population of fulmars in the breeding period is ***not significant*** under the EIA regulations (Table 4.9).

Table 4.9 Summary of effects on the regional population of fulmars in the breeding period

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Low	***Not significant***
Barrier Effect	Negligible	Long term	Low	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality	Negligible	Long term	Low	***Not significant***
All effects combined	Negligible	Long term	Low	***Not significant***

The adverse impacts of the effects assessed will act in a broadly additive manner. In combination it is judged that the magnitude of the three effects on the regional population of fulmars in the non-breeding period is negligible. Furthermore the population has low sensitivity to all effects. It is concluded that the overall impact on the regional population of fulmars in the non-breeding period is ***not significant*** under the EIA regulations (Table 4.10).

Table 4.10 Summary of effects on the regional population of fulmars in the non-breeding period

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Low	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality	Negligible	Long term	Low	***Not significant***
All effects combined	Negligible	Long term	Low	***Not significant***

4.4.2.4 Cumulative Impact Assessment for fulmar

Combining the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts on the regional breeding population of fulmar will be as shown in Table 4.11. The individual impacts for displacement and collision are combined by addition to give the cumulative impact. The flight directions from the colonies that are potentially affected by each wind farm acting as a barrier substantially overlap and therefore the cumulative barrier impact from the three developments is not the sum of the individual predicted impacts. The cumulative barrier impact is derived from the overall spread of flight directions at each colony potentially affected by the three developments forming barriers.

The predicted potential effective loss of <0.6% of the foraging habitat of the regional breeding population is rated as negligible in magnitude (<1%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional population of fulmars in the breeding season is ***not significant*** under the terms of the EIA Regulations.

The three proposed wind farms could form a substantial barrier to the flights of fulmars attending Fowlsheugh, Forth Islands and St Abb's Head colonies. Together these colonies represent approximately only 2% of the regional breeding population. The remaining colonies in the region are too far away to experience anything other than negligible barrier effects during the breeding season from these proposed wind farms.

Table 4.11 Summary of CIA for the three proposed offshore wind farms in south-east Scotland on the regional population of fulmar in the breeding season

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF +1km	Predicted impact from Inch Cape pOWF +1km	Predicted impact from Firth of Forth R3 Zone pOWF + 1km	Cumulative Impact
Displacement: effective loss of foraging habitat during breeding period (%)	100%	<0.1%	<0.1%	0.4%	≤0.6% Negligible
Barrier Effect:	100%	Foraging trips of a minority of birds breeding at Forth Islands and St Abb's Head potentially increase by 4-7% depending on colony.	Foraging trips of a minority of birds breeding at Fowlsheugh and Forth Islands potentially increase by 5-6% depending on colony.	Foraging trips of a minority of birds breeding at Fowlsheugh, Forth Islands and St Abb's Head potentially increase by 13-19% depending on colony.	Foraging trips of <2% of the regional population potentially affect. Affected flights increase by 13-19% depending on colony. Negligible
Collision:-% increase in assumed annual adult mortality rate for most adverse designs (99.5% AR)	0%	No change. No flight activity recorded at rotor height	No CRM data available. Presumed negligible.	No CRM data available. Presumed negligible.	Negligible
Collision: % increase in assumed annual adult mortality rate for least adverse designs (99.5% AR)	0%	No change. No flight activity recorded at rotor height	No CRM data available. Presumed negligible.	No CRM data available. Presumed negligible.	Negligible

The three wind farms could potentially affect a high proportion of the potential flight directions from these three colonies, ranging from about one third of possible flight directions for Fowlsheugh and St Abb's Head, to approximately two-thirds of directions for the Forth Islands colonies. Assuming that the destination of affected flights is 100 km from the colony (this is approximately the closest distance possible beyond the barrier and is greater than the mean foraging distance of 48 km), affected flights are estimated to be on average between 13% and 19% longer (depending on the colony concerned) than the corresponding direct flight to the same destination. The cumulative effect is rated as low in magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). Fulmars are capable of undertaking very long foraging flights (Thaxter et *al*., 2012) and it is therefore likely that they have a moderate capacity to absorb the small increases in flight distances that these barriers could cause. This species is also rated as low for sensitivity to barrier effects by Maclean *et al.,* (2009) and Langston (2010). It is concluded that any adverse effects caused by the cumulative impact of wind farms acting on the regional breeding fulmar population as a barrier is ***not significant*** under the terms of the Electricity Act.

The cumulative assessment of barrier effects above assumes that all areas of the wind farm sites will be developed. In the case of the Firth of Forth Round 3 Zone there will be large areas without wind farm development and the eventual design may therefore effectively present fulmars several smaller barriers with large gaps through which birds will pass, rather than the single large barrier assumed above. Were this to be the case the magnitude of the cumulative barrier effect could be substantially less than suggested above.

Almost no fulmars are predicted to be killed by the three proposed offshore wind farms because almost all flight activity observed in baseline surveys was well below rotor height. The potential impact of collision mortality on the baseline mortality rate of the regional breeding fulmar population is rated as negligible in magnitude (<1%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional population of fulmars in the breeding season is ***not significant*** under the terms of the EIA Regulations.

CIA has not been undertaken for the regional fulmar population in the non-breeding period because the predicted effects of Neart na Gaoithe wind farm at this time of year are very close to no effect. Therefore it is not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for this receptor population.

4.4.2.5 Mitigation measures for fulmar

The assessment does not identify any significant adverse effects on the regional population of fulmars in the breeding or non-breeding periods. Therefore no mitigation measures are required for this species.

4.4.3 Sooty shearwater *Puffinus griseus*

4.4.3.1 Status

Although sooty shearwaters breed in the southern hemisphere on islands off New Zealand, Australia, Chile and the Falkland Islands, the species is regularly recorded on migration off the east coast of Scotland from July to October, but rarely outwith this period (Forrester *et al.,* 2007). Sooty Shearwater is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

4.4.3.2 Neart na Gaoithe Study Area

In Year 1, 227 sooty shearwaters were recorded on surveys in the Neart na Gaoithe study area, with the majority of birds (63.0%) recorded in the buffer area in October (Table 4.2). In Year 2, 179 sooty shearwaters were recorded on surveys, however only four birds were in the offshore site, with 97.8% of birds recorded in the 8 km buffer area. Again, highest numbers were seen in October.

In Year 1, peak numbers of sooty shearwaters occurred in October, with an estimated 162 birds in the offshore site, and 618 birds in the buffer area (Table 4.12) (Figure 4.12). In Year 2, numbers of sooty shearwaters in the offshore site were low, with a peak of seven birds in September and October. In the buffer area, estimated numbers peaked in October, with 892 birds in the buffer area.

Table 4.12 Estimated numbers of sooty shearwaters in the offshore site and 8 km buffer in Years 1 and 2

Month	Estimated nos on water Offshore Site	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying Offshore Site	Estimated total Offshore Site	Estimated total 8 km buffer [1]	Estimated total
Yr1 Nov	0	0	0	7	7	0	7
Yr1 Dec	0	0	0	0	0	0	0
Yr1 Jan	0	0	0	0	0	0	0
Yr1 Feb	0	0	0	0	0	0	0
Yr1 Mar	0	0	0	0	0	0	0
Yr1 Apr	0	0	0	0	0	0	0
Yr1 May	0	0	0	0	0	0	0
Yr1 Jun	0	0	0	0	0	0	0
Yr1 Jul	0	0	0	0	0	0	0
Yr1 Aug	0	0	0	0	0	13	13
Yr1 Sep	0	0	0	7	7	160	167
Yr1 Oct	162	95	278	0	162	618	780
Yr2 Nov							
Yr2 Dec	0	0	0	0	0	0	0
Yr2 Jan	0	0	0	0	0	0	0
Yr2 Feb	0	0	0	0	0	0	0
Yr2 Mar	0	0	0	0	0	0	0
Yr2 Apr	0	0	0	0	0	0	0
Yr2 May	0	0	0	0	0	0	0
Yr2 Jun	0	0	0	0	0	0	0
Yr2 Jul	0	0	0	0	0	0	0
Yr2 Aug	0	0	0	0	0	14	14
Yr2 Sep	0	0	0	7	7	227	234
Yr2 Oct	0	0	0	7	7	892	899

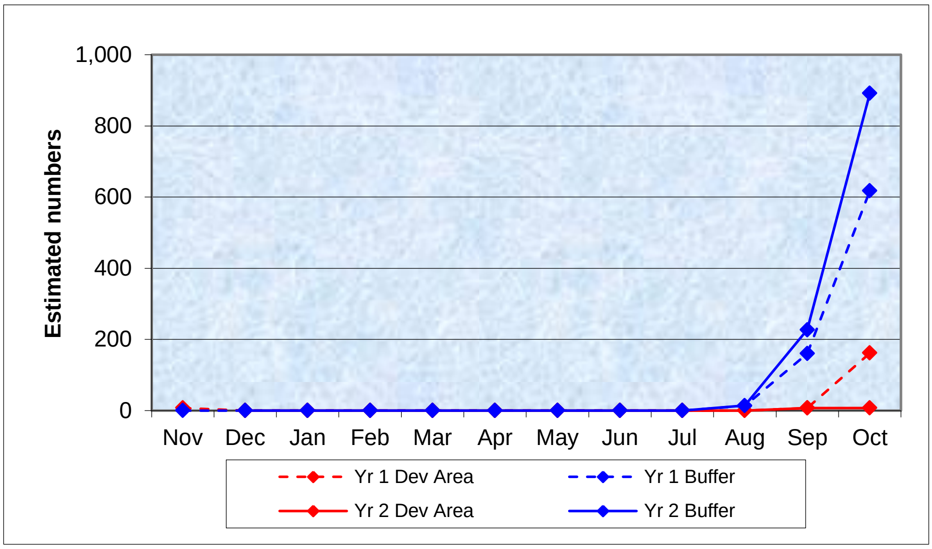


Figure 4.12 Monthly estimated numbers of sooty shearwaters in the Neart na Gaoithe Development & buffer areas in Years 1 and 2

Mean monthly sooty shearwater density was generally low in the offshore site and the buffer area throughout Years 1 and 2, apart from October, when mean density peaked at 1.3 birds/km^2 in the offshore site and 1.0 birds/km^2 in the buffer area in Year 1 (Figure 4.13). Mean density in October of Year 2 was slightly lower. ESAS density data from the surrounding ICES rectangles and across Regional Sea 1 was low in all months.

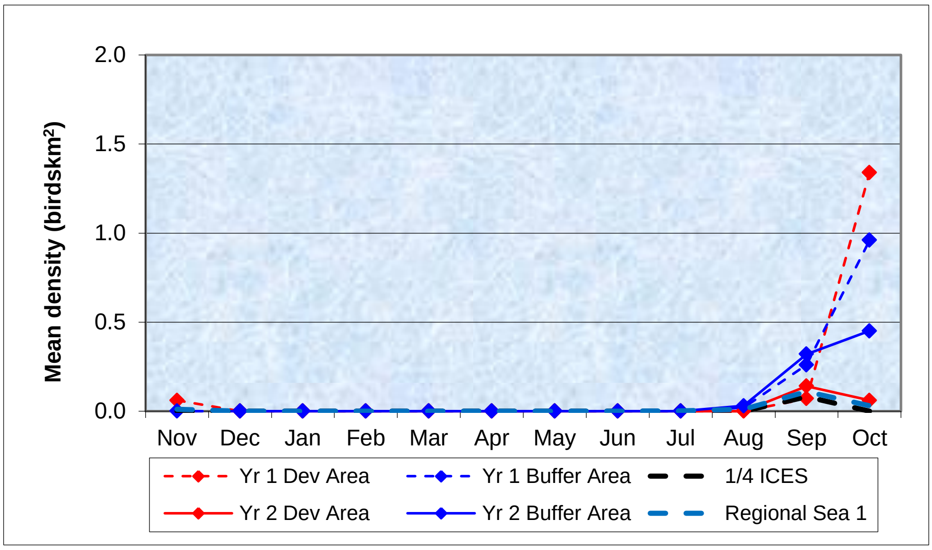


Figure 4.13 Comparison of sooty shearwater monthly mean densities in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

Sooty shearwaters were scattered across the Neart na Gaoithe study area at low to high densities, in October of Year 1 (Figure 4.14). Few birds were recorded in the east of the study area at this time.

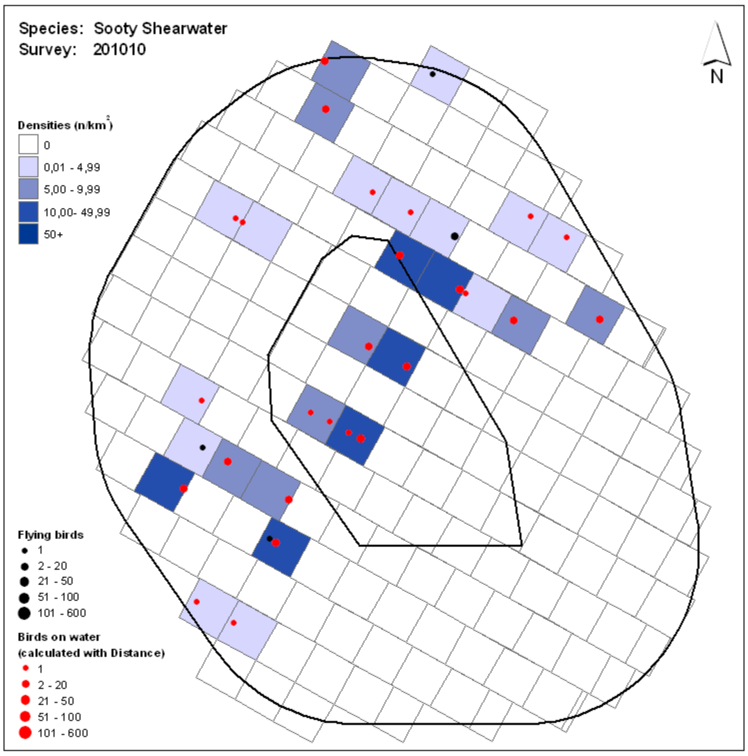


Figure 4.14 Sooty shearwater density in October, Year 1

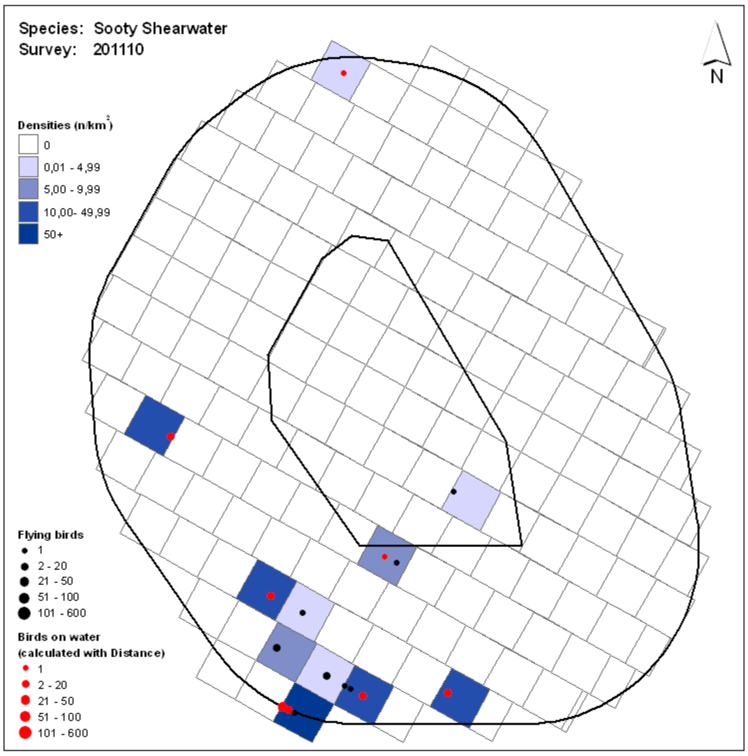


Figure 4.15 Sooty shearwater density in October, Year 2

In October of Year 2, sooty shearwaters were mostly recorded in the south of the Neart na Gaoithe study area at moderate to high densities (Figure 4.15).

A total of 93 sooty shearwaters were recorded in flight in Years 1 and 2, with all birds flying below 7.5 m in height (Table 4.3).

4.4.3.3 Species sensitivity

A recent review assessed sooty shearwater as being at low risk of collision and displacement resulting from offshore wind farms (Langston 2010) (Table 4.4). There are no SPAs designated for sooty shearwater in the breeding or non-breeding seasons in the UK (JNCC, 2012).

4.4.3.4 Assessment

Definition of seasons

Sooty shearwaters were only recorded in the offshore site between August and October. Therefore, this is the only period considered. This time of the year corresponds to the autumn passage and moulting period. Sooty shearwaters do not breed in the UK and the birds recorded in the offshore site are on migration from breeding grounds in the South Atlantic (Wernham *et al*., 2002).

Baseline conditions

In the autumn passage period (August to October) the mean estimated number of sooty shearwaters (birds on the sea and in flight) present in the offshore site was 30 birds, and the mean estimated number in the offshore site buffered to 1 km was 53 birds. On average 90.3% of birds present were on the sea and the remainder were in flight. The numbers present were similar in both survey years.

Populations

The size of the regional autumn passage population is imprecisely known and is likely to vary year-to-year, which presents a difficulty for undertaking the assessment. The regional population was approximately estimated at 2,500 birds from the summary ESAS results data presented by Stone *et al*. (1995).

There is no indication from local bird reports or results from Inch Cape and Round 3 surveys to suggest that the baseline survey years were atypical in terms of the numbers present in the Survey Area (Dickson, 2002; RPS, 2012; SWEL, 2011). The species is classed as being uncommon to common on autumn passage off the Fife coast (Dickson, 2002).

Nature conservation importance

For EIA assessment purposes, the nature conservation importance of sooty shearwaters using the site is rated at moderate during the autumn passage period. The species merits this rating as it is classified by IUCN as Near Threatened because, although it has a very large global population (20 million birds), it is thought to have undergone a moderately rapid decline owing to the impact of fisheries, the harvesting of its young and possibly climate change.

Offshore wind farm studies of sooty shearwater

There are very few records and therefore little field-based evidence of the likely effects of operational wind farms on sooty shearwaters. A review of offshore wind farm effects on birds categorised displacement, barrier and collision risk effects as unknown for sooty shearwater (Diersche and Garthe, 2006). At Horns Rev, Denmark, an area where this species is naturally scarce, there were only two sooty shearwater records, both of single birds flying outside the wind farm (Christensen *et al*., 2004). At Arklow Bank, Ireland, a single sooty shearwater was recorded outside the wind farm area.

In the absence of sooty shearwater data, evidence of the effects of wind farms on other closely related species, in particular Manx shearwater, can give an insight into their likely response to wind farms. At Arklow Bank, analysis of pre and post construction Manx shearwater data reported no significant change in numbers or distribution following construction of the wind farm (Barton *et al*., 2009). At Horns Rev and Egmond aan Zee, shearwaters were naturally too scarce to provide information on the likely effects of wind farms on sooty shearwater.

Based on the recorded flying heights of Manx shearwater and assuming similar flight behaviour for sooty shearwater it is likely that the risk of sooty shearwaters colliding with turbines is low. Of approximately 1,256 flying Manx shearwaters monitored over two years in the vicinity of the Arklow Bank wind farm, none were recorded flying at a height over 20 m above the sea surface and *ca.* 95% were flying at less than 5 m above the sea surface (Barton *et al*., 2009, Barton *et al*., 2010).

EIA Construction Phase assessment for sooty shearwater

The construction phase is of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

Sooty shearwaters are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during construction operations on the regional autumn passage populations of sooty shearwaters is ***not significant*** under the EIA Regulations.

EIA Operational phase assessment for sooty shearwater

Potential for sooty shearwater to be affected by displacement

The value of the offshore site buffered to 1 km as a foraging site to sooty shearwaters in the autumn passage period was estimated from the mean proportion of assumed regional population present from August to October (Table 4.13). On this basis, it is inferred that the offshore site buffered to 1 km provides foraging resources for up to approximately 2.1% of the regional population during the autumn passage period.

Table 4.13 The mean estimated number and percentage of the sooty shearwater population potentially at risk of displacement from the offshore site plus 1 km Buffer during the autumn passage period

Receptor population	Population size (adults)	No. assumed at sea	Dev.Area		Dev.Area + 1km	
			Mean no. at risk	%	Mean no. at risk	%
North Sea, autumn passage (derived from Stone *et al.*, 1995)	15,000	15,000	30	0.2%	53	0.4%
Regional, autumn passage (derived from Stone *et al.*, 1995)	2,500	2,500	30	1.2%	53	2.1%

Likely impacts of displacement on sooty shearwater population

It is assumed that 50% of the potential displacement would be realised. It is unknown whether displaced birds would be disadvantaged or whether they would merely move to alternative foraging areas with capacity to hold more birds. The species undertakes very large movements and therefore displaced birds are likely to move elsewhere to seek out alternative foraging sites. It is considered that the population of this species has low sensitivity to displacement (Langston, 2010) (Table 4.4).

Were 50% of sooty shearwaters to be displaced from the offshore site buffered to 1 km, the impact of this would be the effective loss of up to 1.1% of the foraging habitat used by the regional autumn passage population and categorised as low (1-5%) magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement from foraging areas on the regional population of sooty shearwaters in the autumn passage period is ***not significant*** under the EIA Regulations.

Barrier effects

Sooty shearwaters occur off East Scotland only as a passage migrant (Forrester *et al.*, 2007). During the non-breeding period, birds range over the North Atlantic and North Sea moving along broad migration fronts and are not spatially constrained by the need to attend breeding colonies. Although sooty shearwaters passing through the area may choose to detour around the offshore site the magnitude of such detours would be spatially and temporally negligible in magnitude compared to the ranging behaviour of this species. It is concluded that impact of any barrier effects on sooty shearwater are ***not significant*** under the terms of the Electricity Regulations.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

Sooty shearwaters are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional population of sooty shearwaters in the autumn passage period is ***not significant*** under the EIA Regulations.

Collision Mortality on National and Regional populations

CRM was not undertaken for sooty shearwaters because all birds seen in flight during the baseline surveys were below the proposed minimum rotor swept height of turbines. Therefore, it is not plausible that this species will experience significant mortality from collision with turbine rotors.

The potential effect of the collision mortality of sooty shearwaters on the baseline mortality rate is rated as negligible in magnitude (<1%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of collision mortality on sooty shearwaters is ***not significant*** under the terms of the EIA Regulations.

EIA Decommissioning Phase assessment for sooty shearwater

The decommissioning phase is of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

Sooty shearwaters are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during decommissioning operations on the regional population of sooty shearwaters in the autumn passage period is ***not significant*** under the EIA Regulations.

EIA summary of effects combined for sooty shearwater

The adverse impacts of the effects assessed will act in a broadly additive manner. In combination it is judged that the magnitude of the three effects on the population is low. However, because the population has low sensitivity to all effects, it is concluded that the overall impact on the regional population of sooty shearwaters in the autumn passage period is ***not significant*** under the EIA regulations (Table 4.14).

Table 4.14 Summary of effects on the regional autumn passage population of sooty shearwaters

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat	Low	Long term	Low	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality	Negligible	Long term	Low	***Not significant***
All effects combined	Low	Long term	Low	***Not significant***

4.4.3.5 Cumulative Impact Assessment for sooty shearwater

Combining the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts to the autumn passage populations of sooty shearwater will be as shown in Table 4.15. The individual impacts for displacement and collision are combined by addition to give the cumulative impact.

The predicted potential effective loss of up to 4.9% of the foraging habitat of the regional population of sooty shearwaters in the autumn passage period is rated as low in magnitude (1-5%), temporally long-term and reversible (Table 3.1 and Table 3.2). Bearing in mind the low sensitivity of this species to displacement and its rating of moderate nature conservation importance, it is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional population of sooty shearwaters in the autumn passage period is ***not significant*** under the terms of the EIA Regulations.

Table 4.15 Summary of CIA for the three proposed offshore wind farms in south-east Scotland on the regional population of sooty shearwaters in the autumn passage period

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF	Predicted impact from Inch Cape pOWF	Predicted impact from Firth of Forth R3 Zone pOWF	Cumulative Impact
Displacement: effective loss of foraging habitat during the autumn passage period (%)	50%	1.1%, Low	1.3%, Low	2.5%, Low	4.9%, Low
Collision: % increase in assumed annual adult mortality rate. All wind farm designs.	0%	No change, Negligible	No change, Negligible	No change, Negligible	No change, Negligible

Almost no sooty shearwaters are predicted to be killed by the three proposed offshore wind farms because almost all observed flight activity was well below rotor height. The potential impact of the collision mortality on the baseline mortality rate of the sooty shearwater regional population is rated as negligible in magnitude (<1%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional population of sooty shearwaters in the autumn passage period is ***not significant*** under the terms of the EIA Regulations.

4.4.3.6 Mitigation measures for sooty shearwater

The assessment does not identify any significant adverse effects on the regional population of sooty shearwaters in the autumn passage period. Therefore no mitigation measures are required for this species.

4.4.4 Manx shearwater *Puffinus puffinus*

4.4.4.1 Status

Manx Shearwater is a summer visitor to Scottish waters, occurring at breeding colonies between March and September. Seabird 2000 estimated the British breeding population at 295,089 breeding pairs, with large colonies on the west coast of Scotland e.g. Rum, and off the Welsh coast e.g. Skomer and Skokholm, with no breeding colonies on the east coast of Britain (Mitchell *et al.,* 2004).

Counts of more than 100 birds off the east coast of Scotland are uncommon, and the species is rare in Scottish waters in winter months, as birds migrate to the South Atlantic for the winter, primarily off the east coast of South America (Forrester *et al.,* 2007). Manx shearwaters spend most of their lives at sea, only coming ashore to breed. They typically eat small squid, fish, including sandeels and free-swimming crustaceans, which they catch by shallow plunge-diving or surface feeding (Forrester *et al.,* 2007).

4.4.4.2 Neart na Gaoithe Study Area

In Year 1, 72 Manx shearwaters were recorded on surveys in the Neart na Gaoithe study area (Table 4.2). The majority of birds (77.8%) were recorded in the buffer area, between May and October (Table 12). Larger numbers were present on surveys in Year 2, with a total of 286 birds recorded, the majority of which (90.6%) occurred in the buffer area, with only 27 birds seen in the offshore site.

Due to the low sample size of Manx shearwaters recorded in Years 1 and 2, it was not possible to conduct Distance analysis on the data. Simple abundance rates (birds/km) were calculated instead. Mean monthly Manx shearwater abundance was generally low in the offshore site and the buffer area in Years 1 and 2, with a peak of 0.7 birds/km in the buffer area in September of Year 2 (Figure 4.16). ESAS abundance data from the surrounding ICES rectangles and across Regional Sea 1 was low.

In Years 1 and 2, a total of 154 Manx shearwaters were recorded in flight, with almost all birds flying below 7.5 m in height (Table 4.3). One bird was recorded flying between 7.5 m and 12.5 m in height.

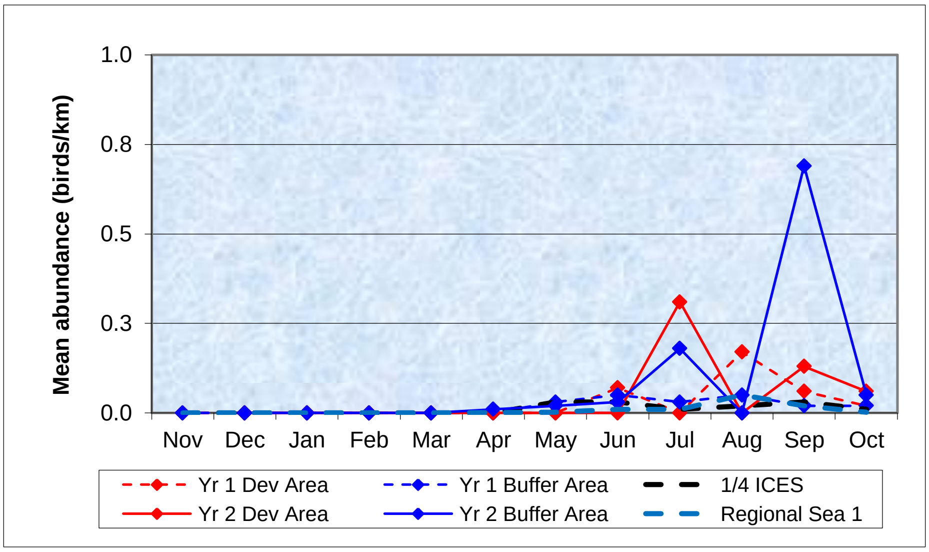


Figure 4.16 Comparison of Manx shearwater monthly mean abundance in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

Manx shearwaters were scattered across the Neart na Gaoithe study area at very low abundances, between May and October of Year 1 (Figure 4.17). Few birds were recorded in the offshore site over the period.

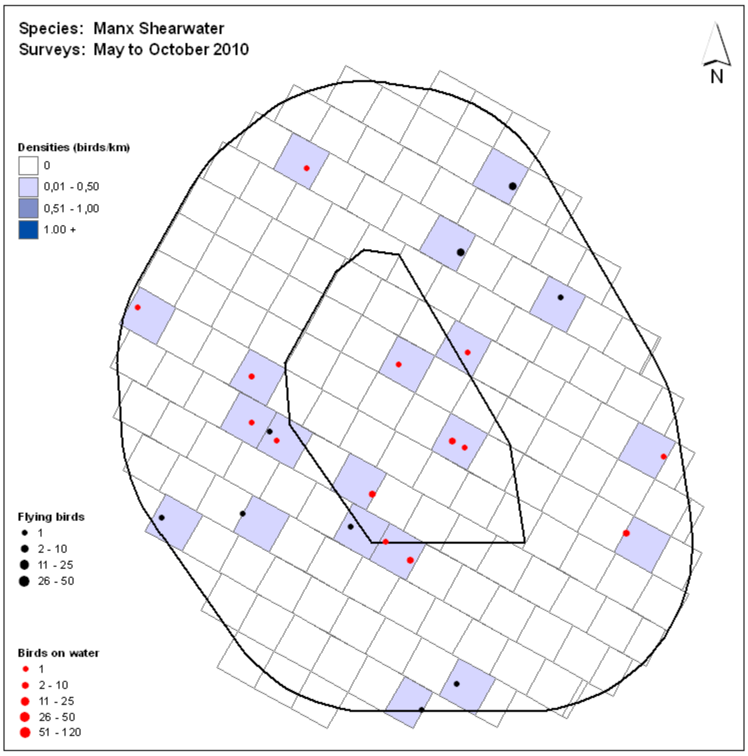


Figure 4.17 Manx shearwater abundance between May and October, Year 1

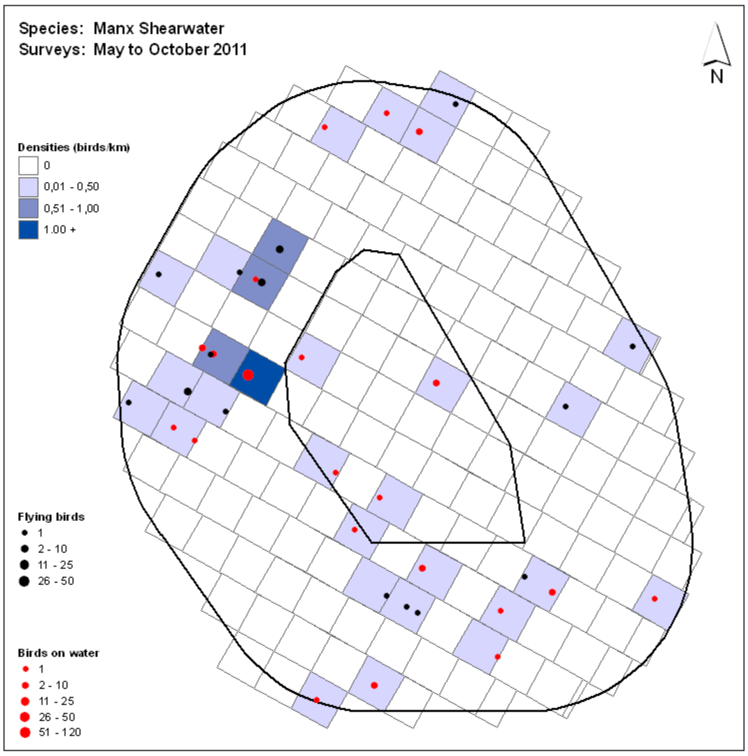


Figure 4.18 Manx shearwater abundance between May and October, Year 2

In Year 2, Manx shearwaters were scattered across the Neart na Gaoithe study area at low abundances, between May and October (Figure 4.18). Highest abundance at this time was recorded to the west of the offshore site.

4.4.4.3 Species sensitivity

Manx shearwater is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

A recent review assessed Manx shearwater as being at moderate risk of habitat loss or changes in prey distribution resulting from offshore wind farms. Risk of collision, and displacement were rated as low. Overall, Manx shearwater was assessed as being at high risk from offshore wind developments, due to the importance of the UK breeding population (Langston 2010) (Table 4.4).

There are no breeding colonies for Manx shearwater on the east coast of Scotland (Mitchell *et al*., 2004) and it is not listed as a qualifying interest species for SPAs in the region that could potentially be affected by the Neart na Gaoithe development.

4.4.4.4 Assessment

All Manx shearwaters recorded during baseline surveys are likely to be migrating birds, probably from breeding sites in western Scotland, or non-breeding individuals. Given that there are no colonies on the east coast of Scotland, the potential for large numbers to migrate through the offshore site is probably small (Wernham *et al*., 2002). This is corroborated by the relatively small number of birds recorded in Years 1 and 2. These findings taken together suggest that Manx shearwater is unlikely to be affected by the Development.

Displacement

It is not known whether Manx shearwaters will be displaced by the proposed Neart na Gaoithe development. The review undertaken by Langston (2010) suggested that Manx shearwaters are at low risk of displacement effects. Manx shearwaters were only recorded between May and October, with the majority of birds seen during autumn passage. The species is highly mobile and pelagic in nature and therefore will be able to relocate elsewhere should displacement effects occur. Overall, it is concluded that the effects of displacement on Manx shearwaters is ***not significant*** under the terms of the EIA Regulations.

Barrier effect

The nearest breeding colonies are on the west coast of Scotland and are beyond the maximum recorded foraging ranges for breeding birds (Thaxter *et al*., 2012). Therefore no barrier effects on breeding birds are predicted to occur during the breeding period. Outwith the breeding season Manx shearwaters migrate many thousands of kilometres to their wintering grounds in the western South Atlantic, most commonly off the coast of Brazil (Wernham *et al*., 2002). Therefore, the potential incremental increase in distance a Manx shearwater may fly should it fly around the wind farm will be negligible compared to the overall distance flown during migration. Overall, it is concluded that the effects of barrier effect on Manx shearwaters is ***not significant*** under the terms of the EIA Regulations.

Collision Mortality

Collision risk modelling was not undertaken for Manx shearwater, however all birds (n=154) recorded on baseline surveys were flying below 22.5 m. Based on this, it is predicted that Manx shearwaters will not experience significant mortality from collision with turbine rotors.

Overall, the potential effect of the collision mortality of Manx shearwaters on the baseline mortality rate is rated as negligible in magnitude, temporally long-term and reversible. It is concluded that the effects of collision mortality on Manx shearwaters is ***not significant*** under the terms of the EIA Regulations.

4.4.5 Balearic shearwater *Puffinus mauretanica*

4.4.5.1 Status

Balearic shearwater is listed as Critically Endangered on the 2010 IUCN (World Conservation Union) Red List, and is also listed on Annex I of the EU Birds Directive (2009/147/EEC), as it has a tiny breeding range on the Balearic Islands and a small, rapidly declining breeding population of between 6,000 and 10,000 individuals (Birdlife International, 2010). The species is also currently red-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

In Scottish waters, Balearic shearwater is a scarce visitor, occurring mainly in August and September, although there are records from all months except March (Forrester *et al.,* 2007). Most records are from the south-west of Scotland, but the species is classed as a rare passage migrant in inshore waters off the Fife coast (Dickson 2002).

4.4.5.2 Neart na Gaoithe Study Area

One Balearic shearwater was recorded on surveys in September of Year 1, flying west, below 7.5 m in height, in the south-east of the Neart na Gaoithe offshore site (Table 4.2) (Figure 4.19). No Balearic shearwaters were recorded in the Survey Area during Year 2 surveys.

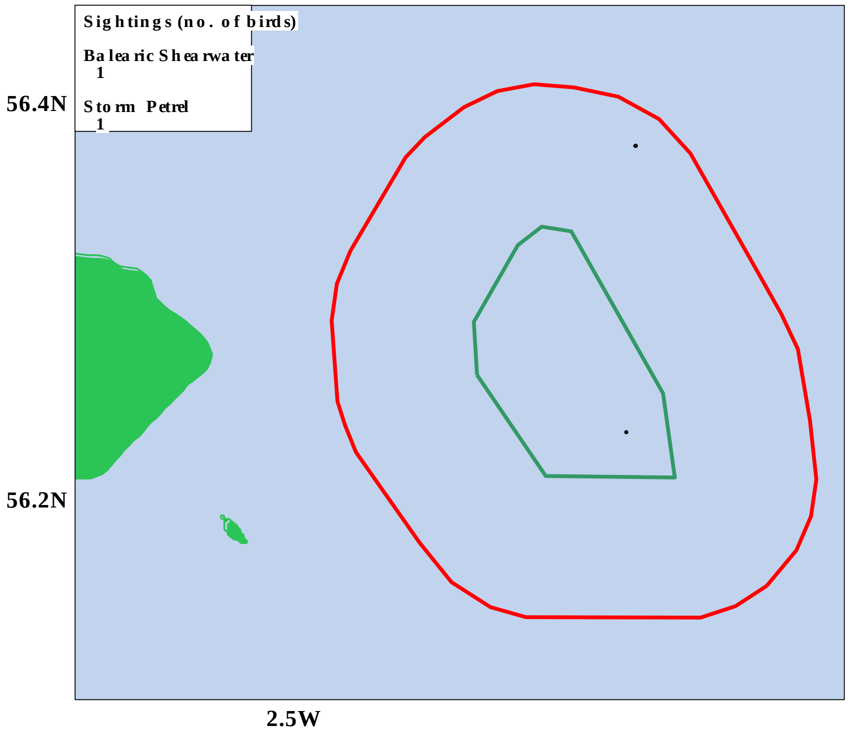


Figure 4.19 Balearic shearwater and storm petrel sightings in Year 1

4.4.5.3 Assessment

As only one Balearic shearwater was recorded in the offshore site in Year 1 baseline surveys, with no birds seen in Year 2, it is considered that Balearic shearwater is unlikely to be affected by the Development.

4.4.6 European Storm-petrel *Hydrobates pelagicus*

4.4.6.1 Status

Storm petrels breed at a few colonies around the UK, primarily on Shetland, Orkney, Western Isles and the west coast of Scotland, as well as on islands off the Welsh coast, Isles of Scilly and the Channel Islands. Seabird 2000 estimated the UK breeding population to be 25,710 pairs, however outside of Orkney and Shetland, there are no breeding colonies on the east coast of Britain (Mitchell *et al.,* 2004). After the breeding season, birds migrate south and spend the winter off the coast of southern Africa.

The species is uncommon off the east coast of Scotland, and is classed as a regular autumn passage migrant off the Fife coast (Dickson 2002).

4.4.6.2 Neart na Gaoithe Study Area

One storm petrel was recorded on surveys in October of Year 1, at the north end of the Neart na Gaoithe buffer area, flying north-west, below 7.5 m in height (Table 4.2) (Figure 4.19). The species was not recorded on Year 2 surveys.

4.4.6.3 Species sensitivity

Storm petrel is listed on Annex I of the EU Birds Directive (2009/147/EEC), and the species is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

A recent review assessed storm petrel as being at moderate risk of habitat loss or changes in prey distribution resulting from offshore wind farms. Risk of collision, and displacement were rated as low. Overall, storm petrel was assessed as being at moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

4.4.6.4 Assessment

As storm petrels were not recorded in the offshore site on baseline surveys, and only one bird was recorded in the buffer area in Year 1, with no birds seen in Year 2, it is considered that storm petrel is unlikely to be affected by the Development.

4.4.7 Northern Gannet *Morus bassanus*

4.4.7.1 Status

Gannets breed in a few, typically very large, colonies around the UK. The second largest UK colony is at Bass Rock, in the outer Firth of Forth, with an estimated breeding population of 55,482 nests in 2009 (Murray 2011).

Gannets feed by plunge diving for fish, typically from around 25 to 30 m above the surface (BTO, 2012). Breeding birds from the Bass Rock colony have been shown by satellite-tagging to range widely across the North Sea, at times travelling as far as the Norwegian coast (Hamer *et al*., 2007). Regular feeding movements occur to the north-east with concentrations of feeding locations off North-east Scotland (Hamer *et al*., 2011).

4.4.7.2 Neart na Gaoithe Study Area

Gannet was the commonest seabird recorded on surveys in the Neart na Gaoithe study area during the baseline surveys, with 13,021 birds recorded in Year 1 and 19,416 birds recorded in Year 2. The majority of gannets were recorded in the buffer area (Table 4.2). Highest numbers were recorded during the breeding season (March to September).

During the breeding season (March to September), the peak estimated number of gannets occurred in September of Year 1, with an estimated 739 birds in the offshore site, and 3,366 birds in the buffer area (Table 4.16) (Figure 4.20). In Year 2, peak estimated numbers of gannets in the offshore site peaked in April (1,634 birds), while estimated numbers in the buffer area peaked in September (5,590 birds).

In the non-breeding season (October to February), the peak estimated number of gannets occurred in October of Year 1, with an estimated 171 birds in the offshore site, and 1,082 birds in the buffer area (Table 4.16) (Figure 4.20). In Year 2, peak estimated numbers of gannets in the offshore site peaked in February (408 birds), while estimated numbers in the buffer area peaked in October (3,295 birds).

Overall, numbers in the Neart na Gaoithe buffer area exceeded 1 % of the national breeding population (4,411 birds) (Wanless *et al.,* 2005) in June, August and September of Year 2 (Figure 4.20).

Table 4.16 Estimated numbers of gannets in the offshore site and 8 km buffer in Years 1 and 2

Month	Estimated nos on water Offshore Site	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying Offshore Site	Estimated total Offshore Site	Estimated total 8 km buffer [1]	Estimated total
Yr1 Nov	0	0	0	20	20	279	299
Yr1 Dec	0	0	0	0	0	29	29
Yr1 Jan	0	0	0	7	7	13	20
Yr1 Feb	0	0	0	20	20	374	394
Yr1 Mar	17	8	36	462	479	1,089	1,568
Yr1 Apr	10	5	21	129	139	943	1,082
Yr1 May	10	4	22	150	160	1,434	1,594
Yr1 Jun	113	55	230	237	350	1,331	1,681
Yr1 Jul	0	0	0	89	89	1,082	1,171
Yr1 Aug	67	35	127	177	244	2,879	3,123
Yr1 Sep	386	272	549	353	739	3,366	4,105
Yr1 Oct	8	5	13	163	171	1,082	1,253
Yr2 Nov	-	-	-	-	-	-	-
Yr2 Dec	0	0	0	7	7	73	80
Yr2 Jan	0	0	0	14	14	651	665
Yr2 Feb	34	14	80	374	408	546	954
Yr2 Mar	16	7	34	79	95	1,221	1,316
Yr2 Apr	1,457	823	2,578	177	1,634	3,499	5,133
Yr2 May	15	6	41	800	815	3,398	4,213
Yr2 Jun	34	21	56	603	637	4,995	5,632
Yr2 Jul	276	192	396	1,248	1,524	2,823	4,347
Yr2 Aug	45	25	83	731	776	5,256	6,032
Yr2 Sep	295	203	430	628	923	5,590	6,513
Yr2 Oct	85	53	135	305	390	3,295	3,685

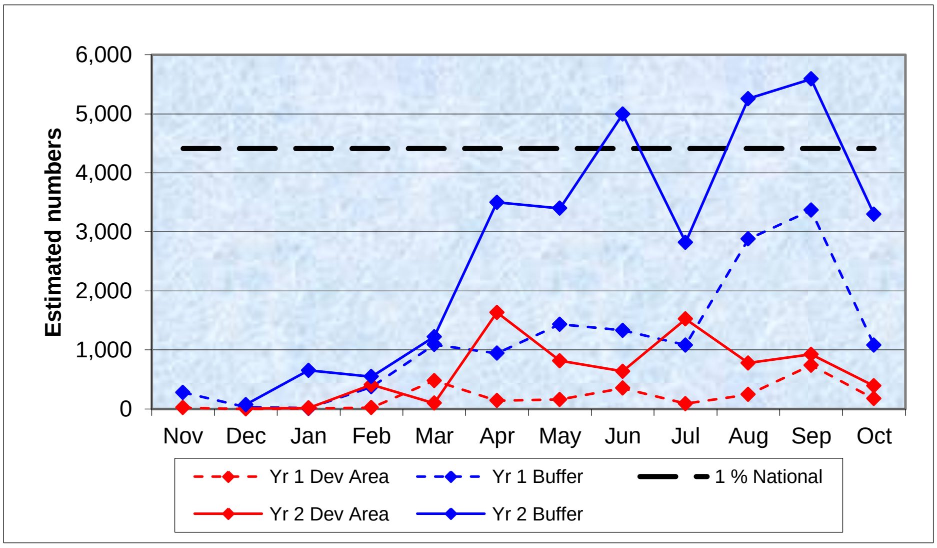


Figure 4.20 Monthly estimated gannet population of Neart na Gaoithe Development & buffer areas in Years 1 and 2

Mean monthly gannet density was similar between the offshore site and the buffer area in Year 1, with a peak in September (Figure 4.21). In Year 2, mean densities showed a similar pattern, although mean density in the offshore site was generally higher than in the buffer area, with a peak in April (18.5 birds/km^2). ESAS mean density data for gannet in the surrounding ICES rectangles and Regional Sea 1 was generally lower than recorded in the study area, most likely reflecting the greater distance from the breeding colony on the Bass Rock.

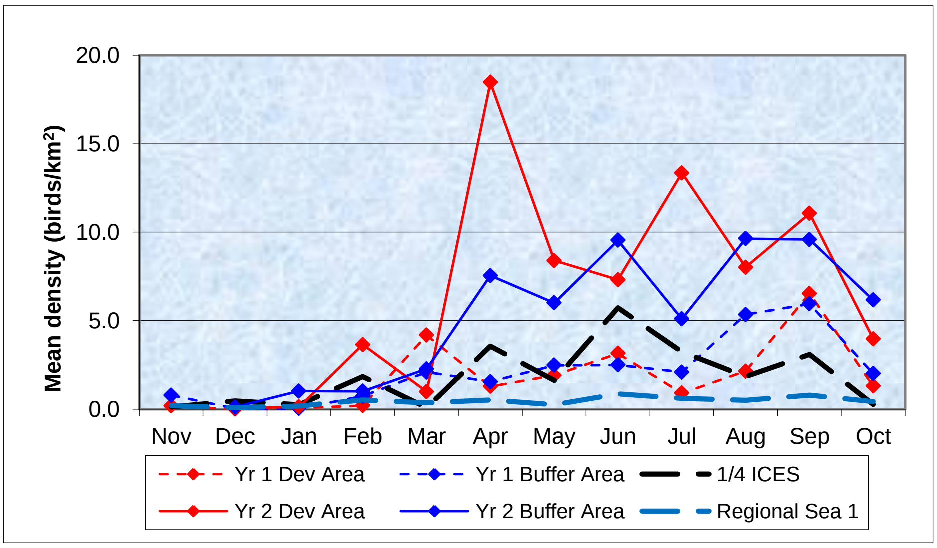


Figure 4.21 Comparison of monthly mean densities for gannet in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

Between November and February of Year 1, gannets were scattered at low to moderate densities across the study area, with low densities recorded in the offshore site at this time (Figure 4.22).

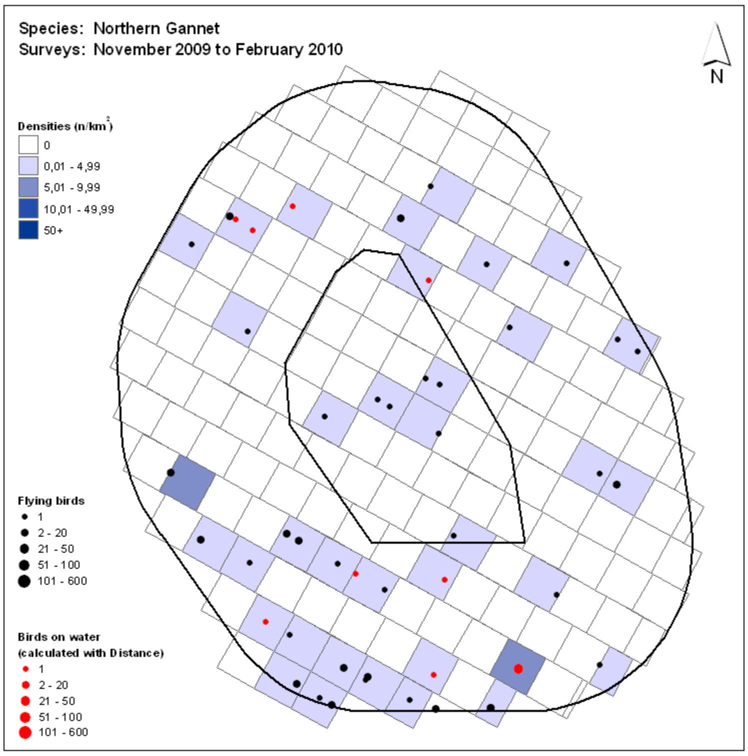


Figure 4.22 Gannet density between November and February, Year 1

Between October and February of Year 2, gannets were more widespread at low to moderate densities across the Neart na Gaoithe study area than was recorded in Year 1 (Figure 4.23). Highest densities were recorded in the south-east of the buffer area at this time.

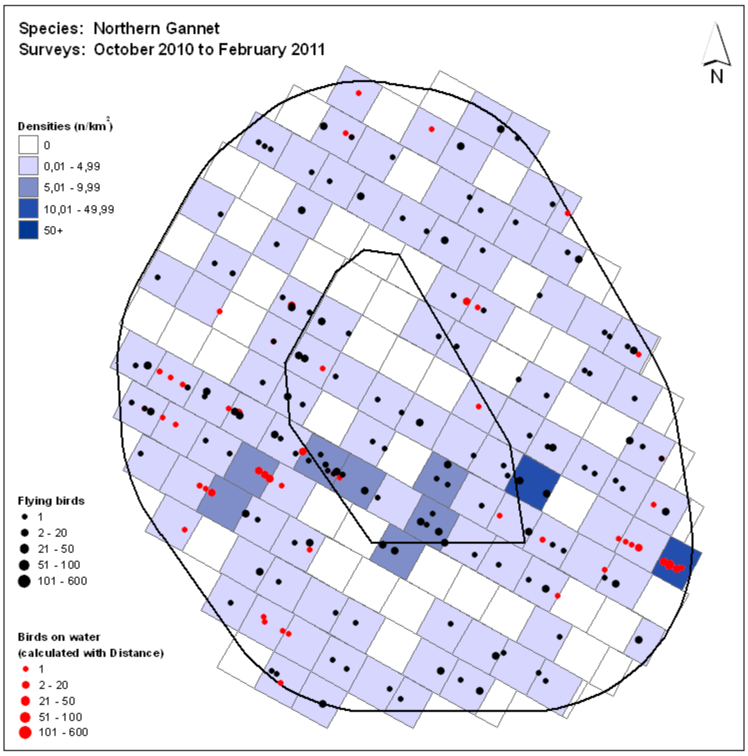


Figure 4.23 Gannet density between October and February, Year 2

Gannets were more numerous and widespread in the Year 1 breeding season (March to September), with low to moderate, occasionally high densities recorded across the Neart na Gaoithe study area over the period (Figure 4.24).

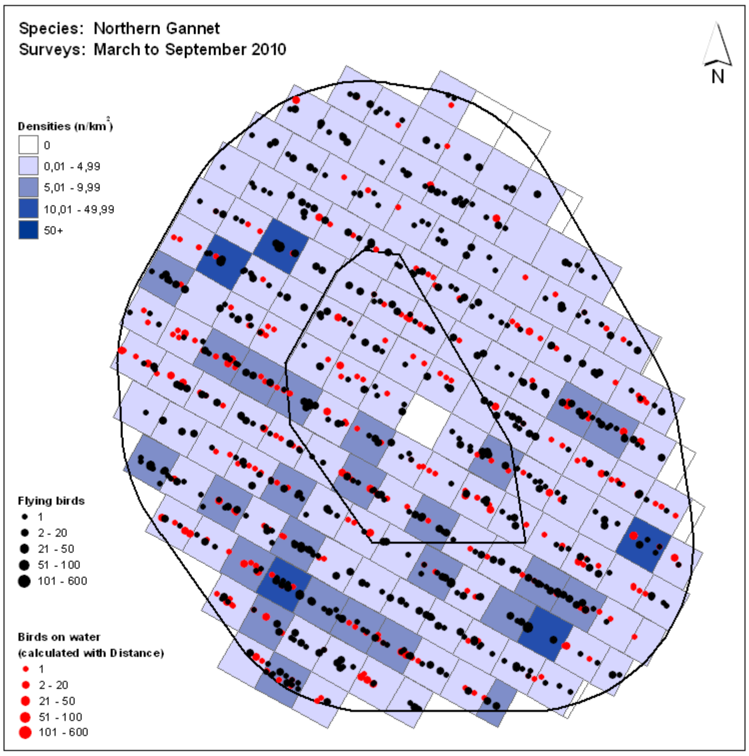


Figure 4.24 Gannet density between March and September, Year 1

Gannet density in the Neart na Gaoithe study area In the Year 2 breeding season was generally higher than recorded in the same period in Year 1, with moderate to high densities recorded in the south-western half of the study area, and mostly low to moderate densities recorded in the north-eastern half (Figure 4.25).

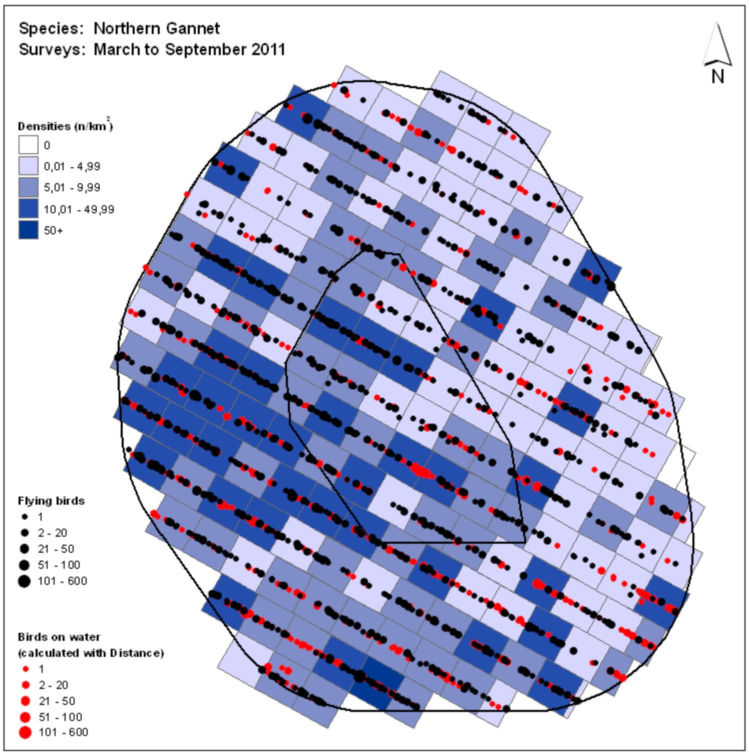
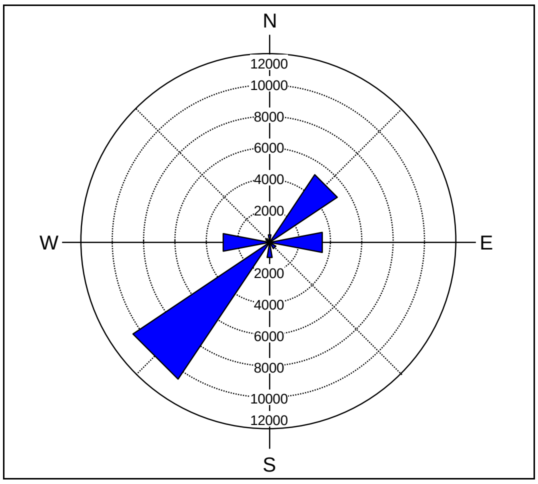


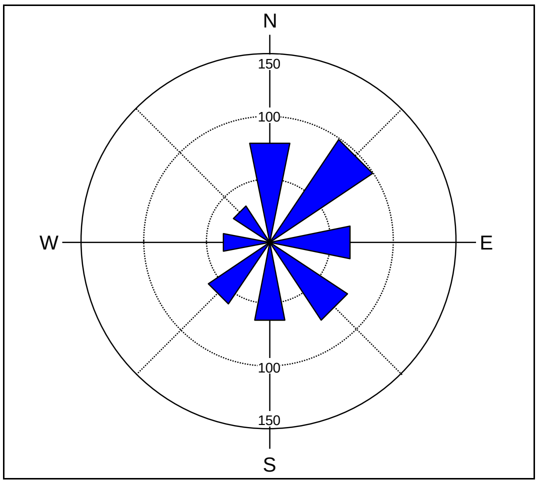
Figure 4.25 Gannet density between March and September, Year 2

In Years 1 and 2, a total of 28,824 gannets were recorded in flight, with 93.5% (26,960 birds) of birds flying below 22.5 m in height i.e. below the turbine swept zone (Table 4.3). A total of 1,864 birds (6.5%) were recorded flying above 22.5 m, i.e. within the rotor swept zone, with the majority of these birds (79.9%) recorded at an estimated 30 m or less. The maximum height estimated was 70 m.

Flight direction was recorded for 24,389 gannets in the breeding season (March to September), with direction recorded for 2,815 gannets in the non-breeding season (October to February) (Figure 4.26).

In the breeding season, slightly less than half of all birds recorded were flying south-west (43.0%) in the general direction of the Bass Rock breeding colony, with 21.3% of birds flying north-east. In the non-breeding period, just over a quarter of birds (26.9%) were recorded flying north-east, with 20.4% flying east. An additional 1,262 birds were recorded as circling (not shown).





March to September (n=24,389 birds) October to February (n=2,815 birds)
Numbers shown on figures are number of birds recorded

Figure 4.26 Flight direction of gannets in the Neart na Gaoithe Study Area in Years 1 and 2

Foraging behaviour was recorded for 1,505 gannets in the Neart na Gaoithe study area in Years 1 and 2, with nine types of foraging behaviour recorded, and unspecified feeding behaviour recorded for a further 30 birds (Table 4.17) (Figure 4.27). Deep plunging was the most frequently recorded foraging behaviour (67.9%).

Table 4.17 Gannet foraging behaviour in the Neart na Gaoithe Study Area in Years 1 and 2

Behaviour	Number of birds
Actively searching	276
Deep plunging	1,022
Pursuit plunging	40
Shallow plunging	5
Surface seizing	6
Scavenging	1
Scooping prey from surface	20
Scavenging at fishing vessel	104
Holding fish	1
Feeding method unspecified	30
Total	**1,505**

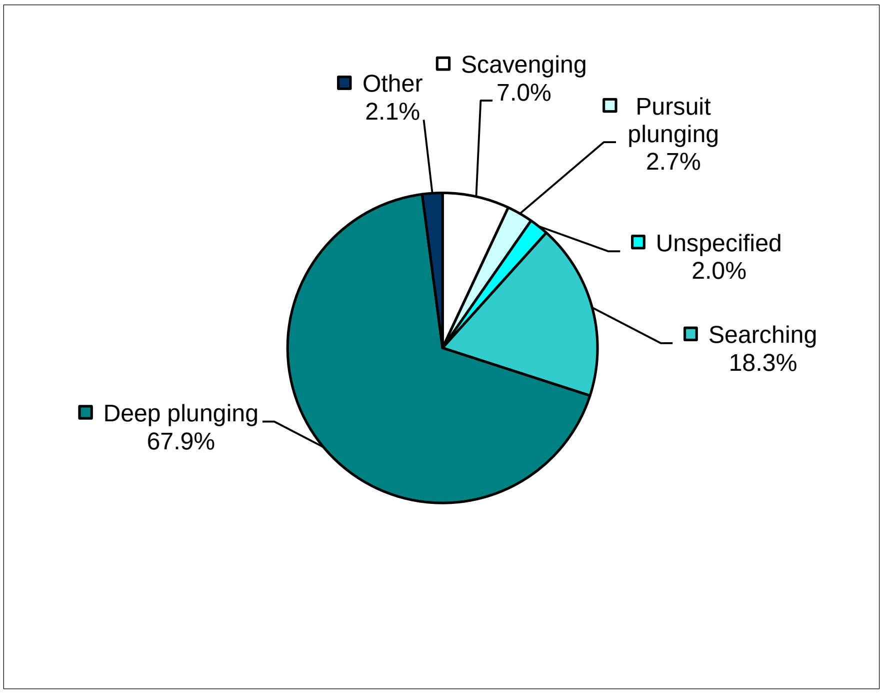


Figure 4.27 Foraging behaviours of gannets in the Neart na Gaoithe Study Area in Years 1 and 2

4.4.7.3 Species sensitivity

Gannet is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009).

A recent review assessed gannet as being at moderate risk of collision resulting from offshore wind farms. Risk of displacement, barrier effects, habitat loss or changes in prey distribution were rated as low. Overall, gannet was assessed as being at moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

Gannet is listed as a qualifying interest species in the breeding season for five SPAs on the UK east coast that could potentially be affected by the Neart na Gaoithe development (Table 4.18). These SPAs held 28.5% of the UK breeding population, and 21.9% of the biogeographic population at the time of designation (JNCC, 2012). Since designation, the populations at all five SPAs have expanded, with the Bass Rock colony (Forth Islands SPA) being the largest in the region (SMP, 2012).

The distance between the offshore site and the Bass Rock colony is within the mean maximum foraging range of 229.4 km, while the distance to the other four SPAs is within the maximum known foraging range of 590 km (Thaxter *et al.*, 2012). Gannet mean maximum foraging range from the two closest breeding SPAs in relation to the offshore site are shown in Figure 4.28.

Table 4.18 SPAs for breeding gannets between Hermaness and Spurn

SPA site	Distance to site (km)	Site total (pairs)[1]	% of biogeo-graphic popn[1]	% of national popn[1]	Recent count[2]	Year
Fair Isle	*356*	*1,166*	*0.4*	*0.6*	*3,582*	*2009*
Forth Islands (Bass Rock)	**16**	**34,400**	**13.1**	**17.1**	**55,482**	**2010**[3]
Flamborough Head & Bempton Cliffs	*259*	*2,501*	*0.95*	*1.2*	*7,859*	*2009*
Hermaness, Saxa Vord & Valla Field	*510*	*12,000*	*4.6*	*6.0*	*24,353*	*2008*
Noss	*428*	*7,310*	*2.8*	*3.6*	*8,652*	*2003*
Total	**-**	**57,377**	**21.9**	**28.5**	**92,511**	**-**

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. 3 Murray 2011. Sites in italics lie within the maximum known foraging range of 590 km. Sites in bold lie within the mean maximum foraging range of 229.4 km (Thaxter *et al.*, 2012).

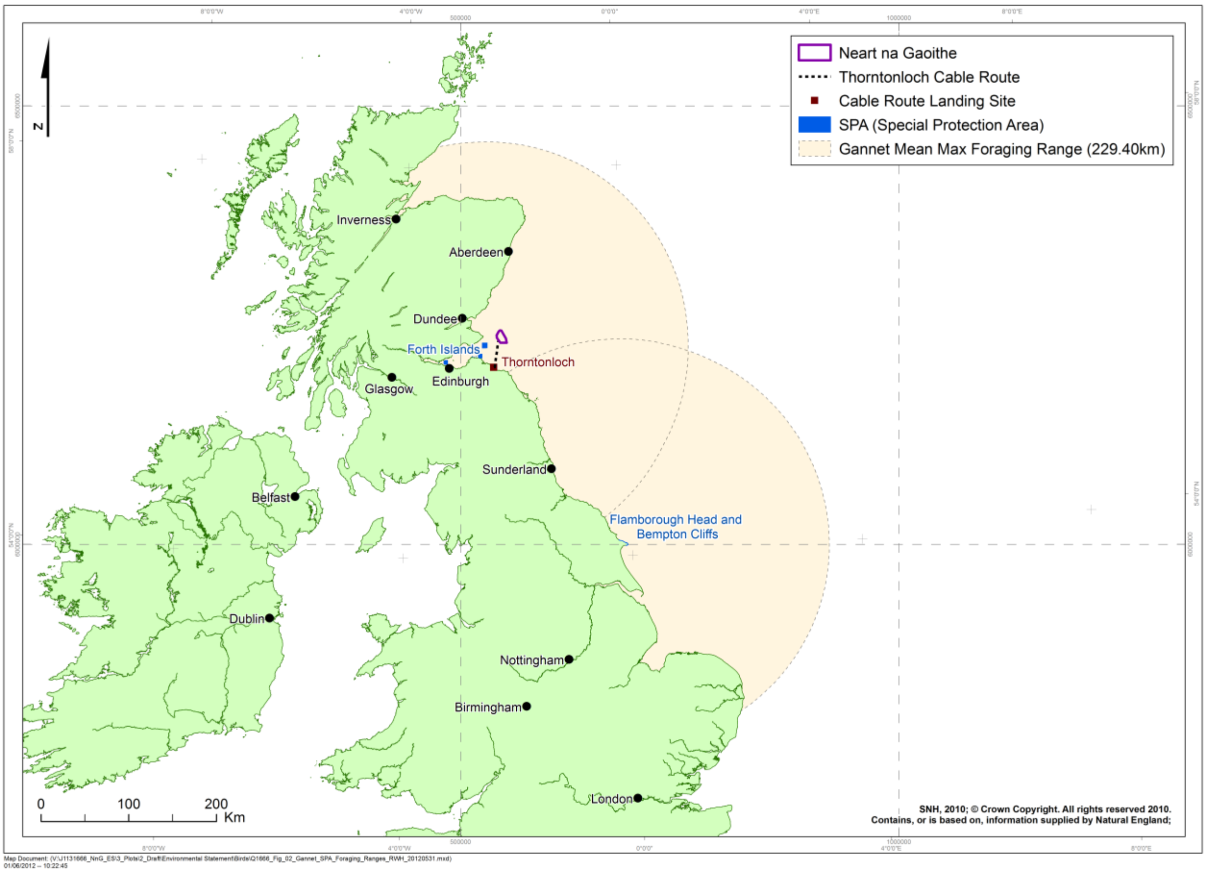


Figure 4.28 Gannet mean maximum foraging range from breeding SPAs in relation to the Development

A recent JNCC statistical analysis of ESAS data investigating possible marine SPAs, identified the waters around the Firth and Tay estuaries Neart na Gaoithe study area as an important area for gannets during the breeding season (Kober *et al.*, 2010).

4.4.7.4 Assessment

Definition of seasons

The annual cycle for gannet is divided into two parts to reflect the biology of the species and the broad pattern of use of the offshore site apparent from the baseline surveys.

The breeding season, the period when breeding adults are attending colonies, is defined as March to September. At this time the vast majority of birds present in the offshore site will be from the Bass Rock breeding colony, the nearest colony to the development.

The non-breeding period is defined as October to February and broadly corresponds to the period when gannets are in their over-wintering area. In this period it is likely that individuals present in the offshore site are a mixture of birds from the Bass Rock and other breeding colonies, including birds from colonies in northern Scotland and Norway (Wernham *et al.,* 2002). Tagging studies of birds from the Bass Rock indicate that the majority winter off West Africa and in the Mediterranean and a significant proportion wintering in the Bay of Biscay or Celtic Sea (Hamer *et al*., 2011).

Baseline conditions

In the breeding season the mean estimated number of gannets present in the offshore site was 627 birds, and the mean number in the offshore site buffered to 1 km was 870 birds. On average 27% of birds present in the breeding season were on the sea and the remainder were in flight. On average approximately 35% of the birds present were either sitting on the sea or demonstrating foraging behaviour when seen. The remaining birds were making direct flights through the area, presumably to and from foraging areas elsewhere.

Based on survey results for the whole Survey Area in this period, 97.1% of gannets for which age was determined were adults, with the remainder immature or juvenile birds. For the purposes of assessment it is assumed that all adult birds were breeding. The very low proportion of immature birds present in the summer strongly suggests that the vast majority of this component of the population was summering away from the breeding grounds. Studies have estimated that mean colony attendance rate at the Bass Rock colony during the breeding season is 43% (Hamer *et al.,* 2007) and this figure is used in estimates of the breeding season importance of the offshore site.

In the non-breeding period (October to February) the mean estimated number of gannets present in the offshore site was 121 birds, and the mean number in the offshore site buffered to 1 km was 164 birds. On average 29% of birds present in the non-breeding period were on the sea and the remainder were in flight. In this period 91.0% of individuals for which age was determined were adults.

The number of gannets recorded during Year 2 of the baseline surveys was much greater than in Year 1; in the breeding season approximately twice as many were seen in Year 2 compared to Year 1 and in the non-breeding period over three times as many were recorded in Year 2. The reasons for greater numbers in Year 2 are unknown but are likely to be linked to natural changes in prey distribution.

Populations

The regional breeding gannet population is defined as all birds breeding less than 229.4 km from the offshore site, the mean maximum foraging range for breeding gannet (Thaxter *et al.,* 2012). The only colonies within this distance are Bass Rock and Troup Head. Recent censuses estimate 55,482 occupied nests at Bass Rock in 2009 (Murray, 2011) and 2,787 occupied nests at Troup Head in 2010 (SMP, 2012). On this basis the regional breeding population is assumed to be 116,538 breeding adults (58,269 pairs). The Bass Rock population has shown a long term increase, averaging 4.0% per annum between 1985 and 2009 (Murray, 2011). To prevent this population change causing assessment of effects to be biased high, the most recent count data (Murray, 2011, SMP, 2012) are used for the assessments of effects, rather than the Seabird 2000 results.

The size of the non-breeding-period regional gannet population is assumed to be 31,200 birds. This is derived by summing the November to February period estimates for localities 4, 5 and 6 given in Skov *et al*. (1995). Although it is likely that many of the birds present in the non-breeding period are from the regional breeding population, birds from further afield, including colonies in Norway, are also likely to be present at this time (Wernham *et al*., 2002).

Nature conservation importance

For EIA assessment purposes, the nature conservation importance of gannets using the site is rated as High throughout the year on account of the regular presence of large numbers of individuals from one or more designated internationally designated population. The mean number present in the breeding season in the offshore site buffered to 1 km exceeds 1% of the regional breeding population. The majority of birds are from the Bass Rock, an internationally important breeding site 27 km south-west of the offshore site, and a component of the Forth Islands SPA.

Offshore wind farm studies of gannet

Post construction monitoring at operational wind farms indicate that gannets are likely to be largely displaced from, and deflected around the footprint of the proposed wind farm (PMSS, 2006, Christensen *et al*., 2004, Leopold *et al*., 2011, Diersche and Garthe, 2006). Furthermore, gannets entering the Egmond aan Zee wind farm in the Netherlands always stopped foraging, decreased flight height to <10 m (i.e. well below rotor height) and cut across its margin suggesting that habitat loss in terms of foraging area is likely to be effectively total within the footprint of the wind farm (Leopold *et al*., 2011). Note however, that most gannets recorded during monitoring at operational wind farms occurred outside the breeding season and there is therefore some uncertainty as to the response of breeding birds, as will be the case at Neart na Gaoithe. Nonetheless, the consistent reports of very high displacement of gannets from offshore wind farms in a variety of different study situations in European marine areas suggests that it is likely that this is how this species will respond to Neart na Gaoithe.

Results of radar and visual studies indicate that gannets are deflected around or away from wind farms when they approach relatively closely to the perimeter (Petersen *et al*., 2006, Leopold *et al*., 2011). For example, at the Egmond aan Zee flying gannets approaching the wind farm changed course as close as 500 m from the perimeter (Lindeboom *et al*., 2011). This is corroborated by radar studies from Horns Rev, Denmark where gannets typically changed course between 500 m and 1000 m from the perimeter (Christensen and Hounisen, 2005), and comments on observed flight-route behaviour at North Hoyle, Wales (RWE Group).

No records of gannets colliding with wind turbines were reported by Diersche and Garthe (2006) in a literature review on the effects of offshore wind farms on seabirds. This review categorised collision risk for gannet as unknown. Of approximately 303 flying gannets recorded during a two-year monitoring period at Arklow Bank wind farm and a surrounding buffer, only *ca.* 10-15% of birds were recorded flying at a height over 20 m (Barton *et al*., 2009, Barton *et al*., 2010). Collision risk will be dependent on the proportion of the at-risk population displaced from the wind farm footprint and the flight behaviour of birds that are not displaced (Leopold *et al*., 2011).

Evidence of gannets flying low through turbine arrays has been suggested as birds habituating to the Egmond aan Zee wind farm (Leopold and Camphuysen, 2008). This would reduce the energetic costs associated with birds detouring around wind farms to access foraging areas.

EIA Construction Phase assessment for gannet

The construction phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

Gannets are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during construction operations on the regional populations of gannets in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

EIA Operational Phase assessment for gannet

Potential for gannet to be affected by displacement

The value of the offshore site buffered to 1 km as a foraging site to gannets from the regional breeding population (essentially the same as the Bass Rock population) was estimated from the proportion of the adults estimated to be at sea (not attending a colony) that were on average present in the offshore site buffered to 1 km during the breeding season (Table 4.19).

Based on the studies reported above, for the purpose of assessment of displacement it is assumed that there will be complete displacement of gannets from the proposed development and a surrounding buffer of 1 km. This is likely to be a cautious assumption as some gannets are likely to continue to forage in the 1 km buffer, at least.

Table 4.19 The mean estimated number of gannets present during the breeding and non-breeding periods, and this value as a percentage of the (at-sea) receptor population potentially at risk of displacement

Receptor population (Source)	Population size (adults)	No. assumed at sea	Dev.Area		Dev.Area + 1km	
			Mean no. at risk	%	Mean no. at risk	%
National, breeding period (Seabird 2000)	440,963	251,349	627	0.2	870	0.3
North Sea, non-breeding period (Skov *et al*., 1995)	157,800	157,800	121	<0.1	164	0.1
Regional, breeding period (Murray 2012)	116,538	66,427	627	0.9	870	1.3
Regional, non-breeding period (Skov *et al*., 1995)	31,200	31,200	121	0.4	164	0.5

It is estimated that on average during the breeding season 1.3 % (870 adults) of the at-sea regional breeding population (i.e. adults not attending the breeding colonies) were present in the offshore site buffered to 1 km (Table 4.19), and that, on the basis of their behaviour recorded during the baseline surveys, approximately 35% (305 adults) of these were potentially foraging (i.e. not merely flying over). These birds are potentially at risk of being displaced from foraging in this area by the proposed wind farm.

In the same way, it is estimated that 0.5% (164 birds) of the regional population in the non-breeding period are on average present in the offshore site buffered to 1 km (Table 4.19) and therefore that this area provides on average up to approximately 0.5% of the foraging resource for the regional population in the non-breeding season.

Likely impacts of displacement on gannet populations

For the purposes of assessment of displacement it is assumed that there will be complete displacement of gannets from the offshore site buffered to 1 km, and, therefore, the potential for displacement effects described above will be fully realised. Research on foraging ranges indicates that Bass Rock gannets have a high capacity to use additional potential feeding areas by flying further from the breeding colony (Hamer *et al.,* 2007). Therefore, it is likely that gannets can compensate for a moderate amount of displacement by choosing to forage elsewhere. On this basis, the regional breeding gannet population is considered to have low sensitivity to displacement effects and it is therefore unlikely that the predicted displacement will result in any discernible population effects on the regional breeding population.

If all gannets were to be displaced during the breeding season from the offshore site buffered to 1 km the impact of this would be the effective loss of up to 1.3% of the foraging habitat of the regional population in the breeding season. This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of displacement on the regional gannet population in the breeding season are ***not significant*** under the EIA Regulations.

If all gannets were to be displaced during the non-breeding period from the offshore site buffered to 1 km the impact of this would be the effective loss of up to 0.5% of the foraging habitat of the regional population in the non-breeding period. This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of displacement on the regional gannet population in the non-breeding period are ***not significant*** under the EIA Regulations.

Barrier Effect

There are two gannet colonies that are within the mean maximum foraging range; Troup Head and Bass Rock. Birds from Troup Head are predicted to occur mainly around that colony in the outer Moray Firth and therefore, during the breeding season nearly all the gannets in the study area originate from the Bass Rock (Hamer *et al*., 2011).

For the purposes of assessment the width of the barrier is assumed to extend 1 km either side of the maximum width of the proposed wind farm. Given the reported response of gannets to offshore wind farms it is further assumed for assessment purposes that the wind farm will present a complete barrier to gannets. This assumption is cautious because it is possible that some gannets will fly through the wind farm. Barrier effects as calculated here concern birds which would otherwise fly through the offshore site to access feeding resources beyond it.

The proposed development buffered to 1 km would potentially form a barrier of 17.8 km wide 27 km north-east of the Bass Rock. This barrier would potentially block approximately 27% of the possible flight directions available to gannets flying out to distances in excess of 27 km from the Bass Rock (Table 3.6). For the purposes of assessment it is assumed that 27% of foraging flights by breeding gannets on Bass Rock (equivalent to 26% of birds breeding in the region) would be affected by the offshore site acting as a barrier. In one respect this figure is likely to be cautious (biased high) because the destinations of some of these flights will be closer than the barrier. However, it is also possible that gannets preferentially select the direction affected by a barrier above the other directions available, in which case the assumed proportion of flights affected could be biased low. No attempt is made to correct for these potential biases; it is likely that they will cancel each other to some extent. The mean foraging distance for gannet is 92 km (Thaxter *et al.,* 2012). Therefore almost all flights in the direction of the wind farm are likely to have destinations beyond the offshore site.

The potential effect the barrier would have on flight distances and times depends on how far the destination areas lie beyond the barrier. The results from tagging studies on gannets breeding at Bass Rock show that they forage over a vast area of the northern North Sea;

commonly travelling distances in excess of 150 km and sometimes up to three times this distance. The mean maximum distances recorded from tagged gannets from the Bass Rock varies across years, depending on food availability, but ranges from between 170 km and 363 km (Hamer *et al.,* 2000; Hamer *et al.,* 2011). It is therefore reasonable to assume that likely destinations of gannet flights affected by barrier effects due to the Development will be at a wide range of distances beyond the offshore site, and commonly many tens of kilometres beyond.

The mean destination distance of gannet foraging flights is 92 km (Thaxter *et al.,* 2012). Acknowledging there is uncertainty in how far on average the destination distance of affected flights are from the colony, for the purpose of assessment a cautious value of 90 km is assumed. This would mean that the flight routes of birds affected by a barrier effect would be increased by approximately 2.2% (Table 3.7).

It is estimated that around 26 % of foraging flights of breeding gannets in the region could be affected by the proposed wind farm acting as a barrier. This potential adverse effect is rated as low in magnitude (although it affects 26 % of individuals in the population, the magnitude of the effect on these birds is rated as low (1-5%)) and temporally long-term and reversible (Table 3.1 and Table 3.2).

Studies on foraging gannets have shown that they are capable of extending foraging distances in response to distribution of prey, suggesting that the species would easily absorb the minor increases in flight distances that a barrier could cause (Hamer *et al.,* 2007; Hamer *et al*., 2011). On this basis, they appear to have a low sensitivity to barrier effects. This species is also rated as low for sensitivity to barrier effects by Maclean *et al.* (2009) and Langston (2010). Therefore, it is concluded that any adverse effects on the regional gannet population in the breeding season caused by the proposed wind farm acting as a barrier is ***not significant*** under the terms of the Electricity Act.

Collision mortality

The results of CRM for the regional breeding population of gannet are summarised in Table 4.20 and Table 4.21. The CRM predictions presented are the number of potential gannet collisions each year for different overall avoidance rates, for the most adverse (128 x 3.6 MW turbines) and least adverse (64 x 7MW turbines) wind farm designs evaluated. Results for the two other wind farm designs evaluated are presented in Appendix 12.2.

For the purposes of assessing collision mortality it is assumed, on the basis of evidence from existing offshore wind farms presented above, that 90% of flying gannets will be displaced outside the wind farm, and therefore the at-risk flight activity by gannets during the operational phase is assumed to be 10% of that recorded during the baseline surveys. This is considered to lead to strongly cautious assumptions regarding collision because the actual proportion displaced outside the wind farm is likely to exceed 90% and the few birds that do fly through the proposed wind farm are likely to lower their flight height to well below the rotor heights. This strongly cautious approach is merited because there is the possibility that during times of gales, fog and at night gannet avoidance behaviour (either far field adjustments to course or last moment evasion) could be less effective than is expected from existing studies.

COWRIE guidance (Maclean *et al.,* 2009) is to use a default avoidance rate of 99.5% for seabirds. There is no specific SNH guidance on avoidance rates for seabirds, and therefore their default value of 98.0% is applicable. However both these values are likely to underestimate the true avoidance rate due to the very high (>90%) far far-field avoidance recently published for this species (Leopold *et al*., 2011). In light of this, an overall avoidance rate of 99.8% is likely to be more appropriate and more closely reflect the true risks whilst still remaining cautious. An overall avoidance rate of 99.8% is ten times higher than 98.0% and is chosen to reflect the assumption that flight activity within the wind farm by gannets will on average be one tenth of that observed in the baseline surveys.

Table 4.20 Proportion of gannets estimated to be at rotor height and the number of collisions predicted by collision rate modelling using Band overall avoidance rates (OAR) of 98.0% and 99.5% for least adverse (64 x 7 MW turbines) and most adverse (128 x 3.6 MW turbines) wind farm designs evaluated. LAT is sea level at lowest astronomical tide, and this is approximately 2 m below the average sea level.

Metric	Design 1, 64 x 7MW			Design 4, 128 x 3.6 MW		
	98.0% OAR	99.5% OAR	98.8% OAR	98.0% OAR	99.5% OAR	98.8% OAR
Rotor swept height range (metres above LAT)	24 - 188			24 - 144		
Flight activity at rotor height	6.5%			6.5%		
Flight activity below rotor height (<22.5 m above sea)	93.5%			93.5%		
Collisions in breeding season (March to September), all ages	602.3	150.6	60.2	926.0	231.5	92.6
Collisions in non-breeding period (October to February), all ages	71.4	17.8	7.1	109.7	27.4	11.0
Total collisions per year, all ages	673.7	168.4	67.4	1035.7	258.9	103.6

The highest potential collision rates are for Wind Farm Design 4 (128 x 3.6 MW turbines) (Table 4.20 and Table 4.21). Under this design and for a 99.8% overall avoidance rate it is estimated that the average number of adult gannet collisions per year from the regional breeding population would be 95.5, with 90.2 collisions in the breeding season and 5.3 collisions in the non-breeding period. The collisions in the non-breeding period are attributed to birds overwintering from breeding sites outside the region. If the avoidance rate was 98.0% these figures would be ten times greater.

Table 4.21 The effect of CRM predicted collision mortality on the adult mortality rate (AMR) of the regional breeding gannet population. Results are presented for least adverse (64 x 7MW turbines) and most adverse (128 x 3.6MW turbines) wind farm designs evaluated, and for three values of overall avoidance rate (OAR).

Wind farm design / Period of year	**% birds present assumed to be from regional breeding popn**	**98.0% OAR**		**99.5% OAR**		**99.8% OAR**	
		CRM deaths	**Change AMR**	**CRM deaths**	**Change AMR**	**CRM deaths**	**Change AMR**
Design 1, 64 x 7MW turbines							
Breeding period (Mar - Sep)	100%	587	6.0%	147	1.5%	59	0.6%
Non-breeding period (Oct - Feb)	50%	35	0.4%	9	0.1%	3	0.04%
Whole year	varies	621	6.3%	155	1.6%	62	0.6%
Design 4, 128 x 3.6MW turbines							
Breeding period (Mar - Sep)	100%	902	9.2%	225	2.3%	90	0.9%
Non-breeding period (Oct - Feb)	50%	53	0.5%	13	0.1%	5	0.1%
Whole year	varies	955	9.8%	238	2.4%	95	0.97%

The baseline annual average mortality of adult birds in the regional breeding population is estimated at 9,789 birds based on an adult annual mortality rate of 8.4% (BTO, 2012). Therefore, an additional loss of 95.1 adults each year from the regional breeding population predicted for Wind farm Design 4 would represent an increase in the adult mortality rate of 0.97%. If a 98.0% avoidance rate is used the change in the adult mortality rate would be 9.7%.

The predicted number of collisions caused by Wind Farm Design 1 (the least adverse) is 62 collisions per year for a 99.8% avoidance rate, i.e., approximately two thirds of those predicted for Wind Farm Design 4 (Table 4.20 and Table 4.21). This number of collisions would cause the baseline adult mortality rate to increase by 0.6%.

Taking into consideration that 99.8% is likely to be much closer to the true avoidance rate than 98.0% yet still remain cautious, it is concluded that for Wind farm Design 4 (the most adverse design, 128 x 3.6MW) collision mortality of gannet is an effect of low magnitude, temporally long-term and reversible.

It is further concluded that for Wind farm Design 1 (the least adverse design, 64 x 7MW) collision mortality of gannet is an effect of negligible magnitude, temporally long-term and reversible.

The regional gannet population in the breeding season has shown a long term increase, averaging 4.0 % per annum between 1985 and the most recent count in 2009 (Murray 2011). Although there has been a reduction in the annual increase in numbers it is clear therefore that recruitment into the breeding population exceeds adult mortality. This is strong evidence that the population could sustain some additional mortality without affecting its viability. This view is supported by the results of recent population modelling studies that show that the Bass Rock gannet population would need additional adult mortality in the order of 2,000 birds per year to lead to a population decline (WWT, 2011).

It is concluded that for the illustrative wind farm designs examined here, the effects of collision mortality on gannets from the regional population in the breeding season are ***not significant*** under the Electricity Regulations.

It is apparent that the above conclusion of collision risk assessment for gannet is highly sensitive to the overall collision avoidance rate used in the predictive calculations. If lower avoidance rates are assumed (i.e., 98.0% or 99.5%) then there is an increase in the potential for a significant effect.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

Gannets are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional populations of gannets in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

4.4.7.5 EIA Decommissioning Phase assessment for gannet

The decommissioning phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

Gannets are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during decommissioning operations on the regional populations of gannets in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

4.4.7.6 EIA summary of effects combined for gannet

The adverse impacts of the effects assessed will act in a broadly additive manner. In combination it is judged that the magnitude of the three effects on the population is low. However, because the population has low sensitivity to all effects, it is judged that the overall impact on the population is ***not significant*** under the EIA regulations (Table 4.22 and Table 4.23).

Table 4.22 Summary of effects on the regional population of gannets in the breeding season

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	Not significant
Operational phase				
Displacement from foraging habitat	Low	Long term	Low	Not significant
Barrier Effect	Low	Long term	Low	Not significant
Vessel disturbance	Negligible	Long term	Low	Not significant
Collision mortality	Negligible or Low	Long term	Low	Not significant
All effects combined	Low	Long term	Low	Not significant

Table 4.23 Summary of effects on the regional population of gannets in the non-breeding season

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	Not significant
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Negligible	Not significant
Vessel disturbance	Negligible	Long term	Low	Not significant
Collision mortality	Negligible or Low	Long term	Low	Not significant
All effects combined	Low	Long term	Low	Not significant

4.4.7.7 Cumulative Impact Assessment for gannet

Combining the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts to the regional population of gannets in the breeding season will be as shown in Table 4.24. The individual impacts for displacement and collision are combined by addition to give the cumulative impact. The flight directions from the Bass Rock gannetry potentially affected by each wind farm acting as a barrier substantially overlap and therefore the cumulative impact from the three developments is not the sum of the individual predicted impacts. The cumulative impact is derived from the overall spread of flight directions potentially affected.

Table 4.24 Summary of CIA for the three proposed offshore wind farm in south-east Scotland on the regional population of gannet in the breeding season

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF +1km	Predicted impact from Inch Cape pOWF +1km	Predicted impact from Firth of Forth Round 3 Zone pOWF + 1km	Cumulative Impact
Displacement – breeding (effective loss of foraging habitat)	100%	≤ 1.3%, Low	≤1.4%, Low	≤5.2%, Moderate	≤7.9%, Moderate
Barrier Effect: % of flight directions from colonies blocked and increase in length of affected flights.	100%	Up to 27% of Bass Rock flights may increases in length by 2.2%	Up to 18% of Bass rock flights may increases in length by 2.9%	Up to 60%, of Bass rock flights may increases in length by 15.9%.	Up to 65%, of Bass rock flights may increases in length by approximately16%.
Collision most adverse designs: % increase in assumed annual adult mortality rate based on 98.0% avoidance rate.	0%	9.8% (955 adult deaths p.a.), Moderate	39.2% (4,004 adult deaths p.a.), High	23.6% (3,264 adult deaths p.a.), High	80.4% (8,224 adult deaths p.a.), Very High
Collision least adverse designs: % increase in assumed annual adult mortality rate based on 98.0% avoidance rate.	0%	6.5% (640 adult deaths p.a.), Moderate	3.6% (352 adult deaths p.a.), Low	20.4% (2,100 adult deaths p.a.), High	30.4% (3,040 adult deaths p.a.), High
Collision most adverse designs: % increase in assumed	0%	2.4% (239 adult deaths p.a.), Low	9.8% (1,001 adult deaths p.a.), Moderate	7.9% (816 adult deaths p.a.), Moderate	20.1% (2,056 adult deaths p.a.), High

annual adult mortality rate based on 99.5% avoidance rate.					
Collision least adverse designs: % increase in assumed annual adult mortality rate based on 99.5% avoidance rate.	0%	1.6% (160 adult deaths p.a.) , Low	0.9% (88 adult deaths p.a.) , Low	5.1% (525 adult deaths p.a.), Moderate	7.6% (760 adult deaths p.a.), Moderate
Collision most adverse designs: % increase in assumed annual adult mortality rate based on 99.8% avoidance rate.	0%	1.0% (96 adult deaths p.a.), Low	3.9% (400 adult deaths p.a.), Low	2.4% (326 adult deaths p.a.), Low	8.0% (822 adult deaths p.a.), Moderate
Collision least adverse designs: % increase in assumed annual adult mortality rate based on 99.8% avoidance rate.	0%	0.7% (64 adult deaths p.a.), Negligible	0.4% (352 adult deaths p.a.), Negligible	2.0% (2,100 adult deaths p.a.), Low	3.0% (3,040 adult deaths p.a.), Low

The cumulative impact of displacement from the three proposed wind farms evaluated is predicted to be a potential effective loss of up to 7.9% of the foraging habitat of the regional population in the breeding season. This cumulative effect is rated as moderate in magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). However, tagging of gannets from Bass Rock over a number of years show that their foraging behaviour is adaptable to changing food supplies, with birds extending their foraging range when feeding resources near the colony are poor (Hamer *et al*., 2011). This suggests that gannets from the Bass Rock have low sensitivity to small losses of feeding habitat. It is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional population of gannets in the breeding season is an effect of ***minor significance*** under the terms of the EIA Regulations.

The predicted cumulative effect of the three proposed wind farms could form a substantial barrier to the flights of gannets from the Bass Rock as it potentially would affect 65% of flight directions. The proportion of flights affected could be greater than this as there is evidence that north-easterly flight headings predominate over other possible directions (Hamer *et al.,* 2011). On average affected flights are estimated to increase in length by approximately 16% compared to the average direct flight to the same destination. The cumulative effect is rated as high in magnitude and temporally long-term and reversible

(Table 3.1 and Table 3.2). Studies on foraging gannets have shown that they are capable of extending foraging distances in response to distribution of prey, suggesting that the species has a high capacity to absorb the minor increases in flight distances that barriers could cause (Hamer *et al.,* 2007; Hamer *et al*., 2011). This species is also rated as low for sensitivity to barrier effects by Maclean *et al.* (2009) and Langston (2010). It is concluded that any adverse effects on the regional gannet population in the breeding season caused by the cumulative effect of wind farms acting as barrier to flying birds is of ***minor significance*** under the terms of the Electricity Act.

The cumulative assessment of barrier effects above assumes that all areas of the wind farm sites will be developed. In the case of the Firth of Forth Round 3 Zone there will be large areas without turbines and the eventual design may therefore effectively present gannets several smaller barriers with large gaps through which birds may pass, rather than the single large barrier assumed above. Were this to be the case, the magnitude of the cumulative barrier effect could be substantially less than suggested above.

Any conclusion on the cumulative impact of collision mortality on the regional gannet population in the breeding season is highly sensitive to the wind farm designs evaluated and the level of avoidance rate used for predictive calculations of the number of collision strikes.

Using an avoidance rate of 99.5% the cumulative impact of the three wind farms in terms of the predicted increase to the annual adult mortality rate is estimated at 7.6% (760 adult collisions p.a.) for the least adverse designs and 20.1% (2,056 adult collisions p.a.) for the most adverse designs. This cumulative impact is rated as high in magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). As discussed earlier, the regional gannet population has shown long term population growth, and modelling studies suggest it would take approximately 2,000 additional adult deaths per annum to cause the population to decline. Therefore this population appears to have low sensitivity to the effects of additional mortality caused by collision strikes. On this basis it is concluded that, using a 99.5% avoidance rate, the cumulative impact of collision mortality caused by the three proposed offshore wind farms and on the regional population of gannets in the breeding season is of ***minor significance*** under the terms of the EIA Regulations.

The equivalent mortality rate change values calculated using a 98.0% avoidance rate are 30.4% (3,046 adult collisions p.a.) for the least adverse designs and 80.4% (8,224 adult collisions p.a.) for the most adverse designs. This level of change to the mortality rate would be a cumulative impact rated as high in magnitude, temporally long-term and reversible. In the context of the population's status and low sensitivity to additional mortality, it is concluded that, using a 98.0% avoidance rate, the cumulative impact of collision deaths would be an effect of ***moderate significance*** under the terms of the EIA Regulations.

The equivalent mortality rate change values calculated using a 98.8% avoidance rate are 3.0% (304 adult collisions p.a.) for the least adverse designs and 8.0% (822 adult collisions p.a.) for the most adverse designs. This level of change to the mortality rate would be a cumulative impact rated as low in magnitude, temporally long-term and reversible. In the context of the population's status and low sensitivity to additional mortality, it is concluded that**,** using a 99.5% avoidance rate, the cumulative impact of collision deaths would be an effect of ***minor significance*** under the terms of the EIA Regulations.

The evidence of very high levels of far field avoidance of offshore wind farms by gannets suggests that a 98.8% avoidance rate is likely to most closely reflect the actual risks from collision for this species.

A separate CIA has not been undertaken for the regional gannet population in the non-breeding-period. This is because the cumulative assessment above for collision mortality on the regional breeding population includes collision deaths for the whole year and assumes that all the birds killed outside the breeding period are from the regional breeding population. The predicted displacement of gannets due to Neart na Gaoithe wind farm in the non-breeding period is negligible and the sensitivity of the population to displacement at this time of year is also negligible. Therefore it is not plausible that the Neart na Gaoithe Development could contribute to a significant cumulative impact for this receptor population.

4.4.7.8 Mitigation measures for gannet

The assessment does not identify any significant adverse effects on the regional populations of gannets in the breeding or non-breeding periods. Therefore no mitigation measures are required for this species.

4.4.8 Cormorant *Phalacrocorax carbo*

4.4.8.1 Status

Cormorants breed in colonies, and their distribution is closely linked to sheltered shallow coastal waters, usually less than 20 m deep, where foraging birds can reach the seabed. Cormorants typically prey on a wide range of small fish species, from shallow, inshore waters (Forrester *et al*., 2007). Seabird 2000 recorded 6,824 pairs of cormorants breeding around the coast of Britain, with a further 1,646 pairs breeding inland (Mitchell *et al*., 2004).

4.4.8.2 Neart na Gaoithe Study Area

The species was not recorded in the offshore site in Years 1 and 2. In the buffer area, one cormorant was recorded flying north, below 7.5 m in height, in March of Year 1, with three birds recorded in September of Year 2 (Table 4.2) (Figure 4.29).

It is likely that water depth in the Neart na Gaoithe Study Area is not optimal for foraging cormorants, hence the lack of records of this species.

10.3.1.1 Species sensitivity

A recent review assessed cormorant as being at moderate risk of collision, barrier effects and habitat loss or changes in prey distribution resulting from offshore wind farms. Risk of displacement was rated as low. Overall, cormorant was assessed as being at moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

Cormorant is listed as a qualifying interest species in the breeding season for two SPAs on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development (Table 4.25). These SPAs held 6.2% of the UK breeding population and 1.1% of the biogeographic population at the time of designation (JNCC, 2012). The distance between the offshore site and the Forth Islands SPA is less than the mean maximum foraging range of this species (25 km) (Thaxter *et al.*, 2012).

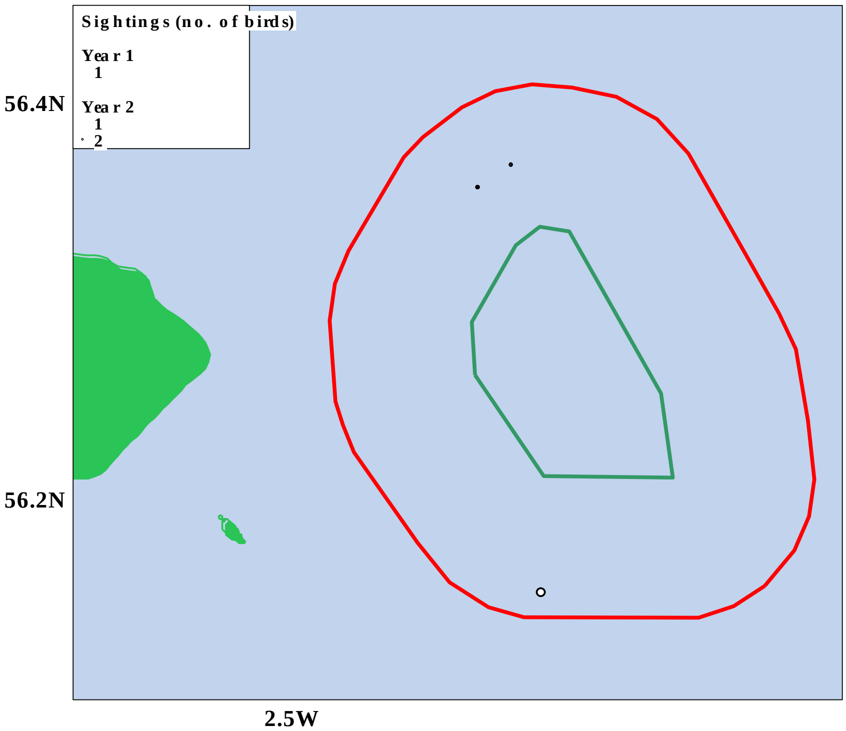


Figure 4.29 Cormorant sightings in Years 1 and 2

Table 4.25 SPAs for cormorants in the breeding season between Peterhead and Blyth

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count [2]	Year
Firth of Forth islands	**16**	**240**	**0.6**	**3.4**	**242**	**2007**
Farne Islands	72	194	0.5	2.8	141	2009
Total	**-**	**434**	**1.1**	**6.2**	**383**	**-**

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. Sites in italics lie within the maximum known foraging range of 35 km. Sites in bold lie within the mean maximum foraging range of 25 km (Thaxter *et al.*, 2012).

The species is also listed as a qualifying interest species in the non-breeding season for two SPAs on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development (Table 4.26). These SPAs held 7.0% of the UK non-breeding population, and 0.8% of the biogeographic population at the time of designation (JNCC, 2012). Recent five-year means are also shown for comparison, where available.

Table 4.26 SPAs for cormorants on the UK east coast between Peterhead and Blyth in the non-breeding season

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count
Firth of Forth	15	697	0.6	5.3	614 [3]
Firth of Tay & Eden Estuary	30	230	0.2	1.7	224 [2]
Total	-	**927**	**0.8**	**7.0**	**838**

1 Sources: 1 JNCC (2012) – SPA online species accounts. 2 Calbrade *et al*., 2010
3 Holt *et al*., 2011 x No data available

4.4.8.3 Assessment

The water depth in the Survey Area is approximately between 40 and 60 m (Harker & Buse 2009) and is therefore likely to be too deep to attract foraging birds from the Firth of Forth SPA population. The foraging ecology of this species together with the results from Years 1 and 2 baseline surveys indicate that cormorant is unlikely to be affected by the Development.

4.4.9 European Shag *Phalacrocorax aristotelis*

4.4.9.1 Status

Shags are surface-diving bottom-foraging birds that generally forage in inshore waters, at depths of between 20 and 40 m. The main prey species is the lesser sandeel, caught on or near the sea bed (Forrester *et al*., 2007). Seabird 2000 recorded 28,579 pairs of shags around the coast of Britain (Mitchell *et al*., 2004). The nearest large colony to the Neart na Gaoithe development is the Isle of May.

4.4.9.2 Neart na Gaoithe Study Area

There were no shags recorded in the offshore site in Years 1 and 2 (Table 4.2). Low numbers of shags (11 birds) were recorded in the Neart na Gaoithe buffer area in Year 1, with four birds in March, three in April and four in October (Figure 4.30). There were fewer shags recorded in the buffer area in Year 2, with 2 in February and four in March.

In Year 1, all shags were recorded in the western sector of the buffer area, which was thought likely to be related to water depth, as this area is generally shallower than further east in the study area. Shags were more scattered across the study area in Year 2.

In Years 1 and 2, a total of 16 shags were recorded in flight, with all birds recorded flying below 17.5 m.

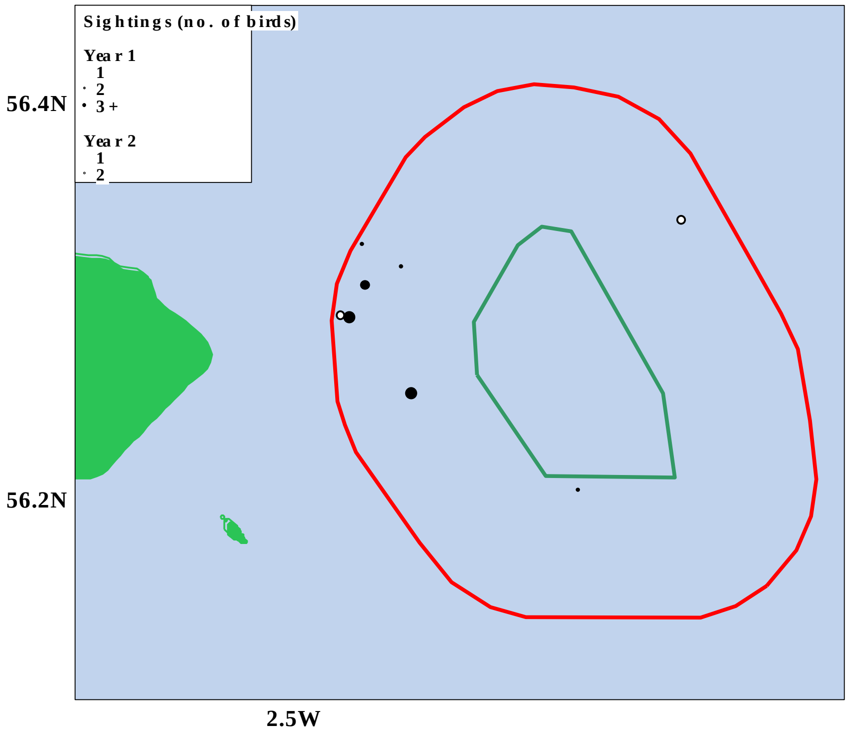


Figure 4.30 Shag sightings in Years 1 and 2

4.4.9.3 Species sensitivity

Shag is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009). A recent review assessed shag as being at moderate risk of displacement, barrier effects and habitat loss or changes in prey distribution resulting from offshore wind farms. Risk of collision was rated as low. Overall, shag was assessed as being at moderate risk from offshore wind developments Langston (2010) (Table 4.4).

Shag is listed as a qualifying interest species in the breeding season for four SPAs on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development (Table 4.27). These SPAs held 14.9% of the UK breeding population and 4.4% of the biogeographic population at the time of designation (JNCC, 2012). The distance between the offshore site and the Forth Islands SPA is less than the maximum foraging range of this species (17 km) but outside the mean maximum foraging range (14.5 km) (Thaxter *et al.*, 2012).

Table 4.27 SPAs for shags in the breeding season between Peterhead and Blyth

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count [2]	Year
Buchan Ness to Collieston Coast	113	1,045	0.8	2.8	344	2007
Farne Islands	72	994	0.8	2.7	838	2009
Firth of Forth Islands	*16*	*2,887*	*2.3*	*7.7*	*934*	*2009*
St Abb's Head to Fast Castle	31	651	0.5	1.7	131	2008
Total	-	**5,577**	**4.4**	**14.9**	**2,247**	-

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. Sites in italics lie within the maximum known foraging range of 17 km. Sites in bold lie within the mean maximum foraging range of 14.5 km (Birdlife International 2012).

4.4.9.4 Assessment

Shags generally forage in water depths between 20 and 40 m and therefore only the shallowest areas in the study area are likely to attract foraging birds. The distribution of birds recorded during Year 1 baseline surveys corroborates this, with all shags recorded in the shallower western sector of the buffer area, although birds were more scattered across the Survey Area in Year 2. The foraging ecology of this species together with the results from baseline surveys indicate that shag is unlikely to be affected by the Development.

4.4.10 Common Eider *Somateria mollissima*

4.4.10.1 Status

The eider is the commonest species of seaduck in the UK, with a mainly sedentary breeding population of around 31,650 pairs (RSPB 2012). Large numbers breed on islands in the Firth of Forth in summer, while Tayport, near St Andrews is important for wintering birds, with a 5-year mean between 2004/05 and 2008/09 of 11,500 birds (Calbrade *et al*., 2010).

Eiders are an inshore species, generally found within 10 km of the coast. The main prey species are blue mussels, as well as sea urchins, starfish and other marine invertebrates (Forrester *et al*., 2007).

4.4.10.2 Neart na Gaoithe Study Area

Low numbers of eider were recorded in the Neart na Gaoithe study area on surveys in Year 1, with nine birds recorded in the offshore site, and 11 in the buffer area between November and December (Table 4.2) (Figure 4.31). Just two eider were recorded in Year 2, in the buffer area in January.

At around 50 m water depth, the Neart na Gaoithe offshore site is likely to be too deep for eider to feed profitably. This is supported by survey results from Years 1 and 2, as well as published information on the species distribution in the region (Barton and Pollock 2004, Söhle *et al*., 2007, Stone *et al*., 1995).

All 22 eider were recorded flying below 17.5 m in height, with 12 birds below 7.5 m, eight between 7.5 and 12.5 m and two birds flying between 12.5 m and 17.5 m (Table 4.3). The majority of birds were flying west or north-west (95.0%).

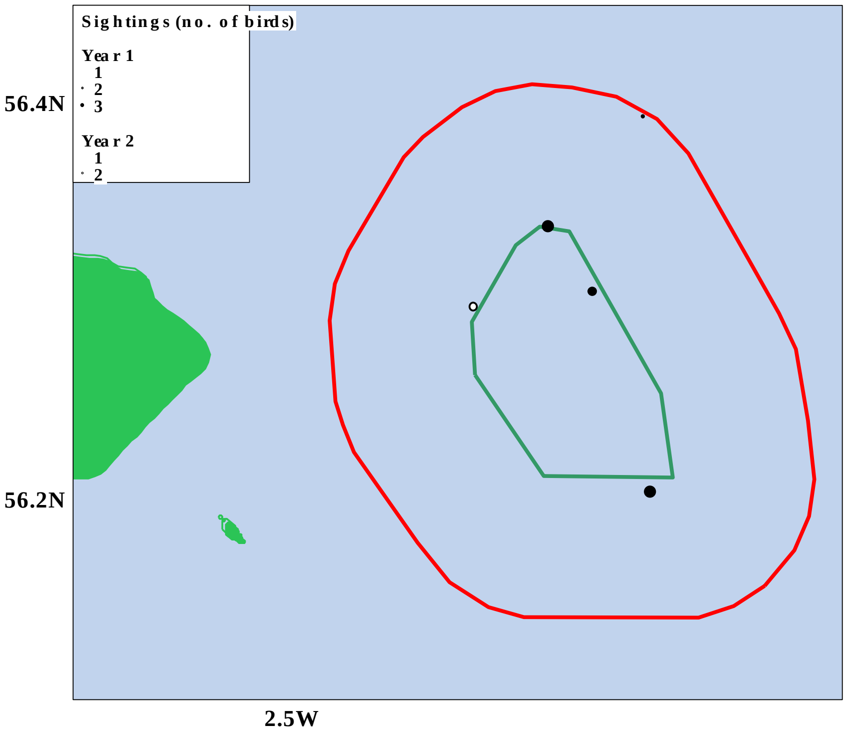


Figure 4.31 Eider sightings in Years 1 and 2

4.4.10.3 Species sensitivity

Eider is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009). A recent review assessed eider as being at moderate risk of barrier effects and habitat loss or changes in prey distribution resulting from offshore wind farms. Risk of collision and displacement were ranked as low. Overall, eider was assessed as being at moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

Eider is listed as a qualifying interest species in the non-breeding season for five SPAs on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development (Table 4.28). These SPAs held 19.5% of the UK non-breeding population, and 0.9% of the biogeographic population at the time of designation (JNCC, 2012). Recent five-year means are also shown for comparison, where available.

Table 4.28 SPAs for eider on the UK east coast between Peterhead and Blyth in the non-breeding season

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count
Firth of Forth	15	7,887	0.5	10.2	5,297 [3]
Firth of Tay and Eden Estuary	30	2,061	0.1	2.7	7,453 [3]
Lindisfarne	53	1,568	0.1	2	930 [3]
Montrose Basin	44	1,794	0.1	2.3	2,376 [3]
Ythan Estuary, Sands of Forvie and Meikle Loch	110	1,778	0.1	2.3	3,404 [3]
Total	-	**15,088**	**0.9**	**19.5**	**19,460**

1 Sources: 1 JNCC (2012) – SPA online species accounts. 2 Calbrade *et al.*, 2010
3 Holt *et al.*, 2011

4.4.10.4 Assessment

The foraging ecology and habitat preference of this species together with the low numbers of eider recorded in the offshore site during baseline surveys indicate that this species is unlikely to be affected by the Development.

4.4.11 Common scoter *Melanitta nigra*

4.4.11.1 Status

Common scoter typically winter on shallow inshore waters less than 20 m deep and generally between ca. 500 m and 2 km from shore (Birdlife International, 2010). Most of the UK winter population of common scoter tends to be found in a few large flocks off the mouths of major estuaries around the coast of Britain. A review of numbers for the UK and recent survey work at key sites suggests that the number of wintering common scoter is likely to be in the region of 50,000 birds (Kershaw & Cranswick 2003). The UK breeding population of common scoter has declined by more than 50% in recent years, and was estimated at between 9 and 52 pairs in 2007, all in northern Scotland (Holling *et al.*, 2010).

4.4.11.2 Neart na Gaoithe Study Area

Five common scoter were recorded on surveys in Year 1, in the centre of the Neart na Gaoithe offshore site in October (Table 4.2) (Figure 4.32). All five birds were recorded flying north-west, below 7.5 m in height. Just two common scoter were recorded in Year 2, in the buffer area in September.

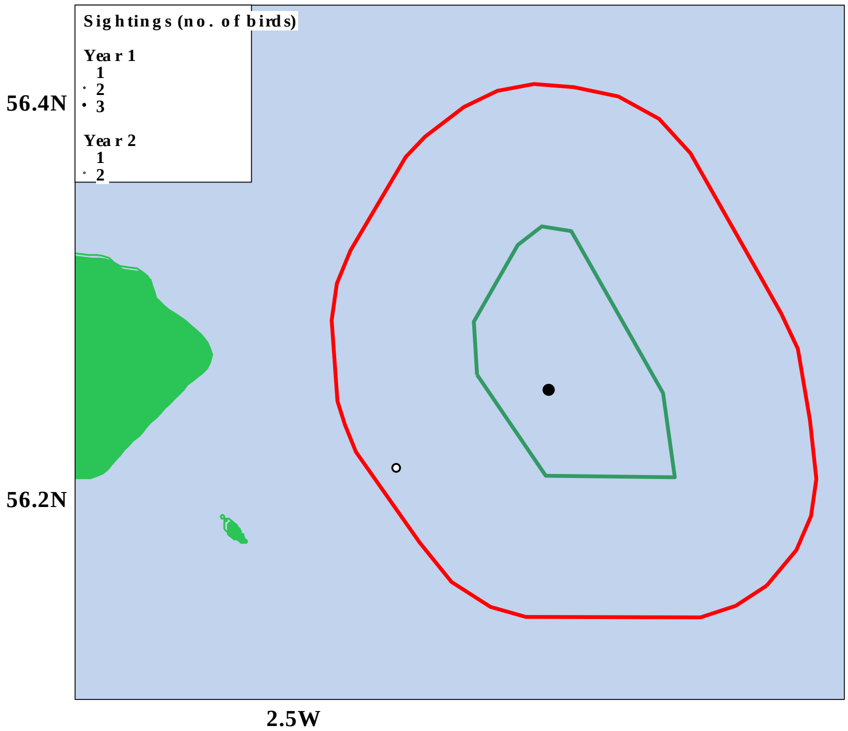


Figure 4.32 Common scoter sightings in Years 1 and 2

4.4.11.3 Species sensitivity

Common scoter is currently red-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009).

A recent review assessed common scoter as being at moderate risk of displacement, barrier effects and habitat loss or changes in prey distribution resulting from offshore wind farms. Risk of collision was ranked as low. Overall, common scoter was assessed as being at moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

Common scoter is listed as a qualifying interest species in the non-breeding season for three SPAs on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development (Table 4.29). These SPAs held 17.3% of the UK non-breeding population, and > 0.3% of the biogeographic population at the time of designation (JNCC, 2012). Recent five-year means are also shown for comparison, where available.

Table 4.29 SPAs for common scoter on the UK east coast between Peterhead and Blyth in the non-breeding season

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count [2]
Firth of Forth	15	2,653	0.2	9.6	1,393 [2]
Firth of Tay & Eden Estuary	30	1,444	0.1	5.3	632 [2]
Lindisfarne	53	654	<0.1	2.4	257 [2]
Total	-	**4,751**	**> 0.3**	**17.3**	**2,282**

1 Sources: 1 JNCC (2012) – SPA online species accounts. 2 Calbrade *et al*., 2010
3 Holt *et al*., 2011

4.4.11.4 Assessment

The non-breeding habitat preference of this species together with the low numbers recorded in the offshore site on baseline surveys indicate that common scoter is unlikely to be affected by the Development.

4.4.12 Grey phalarope *Phalaropus fulicarius*

4.4.12.1 Status

Grey phalarope is an uncommon autumn passage migrant in Scotland (Forrester *et al*., 2007). Off the Fife coast, it is described as a very scarce winter visitor (Dickson 2002). Grey phalaropes do not breed in Britain.

4.4.12.2 Neart na Gaoithe Study Area

One grey phalarope was recorded on surveys in Year 1, on the water in the offshore site in November (Table 4.2) (Figure 4.33). In Year 2, two grey phalaropes were recorded on surveys, in the buffer area in October.

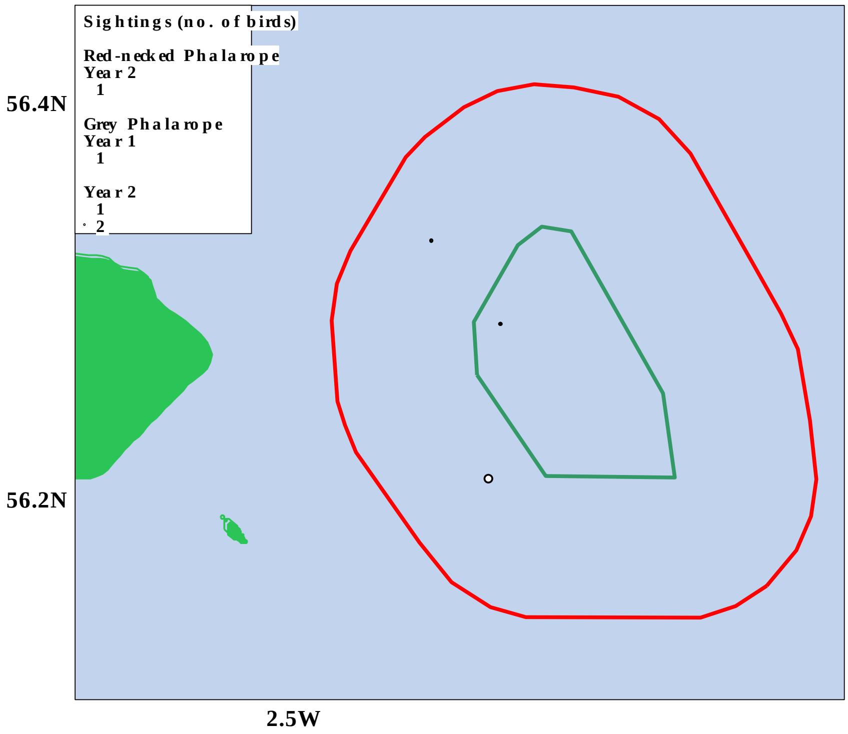


Figure 4.33 Red-necked and grey phalarope sightings in Years 1 and 2

4.4.12.3 Assessment

The small numbers of birds recorded in the offshore site on baseline surveys indicate that grey phalarope is unlikely to be affected by the Development.

4.4.13 Red-necked Phalarope *Phalaropus lobatus*

4.4.13.1 Status

Red-necked phalarope is a rare breeding bird in the UK, with approximately 17 pairs recorded in the Outer Hebrides and Shetland in 2008 (Holling *et al.*, 2010). Away from the breeding grounds, there are only a few Scottish records per year, with most sightings between May and September (Forrester *et al.*, 2007). Off the Angus and Dundee coast, it is described as a rare passage migrant (Carmichael 2002). Red-necked phalarope is listed on Annex I of the EU Birds Directive (2009/147/EEC), and the species is currently red-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009).

4.4.13.2 Neart na Gaoithe Study Area

One red-necked phalarope was recorded on surveys in Year 2, in the buffer area in September (Table 4.2) (Figure 4.33).

4.4.13.3 Assessment

The very low number of birds recorded on baseline surveys in Year 2 indicate that red-necked phalarope is unlikely to be affected by the Development.

4.4.14 Pomarine skua *Stercorarius pomarinus*

4.4.14.1 Status

Pomarine skua is a regular, but uncommon passage migrant past Scottish coasts, in spring and autumn. Numbers recorded vary between years (Forrester *et al*., 2007). Off the Fife coast, it is described as a very scarce spring migrant, and is uncommon in autumn (Dickson 2002). Pomarine skuas do not breed in Britain.

4.4.14.2 Neart na Gaoithe Study Area

In Year 1, six pomarine skuas were recorded on surveys during October in the west and south-west of the Neart na Gaoithe buffer area (Table 4.2) (Figure 4.34). No pomarine skuas were recorded on surveys in Year 2. The species was not recorded in the offshore site on baseline surveys.

4.4.14.3 Assessment

As pomarine skuas were not recorded in the offshore site on baseline surveys, and only low number of birds were recorded in the buffer area on Year 1 surveys, with no birds seen in Year 2, it is considered that pomarine skua is unlikely to be affected by the Development.

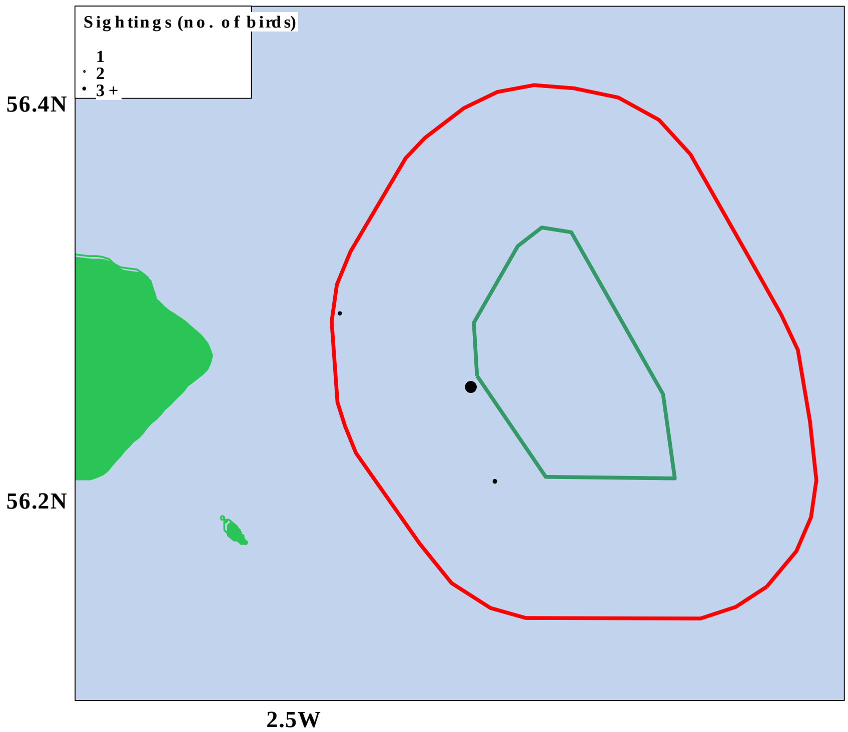


Figure 4.34 Pomarine skua sightings in Year 1

4.4.15 Arctic skua *Stercorarius parasiticus*

4.4.15.1 Status

Arctic Skua is a coastal passage migrant in both spring and autumn in Scotland, as well as a scarce breeding species, restricted to Shetland, Orkney, north Scotland and the Western Isles (Forrester *et al.*, 2007). Seabird 2000 recorded a breeding population of 2,136 pairs in Scotland (Mitchell *et al.*, 2004).

4.4.15.2 Neart na Gaoithe Study Area

A total of six Arctic skuas were recorded on surveys in Year 1, in the buffer area, between July and October 2010, with a peak of four in September (Table 4.2) (Figure 4.35). In Year 2, a total of 18 Arctic skuas were recorded in the buffer area between August and October, with a peak of 11 in October. Birds were scattered widely across the buffer area in both years. No Arctic skuas were recorded in the offshore site in Years 1 and 2.

A total of 20 Arctic skuas were recorded in flight in Years 1 and 2, with the majority of birds (95.0%) flying below 22.5 m (Table 4.3). One bird was recorded flying above 22.5 m, i.e. within the rotor-swept zone, at an estimated height of 30 m.

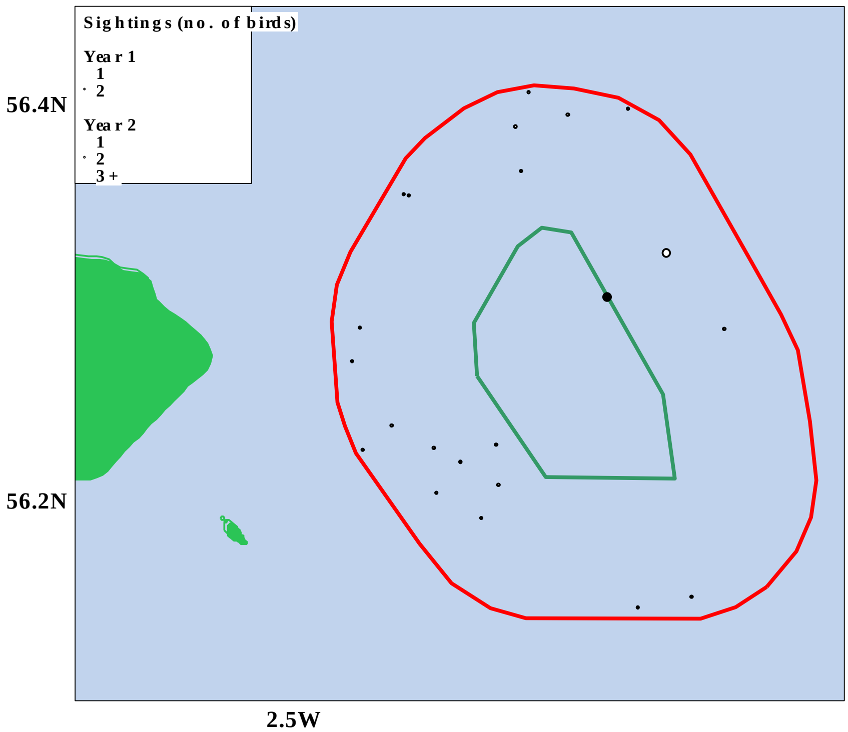


Figure 4.35 Arctic skua sightings in Years 1 and 2

4.4.15.3 Species sensitivity

Arctic skua is currently red-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009).

A recent review assessed Arctic skua as being at low risk of displacement, barrier effects and habitat loss or changes in prey distribution resulting from offshore wind farms. Risk of collision was ranked as moderate. Overall, Arctic skua was assessed as being at moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

Arctic skua is listed as a qualifying interest species in the breeding season for seven SPAs on Shetland and Orkney that could potentially be affected by the Neart na Gaoithe development (Table 4.30). These SPAs held 24.3% of the UK breeding population and 2.7% of the biogeographic population at the time of designation (JNCC, 2012). The distance between the offshore site and these SPA colonies is considerably greater than the maximum known foraging range of Arctic skua (75 km) (Thaxter *et al.*, 2012).

Table 4.30 SPAs for Arctic skuas in the breeding season on Shetland and Orkney

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count [2]	Year
Fair Isle	356	74	0.3	2.3	65	2009
Fetlar	477	130	0.4	4.1	96	2001
Foula	424	125	0.4	3.9	63	2009
Hoy	301	59	0.2	1.8	70	2000
Papa Westray (North Hill & Holm)	352	135	0.5	4.2	66	2009
Rousay	337	180	0.6	5.6	133	2000
West Westray	342	77	0.3	2.4	88	2000
Total	-	**780**	**2.7**	**24.3**	**581**	-

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. Sites in italics lie within the maximum known foraging range of 75 km. Sites in bold lie within the mean maximum foraging range of 62.5 km (Birdlife International 2012).

4.4.15.4 Assessment

The majority of Arctic skuas recorded during baseline surveys are likely to be migrating birds, probably from breeding sites in northern Scotland, Orkney and Shetland. Given the relatively small size of these breeding populations the potential for large numbers to migrate through the offshore site is probably small (Wernham *et al.*, 2002). This is corroborated by the small number of birds recorded in Years 1 and 2.

Populations

There is no published estimate of the size of the regional autumn passage population of Arctic skua. The EIA assessment was therefore based on the number of breeding Arctic skuas in Scotland; 2,136 pairs or 4,272 breeding adults based on the Seabird 2000 census (Mitchell *et al*., 2004).

Displacement

It is not known whether Arctic skuas will be displaced by the proposed Neart na Gaoithe development. The review undertaken by Langston (2010) suggested that Arctic skuas are at low risk of displacement effects. Arctic skuas were only recorded on baseline surveys between July and October, with the majority of birds seen during autumn passage. No Artic skuas were recorded within the offshore site during the baseline surveys. The species is highly mobile and pelagic in nature and therefore will be able to relocate elsewhere should displacement effects occur. Overall, it is concluded that the effects of displacement on Arctic skuas is ***not significant*** under the terms of the EIA Regulations.

Barrier effect

The nearest breeding colonies are on Shetland and Orkney and are beyond the maximum recorded foraging ranges for breeding birds (Thaxter *et al*., 2012). Therefore no barrier effects on breeding birds are predicted to occur during the breeding period. Outwith the breeding season, Arctic skuas undertake migrations of many thousands of kilometres to their wintering grounds off Australia, South Africa and southern South America (Wernham *et al*., 2002). Therefore, the potential incremental increase in distance an Arctic skua may fly should it fly around the wind farm will be negligible compared to the overall distance flown during migration. Overall, it is concluded that the effects of barrier effect on Arctic skuas is ***not significant*** under the terms of the EIA Regulations.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

Arctic skuas are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement, and that no Arctic skuas were recorded in the offshore site on baseline surveys it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional autumn passage populations of Arctic skuas is ***not significant*** under the EIA Regulations.

Collision Mortality

Collision risk modelling was undertaken for Arctic skua based on an assumed population of 1,000 birds passing through the offshore site per year, with 4% of birds at rotor height. This resulted in a total of 0.15 collisions predicted per year, based on an avoidance rate of 98.0%, or 0.04 collisions predicted per year, based on an avoidance rate of 99.5%. Further details are presented in Appendix 12.2.

Based on a baseline annual adult mortality rate of 11.4% (BTO, 2012), the additional collision mortality would lead to an increase of 0.1% in the annual adult mortality rate (i.e., to 11.5% p.a.) for collisions predicted using a 98.0% avoidance rate, and to an increase of 0.03% for collisions predicted for a 99.5% avoidance rate. In both case the effect is of negligible magnitude.

Scaling this up to the size of the Scottish Arctic skua population of 4,272 breeding adults (Mitchell *et al.,* 2004) that might potentially pass through the turbines would give a worst case scenario of 10 and 1 collisions per annum for 98.0% and 99.5% avoidance rates respectively. The effect of this on the population's adult mortality rate remains the same as shown above.

This worst case scenario assumes that all birds in the Scottish breeding population pass through the wind farm at rotor height twice a year, which is extremely unrealistic for two reasons. First, some Arctic skuas will leave the breeding colonies and migrate south down the west coast of Scotland, and therefore will not pass through the offshore site. Second, Arctic skuas migrate on a relatively broad front that is wider than the offshore site. Therefore, only a relatively small proportion (say, <25%) of the birds migrating down the east coast of Scotland would be expected to pass through the offshore site. However, as no Arctic skuas were recorded in the offshore site on baseline surveys it is suggested that 25% would still be a considerable overestimate. It is likely therefore that the actual effects of collision mortality on migrating Arctic skuas is considerably lower than the worst case scenario figures presented above.

Overall, the potential effect of the collision mortality of Arctic skuas on the baseline mortality rate is rated as negligible in magnitude, temporally long-term and reversible. It is concluded that the effects of collision mortality on Arctic skuas is therefore ***not significant*** under the terms of the EIA Regulations.

4.4.16 Great skua Stercorarius skua

4.4.16.1 Status

In addition to occurring in coastal waters on spring and autumn passage, great skuas are a localised breeding species in Britain, restricted to Shetland, Orkney and the Western Isles. Small numbers occur in Scottish waters in winter months (Forrester *et al*., 2007). Great Skuas breed close to other seabird colonies, in order to scavenge and parasitise food from other seabirds, as well as predating other birds and nests. Seabird 2000 recorded 9,634 pairs in Scotland (Mitchell *et al*., 2004).

4.4.16.2 Neart na Gaoithe Study Area

In Year 1, 24 great skuas were recorded on surveys in the Neart na Gaoithe study area, with one bird in the offshore site, and 23 birds scattered widely across the buffer area. One bird was seen in spring, and 26 were in the autumn and early winter months. Peak numbers were recorded in September (16 birds) (Table 4.2). Fewer great skuas were seen on surveys in Year 2, with 16 birds recorded in the buffer area. Most birds were recorded between August and October, with a peak of 9 birds in October (Figure 4.36).

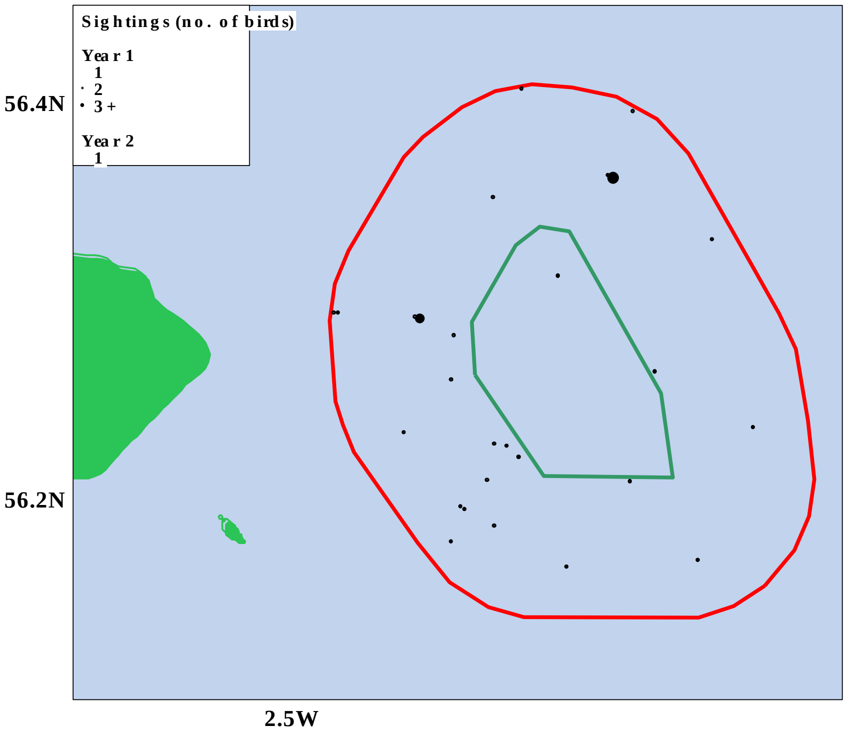


Figure 4.36 Great skua sightings in Years 1 and 2

A total of 36 great skuas were recorded in flight in Years 1 and 2, with the majority (97.2%) flying below 22.5 m in height (Table 4.3). One bird was recorded flying above 22.5 m, i.e. within the rotor swept zone, at an estimated height of 25 m.

Species sensitivity

Great skua is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009).

A recent review assessed great skua (and other skua species) as being at moderate risk of collision with turbines. Risk of displacement, barrier effects, habitat loss or changes in prey distribution resulting from offshore wind farms were ranked as low. Overall, great skua was assessed as being at high risk from offshore wind developments, when the importance of the UK breeding population was taken into account (Langston 2010) (Table 4.4).

Great skua is listed as a qualifying interest species in the breeding season for seven SPAs on Shetland and Orkney that could potentially be affected by the Neart na Gaoithe development (Table 4.31). These SPAs held 69.1% of the UK breeding population and 43.4% of the biogeographic population at the time of designation (JNCC, 2012). The distance between the offshore site and these SPA colonies is greater the maximum known foraging range of great skua (219 km) (Thaxter *et al.*, 2012).

Table 4.31 SPAs for great skuas in the breeding season on Shetland and Orkney

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count [2]	Year
Fair Isle	356	130	1.0	1.5	277	2009
Fetlar	477	512	3.8	6.0	593	2001
Foula	424	2,170	16.0	25.5	2,293	2000
Hermaness, Saxa Vord & Valla Field	510	630	4.6	7.4	572	2002
Hoy	301	1,900	14.0	22.4	42	2007
Noss	428	410	3.0	4.8	365	2007
Ronas Hill - North Roe & Tingon	474	130	1.0	1.5	x	
Total	-	**5,882**	**43.4**	**69.1**	**4,142+**	-

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. Sites in italics lie within the maximum known foraging range of 219 km. Sites in bold lie within the mean maximum foraging range of 86.4 km (Thaxter *et al.*, 2012).

4.4.16.3 Assessment

Most if not all great skuas recorded during baseline surveys are likely to be migrating birds, probably from breeding sites in northern Scotland, Orkney and Shetland. Given the relatively small size of these breeding populations the potential for large numbers to migrate through the offshore site is probably small (Wernham *et al.*, 2002). This is corroborated by the small number of birds recorded in Years 1 and 2.

Populations

There is no published estimate of the size of the regional autumn passage population of great skua. The EIA assessment was therefore based on the number of breeding great skuas in Scotland; 9,634 pairs or 19,268 breeding adults based on the Seabird 2000 census (Mitchell *et al.*, 2004).

Displacement

It is not known whether great skuas will be displaced by the proposed Neart na Gaoithe development. The review undertaken by Langston (2010) suggested that great skuas are at low risk of displacement effects. The majority of Great skuas were seen during autumn passage. The species is highly mobile and pelagic in nature and therefore will be able to relocate elsewhere should displacement effects occur. Overall, it is concluded that the effects of displacement on great skuas is ***not significant*** under the terms of the EIA Regulations.

Barrier effect

The nearest breeding colonies are on Shetland and Orkney and are beyond the maximum recorded foraging ranges for breeding birds (Thaxter *et al.*, 2012). Therefore no barrier effects on breeding birds are predicted to occur during the breeding period. Outwith the breeding season, great skuas undertake migrations of many hundreds of kilometres to their wintering grounds off southern Europe (Wernham *et al.*, 2002). Therefore, the potential incremental increase in distance a great skua may fly should it fly around the wind farm will be negligible compared to the overall distance flown during migration. Overall, it is concluded that the effects of barrier effect on great skuas is ***not significant*** under the terms of the EIA Regulations.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

Great skuas are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement, and the low numbers of great skuas in the offshore site recorded on baseline surveys it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional autumn passage populations of great skuas is ***not significant*** under the EIA Regulations.

Collision Mortality

Collision risk modelling was undertaken for great skua based on an assumed population of 1,000 birds passing through the offshore site per year, with 4% of birds at rotor height (Cook, *et al.,* 2012). This resulted in a total of 0.18 collisions predicted per year, based on an avoidance rate of 98.0%, or 0.04 collisions predicted per year, based on an avoidance rate of 99.5%. Further details are presented in Appendix 12.2.

Based on a baseline annual adult mortality rate of 11.2% (BTO, 2012), the additional collision mortality would lead to an increase of 0.2% in the annual adult mortality rate (i.e., to 11.4% p.a.) for collisions predicted using a 98.0% avoidance rate, and to an increase of 0.04% for collisions predicted for a 99.5% avoidance rate. In both case the effect is of negligible magnitude.

Scaling this up to the size of the Scottish great skua population of 19,268 breeding adults (Mitchell *et al.,* 2004) that might potentially pass through the turbines would give a worst case scenario of 39 and 8 collisions per annum for 98.0% and 99.5% avoidance rates respectively. The effect of this on the population's adult mortality rate remains the same as shown above.

This worst case scenario assumes that all birds in the Scottish breeding population pass through the wind farm at rotor height twice a year, which is extremely unrealistic for two reasons. First, some great skuas will leave the breeding colonies and migrate south down the west coast of Scotland, and therefore will not pass through the offshore site.. Second, great skuas migrate on a relatively broad front that is wider than the offshore site. Therefore, only a relatively small proportion (say, <25%) of the population would be expected to pass through the offshore site, although numbers recorded in the offshore site on baseline surveys suggest that 25% would still be a considerable overestimate. It is likely therefore that the actual effects of collision mortality on migrating great skuas is considerably lower than the worst case scenario figures presented above.

Overall, the potential effect of the collision mortality of great skuas on the baseline mortality rate is rated as negligible in magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of collision mortality on great skuas is therefore ***not significant*** under the terms of the EIA Regulations.

4.4.17 Little gull *Larus minutus*

4.4.17.1 Status

Little gull occurs on passage in Scottish waters, in spring, and more commonly in autumn (Forrester *et al*., 2007). Off the Fife coast, it is considered a passage migrant, mainly in autumn, with small numbers also occurring in winter (Dickson 2002). There have been one, possible two breeding records of little gulls in Scotland in 1988 and 1991, and five unsuccessful breeding attempts in England up to 2007 (Holling *et al*., 2010). Little gulls feed in flight or on the water, by pecking at the water surface for small items of food, often with other species such as kittiwakes.

4.4.17.2 Neart na Gaoithe Study Area

In Year 1, 298 little gulls were recorded on surveys in the Neart na Gaoithe study area, with the majority of birds (89.3%) recorded in the buffer area, mainly in October (Table 4.2). Numbers recorded on surveys in Year 2 were slightly lower (220 birds), although the majority of birds (97.3%) were again recorded in the buffer area in September and October.

In Year 1, peak numbers of little gulls occurred in October, with an estimated 314 birds in the offshore site, and 1,467 birds in the buffer area (Table 4.32) (Figure 4.37). In Year 2, estimated numbers of little gulls in the offshore site were lower, with a peak of 41 birds in September. Estimated numbers in the buffer area in Year 2 were highest in October, with a peak of 2,923 birds. Outside of September and October, estimated numbers of little gulls in the Neart na Gaoithe study area were low.

Overall, estimated numbers of little gulls in the Neart na Gaoithe buffer area exceeded the 1% threshold for internationally important numbers of little gulls (1,230 birds – Holt *et al*., 2011) in October of both Years 1 and 2 (Figure 4.37).

Table 4.32 Estimated numbers of little gulls in the offshore site and 8 km buffer in Years 1 and 2

Month	Estimated nos on water Offshore Site	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying Offshore Site	Estimated total Offshore Site	Estimated total 8 km buffer [1]	Estimated total
Yr1 Nov	0	0	0	0	0	61	61
Yr1 Dec	0	0	0	0	0	0	0
Yr1 Jan	0	0	0	0	0	0	0
Yr1 Feb	0	0	0	0	0	0	0
Yr1 Mar	0	0	0	0	0	0	0
Yr1 Apr	0	0	0	0	0	0	0
Yr1 May	0	0	0	0	0	0	0
Yr1 Jun	0	0	0	0	0	0	0
Yr1 Jul	0	0	0	0	0	0	0
Yr1 Aug	10	4	30	0	10	62	72
Yr1 Sep	0	0	0	0	0	149	149
Yr1 Oct	307	149	633	7	314	1,467	1,781
Yr2 Nov							
Yr2 Dec	0	0	0	0	0	0	0
Yr2 Jan	0	0	0	0	0	0	0
Yr2 Feb	0	0	0	0	0	0	0
Yr2 Mar	0	0	0	0	0	0	0
Yr2 Apr	0	0	0	0	0	0	0
Yr2 May	0	0	0	0	0	0	0
Yr2 Jun	0	0	0	0	0	0	0
Yr2 Jul	0	0	0	0	0	0	0
Yr2 Aug	0	0	0	0	0	0	0
Yr2 Sep	0	0	0	41	41	642	683
Yr2 Oct	0	0	0	0	0	2,923	2,923

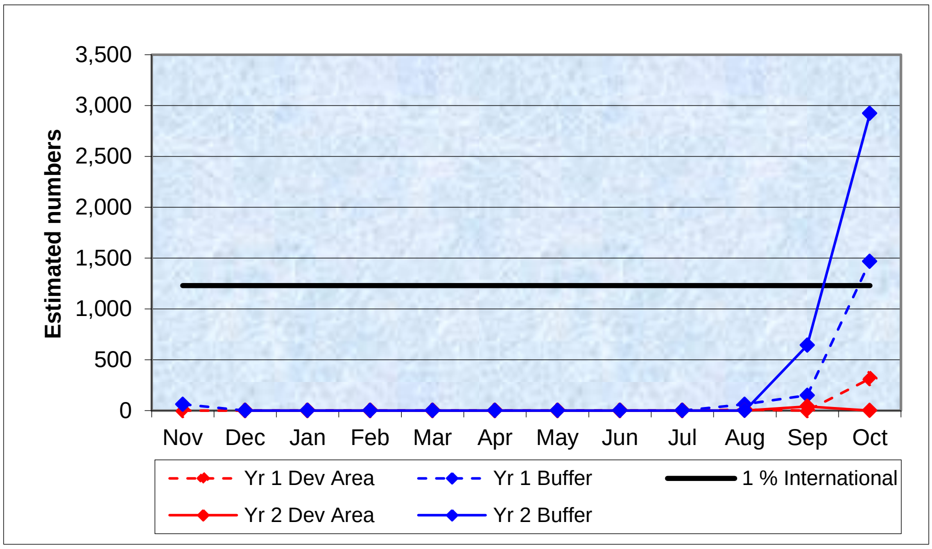


Figure 4.37 Monthly estimated numbers of little gulls in the Neart na Gaoithe Development & buffer areas in Years 1 and 2

Mean monthly little gull density was similar between the offshore site and the buffer area in Years 1 and 2, with very low densities recorded for most of the year, and a peak in October, although low densities were recorded in the offshore site in Year 2 in October (Figure 4.38). Mean densities of little gulls from ESAS data in the surrounding ICES rectangles and Regional Sea 1 were low throughout the year, with a slight increase in September.

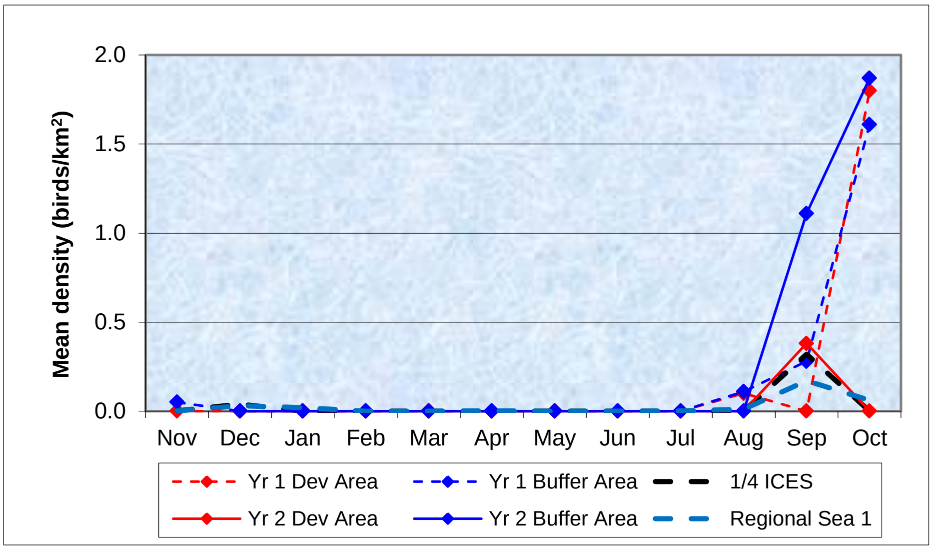


Figure 4.38 Comparison of monthly mean densities for little gull in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

In Year 1, highest densities of little gulls were recorded between August and October, in the north-west corner of the Neart na Gaoithe study area, with fewer birds recorded to the south of the offshore site (Figure 4.39).

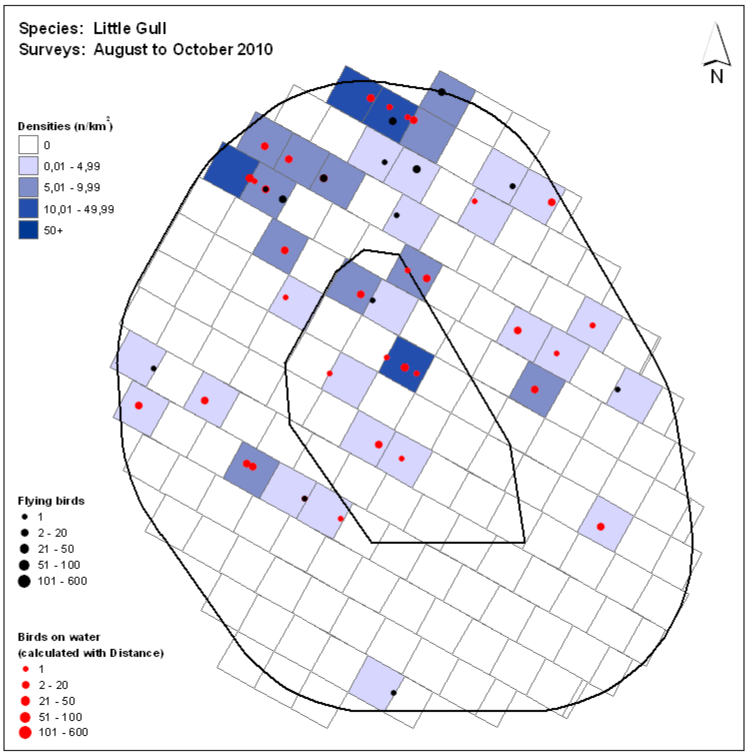


Figure 4.39 Little gull density between August and October, Year 1

A similar distribution pattern was recorded in Year 2 between August and October, with highest densities of little gulls again recorded in the north-west corner of the buffer area (Figure 4.40). Fewer little gulls were recorded in the offshore site between August and October than in Year 1.

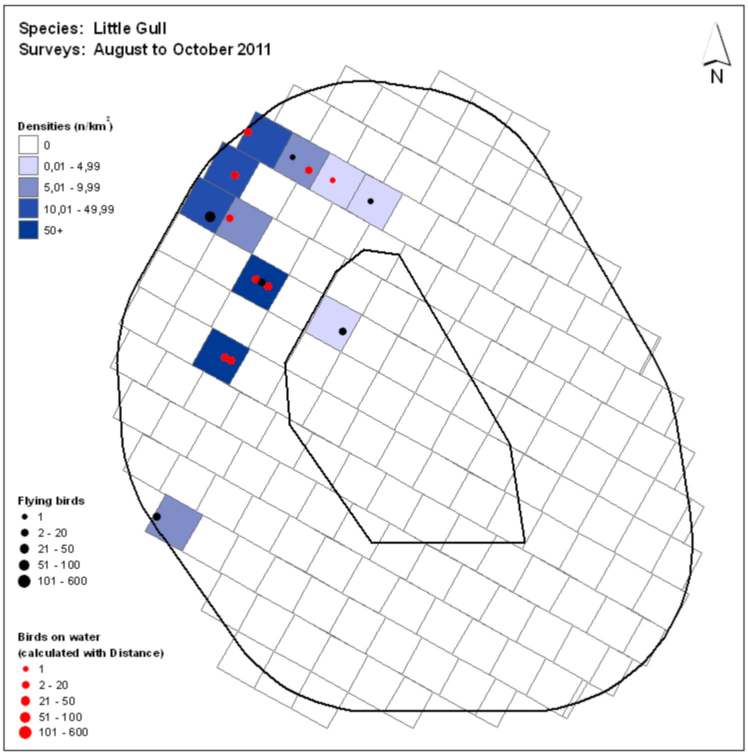


Figure 4.40 Little gull density between August and October, Year 2

In Years 1 and 2, 271 little gulls were recorded in flight, with 69.4% of all birds flying below 22.5 m in height (Table 4.3). A total of 83 birds (30.6%) were recorded flying above 22.5 m, i.e. within the rotor swept zone, at estimated heights of between 25 and 30 m.

Foraging behaviour was recorded for 84 little gulls in the Neart na Gaoithe study area in Years 1 and 2, with three types of foraging behaviour recorded (Table 4.33). Actively searching was the most frequently recorded foraging behaviour (64.3%).

Table 4.33 Little gull foraging behaviour in the Neart na Gaoithe Study Area in Years 1 and 2

Behaviour	Number of birds
Actively searching	54
Dipping	29
Surface pecking	1
Total	**84**

4.4.17.3 Species sensitivity

Little gull is listed on Annex I of the EU Birds Directive (2009/147/EEC), and the species is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

A recent review assessed little gull as being at low risk of collision with turbines, displacement, barrier effects, habitat loss or changes in prey distribution resulting from offshore wind farms (Langston 2010) (Table 4.4).

4.4.17.4 Assessment

Definition of seasons

Little gulls were only recorded in the offshore site between August and October. Therefore, this is the only period considered. This time of the year corresponds to the autumn passage and the period they moult their flight feathers. Little gulls do not regularly breed in the UK and the birds recorded in the offshore site are most likely to be from breeding grounds in the Baltic region (Wernham *et al*., 2002).

Baseline conditions

In the autumn passage period (August to October) the mean estimated number of little gulls (birds on the sea and in flight) present in the offshore site was 58, and the mean estimated number in the offshore site buffered to 1 km was 74. On average 87% of birds present in the offshore site buffered to 1 km were on the sea and the remainder were in flight. Little gulls seen in the Survey Area comprised a mix of 70% adults and 30% immature and juvenile birds.

The numbers present in the buffer area in autumn passage period of Year 2 of the surveys were approximately twice those present in Year 1. The reasons for this difference are unknown but may reflect between-year differences in prey availability.

The outer Firth of Forth and outer Tay area is recognised as one of the few places in the UK where relatively large numbers of little gull occur. Skov *et al*. (1995) estimated that this locality holds approximately one fifth of the little gulls present in the UK in the autumn.

Populations

The size of the UK autumn passage population is unknown and this presents a difficulty for undertaking the assessment. Skov *et al*. (1995) gives an autumn passage population estimate of 2,400 birds for the east coast of the UK and English Channel based ESAS survey data. However, ESAS surveys do not include inland waters and estuaries and some of these (e.g. Hornsea Mere, Tay Estuary) are known to regularly attract this species in autumn e.g. a peak of 21,500 little gulls at Hornsea Mere in August 2007. The five-year mean between 2005 and 2010 for Hornsea Mere was 5,868 birds, while the five-year mean for the Tay Estuary over the same period was 116 birds (Holt *et al.,* 2011). Combining the above figures and factoring in the ESAS east coast estimate from Skov *et al*. (1995), it is tentatively estimated that the average autumn UK population of little gulls is in the region of 7,500 birds. This figure is used for assessment of effects on the national population. There is also evidence from counts at coastal sites that there are large year-to-year variations in the number of birds that come across the North Sea to Britain (Forrester *et al.,* 2007) and that in some years the total number of individuals greatly exceeds 5,000 (Hartley 2004).

Analysis of ESAS data by Skov *et al*. (1995) identifies a geographically discrete autumn passage concentration in the outer Firth of Forth and Firth of Tay (referred to as Tay Bay by Skov *et al*.) and this is taken to be the regional population for assessment purposes. There is uncertainty regarding the current size of this population as the number estimated by Skov *et al*. (450 birds) is far lower than the typical total of about 1,000 birds seen at coastal roost counts in Fife and Lothian (Forrester *et al.,* 2007). Furthermore, survey work commissioned in recent years to inform the proposed wind farms in the Firth of Forth area show that this species is more common than previously appreciated (or numbers have increased), with estimates for the Neart na Gaoithe study area of 1,781 individuals in October of Year 1 and 2,923 individuals in October of Year 2. For the purposes of assessment the regional autumn passage population is assumed to be 3,100 birds. This is the mean of the Years 1 and 2 October estimates for the Neart na Gaoithe study area (2,350 birds) plus the maximum numbers recorded in the first year of baseline surveys in the Inch Cape survey area (approximately 370 birds, RPS 2012) and Firth of Forth Round 3 Zone survey area (estimated 378 birds, SWEL, 2011). This is likely to be below the actual regional population size because the combined survey area covered by these three projects excludes large areas of the region including much of the outer Tay Estuary where little gull are known to commonly occur (Skov *et al.,* 1995).

This species has undergone a recent period of sustained population recovery in Western Europe and is now considered to be in favourable conservation status (BirdLife International, 2004). Therefore, it is likely that the regional autumn passage population has low sensitivity to additional mortality.

Nature conservation importance

For EIA assessment purposes, the nature conservation importance of little gulls using the site is rated as high during the autumn passage period. The species merits this rating because it is listed on Annex 1 of the Birds Directive and because the numbers present in the offshore site in the autumn period may be of national importance (they represent approximately 1% of the UK population).

Offshore wind farms studies of little gull

Evidence from existing projects regarding the extent to which little gulls are displaced from operational wind farms, and therefore the extent to which this species is likely to be displaced from the proposed development is unclear. At Horns Rev, Denmark, little gulls were relatively less abundant in the wind farm compared to the wider survey area, but not significantly so, during the pre-construction and construction phases, but were significantly more abundant in the wind farm during the operational phase (Diersche and Garthe, 2006). Additionally, during one aerial survey most little gulls observed were foraging between the turbines. Although these results suggest that little gulls showed a preference for the operational wind farm, this effect was not evident during the spring migration. At Egmond aan Zee, the Netherlands, little gulls were principally recorded during the spring migration. Results from this study showed statistically significant avoidance of the wind farm during one survey visit and non-significant results (neither attraction nor avoidance) for a further six survey visits (Leopold *et al*., 2011). At Arklow Bank, Ireland, where large numbers of little gulls were recorded during the autumn and winter periods, numbers increased after turbines became operational compared to baseline surveys, with the increase concentrated in the vicinity of the turbines (Barton *et al*., 2009, Barton *et al*., 2010).

At Horns Rev, Denmark, visual monitoring from an observation platform positioned at the edge of the wind farm found that 13% (sample size not given) of flying little gulls were either within or flying into the wind farm, indicating that turbines act as only a partial barrier to flying little gulls. At Egmond aan Zee, where over 90% of little gulls were recorded during spring migration, there was little statistical evidence to indicate that little gulls avoided, or were attracted to the wind farm. However, the authors report that little gulls were rarely seen inside the wind farm and most appeared to "prefer flying around the wind farm rather than entering it" (Leopold *et al*., 2011). In a summary of wind farm effects on birds in German marine areas few or no barrier effects on little gull were reported (collated data in Diersche and Garthe, 2006).

The risk of little gulls colliding with wind turbines is likely to be low based on reported flying heights and recorded fatalities from operational wind farms. At Arklow Bank, of approximately 2,000 records of flying little gulls collected over a two-year period, mostly relating to the autumn migration period, less than 5% of birds were recorded flying at a height over 20 m, with over 80% of the total flying between 0 – 5 m above the sea surface (Barton *et al*., 2009, Barton *et al*., 2010). The review of offshore wind farm effects on birds categorises little gull collision risk as unknown (Diersche and Garthe, 2006) and no little gull fatalities were reported in a review of the number of collision victims at wind farms in eight

European countries (Hötker *et al*., 2006), although the very low probability of detecting seabird fatalities should be recognised.

EIA Construction Phase assessment for little gull

The construction phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

Little gulls are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during construction operations on the regional autumn passage populations of little gulls is ***not significant*** under the EIA Regulations.

EIA Operational phase assessment for little gull

Potential for little gull populations to be affected by displacement

The value of the offshore site buffered to 1 km as a foraging site for little gulls in the autumn passage period was estimated from the mean proportion of assumed North Sea, national and regional populations present from August to October (Table 4.34).

On this basis, it is inferred that the offshore site buffered to 1 km provides foraging resources for up to 0.8% of the North Sea (international) population, 1.0% of the national population and 2.4% of the regional population during the autumn passage period. It is concluded that the offshore site is likely to be of national importance (>1% of population present) and is certainly of regional importance for this species at this time of year.

Table 4.34 The mean estimated number and percentage of the at-sea little gull population potentially at risk of displacement from the offshore site and 1 km buffer during the autumn passage period

Receptor population	Population size (adults)	No. assumed at sea	Dev. Area		Dev. Area + 1km	
			Mean no. at risk	%	Mean no. at risk	%
National, autumn period (Skov *et al.*, 1995, Stone *et al.*, 1995)	7,500	7,500	58	0.8	74	1.0
North Sea, autumn period (Skov *et al.*, 1995)	9,000	9,000	58	0.6	74	0.8
Regional, autumn period (Skov *et al.*, 1995, Forrester *et al.*, 2007)	3,100	3,100	58	1.9	74	2.4

Likely impacts of displacement on little gull populations

For the purposes of assessment, it is assumed that 25% of the potential displacement would be realised, a figure that is likely to be cautious as evidence from operational wind farms mostly indicates little or no displacement effect from wind farms on this species. It is unknown whether displaced birds would be disadvantaged or whether they would merely move to alternative foraging areas with capacity to hold more birds. The species undertakes large movements and therefore displaced birds are likely to move elsewhere to seek out alternative foraging sites. Nevertheless, the species is also concentrated into relatively small favoured areas (Skov *et al.,* 1995) and the extent to which these are used to capacity is unknown. Recognising this uncertainty but also taking a cautionary approach, it is considered that the regional autumn passage population of little gull potentially has moderate sensitivity to displacement from foraging areas.

Were 25% of little gulls to be displaced from the offshore site buffered to 1 km the impact of this would be the effective loss of up to 0.6% of the foraging habitat of the regional autumn passage population (25% of the value shown in Table 4.34). This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2).

Taking into consideration the above assessment and the high NCI of this species it is concluded the impact of displacement from foraging areas on the regional autumn passage population of little gulls is ***not significant*** under the EIA Regulations.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

Little gulls are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional autumn passage populations of little gulls is ***not significant*** under the EIA Regulations.

Collision mortality

For the purposes of assessment it is assumed that there will be no far-field avoidance of the offshore site by little gulls. Therefore, the predicted mortality for wind farm designs and assumed avoidance rate shown in Table 4.35 and Table 4.36 are the same as that likely to occur.

There is no published estimate of the annual adult mortality rate for little gull. For the purposes of assessment a value of 10.0% is assumed. This is the same as for black-headed gull (BTO, 2012), a closely related species with a similar life history.

COWRIE guidance (Maclean *et al.,* 2009) is to use a default avoidance rate of 99.5% for seabirds. There is no specific SNH guidance on avoidance rates for seabirds, and therefore their default value of 98.0% is applicable.

Approximately a third of flying little gulls observed in baseline surveys were within the height range swept by the proposed turbine rotors (Table 4.35).

Table 4.35 Proportion of little gull estimated to be at rotor height and the number of collisions predicted by collision rate modelling (CRM) using avoidance rates (AR) of 98.0% and 99.5% for least adverse and most adverse wind farm designs evaluated. LAT is sea level at lowest astronomical tide, and this is approximately 2.0 m below the average sea level

Metric	Design 1, 64 x 7MW		Design 4, 128 x 3.6 MW	
	98.0% AR	99.5% AR	98.0% AR	99.5% AR
Rotor swept height range (metres above LAT)	24 - 188		24 - 144	
Flight activity at rotor height	30.6%		30.6%	
Flight activity below rotor height (<22.5 m above sea)	69.4%		69.4%	
Collisions in autumn passage period (August to October), all ages	6.0	1.5	9.0	2.3

Table 4.36 The indicative effect of CRM predicted collision mortality on the adult mortality rate (AMR) of little gull populations

Wind farm Design / Receptor population	Receptor population size	98.0% OAR		99.5% OAR	
		CRM deaths	Change AMR	CRM deaths	Change AMR
Design 1, 64 x 7MW turbines					
North Sea	9,000	12	1.3%	3	0.3%
National, all year	7,500	12	1.6%	3	0.4%
Regional, all year	3,100	12	3.9%	3	1.0%
Design 4, 128 x 3.6MW turbines					
North Sea	9,000	17	1.9%	4	0.5%
National, all year	7,500	17	2.3%	4	0.6%
Regional, all year	3,100	17	5.5%	4	1.4%

The highest potential collision rates are for Wind Farm Design 4 (128 x 3.6 MW turbines). Under this design it is estimated that the average number of little gulls collisions would be 17 birds using an avoidance rate of 98.0%, and 4.3 birds using an avoidance rate 99.5%) (Table 4.36). The corresponding figure for the least adverse design (Wind Farm Design 1) is 12 collsions per annum.

The impact of this additional mortality based on an avoidance rate of 98.0% would be an increase in the assumed annual adult mortality rate of between 1.6% (Wind farm Design 1) and 2.3% (Wind farm Design 4) of the national population, and between 3.9% (Wind farm Design 1) and 5.5% (Wind farm Design 4) of the regional population (Table 4.36). The potential impact of this level of collision mortality on both the national and regional little gull population is an effect of low magnitude (1-5%) and temporally long-term and reversible (Table 3.1 and Table 3.2). Given that this species is categorised as high NCI, it is concluded that the effects of collision mortality on the regional autumn passage population is a ***moderate significant*** effect under the terms of the Electricity Regulations. It should be noted that this conclusion is potentially sensitive to the size of the regional and national population used for comparison, about which there is some uncertainty. However, even if the regional population was twice as high as is assumed (i.e. 6,200 birds instead of 3,100), then this impact would remain of moderate significance under the EIA Regulations, for all wind farm designs examined and assessed using a 98.0% avoidance rate (but not significant using a 99.5% rate). The assessment conclusions are also potentially sensitive to the baseline adult mortality rate used in calculations. This is unknown for little gull and was therefore assumed to be 10%, this being the value for black-headed gull. If a substantially less cautious baseline mortality rate of 15% is assumed (lower than for any UK species of gull reported on the BTO BirdFacts website) the impact on the survival rate due to predicted collision mortality would remain at moderately significant.

Using an avoidance rate of 99.5% to predict the number of collision strikes, the additional mortality would cause the assumed annual adult mortality rate of the regional population to increase by between 1.0% (Wind farm Design 1) to 1.4% (Wind farm Design 4) (Table 4.36). Assessed for 99.5% avoidance rate, the potential impact of the predicted collision mortality on the regional little gull population is an effect of low magnitude (1-5%) and temporally long-term and reversible (Table 3.1 and Table 3.2). Given that this species is categorised as high NCI, it is concluded that the effects of collision mortality on the national autumn passage population is likely to be judged as an effect of ***minor significance*** under the terms of the Electricity Act.

EIA Decommissioning Phase assessment for little gull

The decommissioning phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

Little gulls are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during decommissioning operations on the regional autumn passage population of little gulls is ***not significant*** under the EIA Regulations.

EIA summary of effects combined for little gull

The adverse impacts of the effects assessed will act in a broadly additive manner. In combination it is judged that the magnitude of the effects on the regional population is low. The population is categorised as having moderate sensitivity to collision and displacement effects and is categorised as having high NCI.

Conclusions on the overall impact of the proposed development are sensitive to the level of avoidance assumed for predictions of collision deaths. If a 98.0% avoidance rate is assumed it is concluded that the overall impact of the proposed development on the regional population in the autumn passage period is an effect of ***moderate significance*** under the EIA regulations (Table 4.37).

If a 99.5% avoidance rate is assumed it is concluded that the overall impact of the proposed development on the regional population in the autumn passage period is an effect of ***minor significance*** under the EIA regulations.

Table 4.37 Summary of effects on the regional population of little gulls in the autumn passage period

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Moderate	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality for 98.0% A R	Moderate	Long term	Moderate	***Moderate significance***
Collision mortality for 99.5% A R	Low	Long term	Moderate	***Minor significance***
All effects combined (collision 98.0% A R)	Moderate	Long term	Moderate	***Moderate significance***
All effects combined (collision 99.5% A R)	Low	Long term	Moderate	***Minor significance***

4.4.17.5 Cumulative Impact Assessment for little gull

Adding the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts on the regional autumn passage population of little gull will be as shown in Table 4.38. The Year 1 reports of survey work at Inch Cape and the Firth of Forth Round 3 Zone (RPS, 2012; SWEL, 2011) do not give collision mortality estimates for this species despite moderate numbers being recorded. It is assumed for the purposes of this CIA the magnitude category of the collision impact by these two developments is the same as for Neart na Gaoithe (i.e., 'moderate' for calculations based on an avoidance rate of 98.0% and 'low' for an avoidance rate of 99.5%).

Predicted displacement causing the potential effective loss of up to 1.7% of the foraging habitat of the regional autumn passage population of little gull is rated as an effect of low magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). Bearing in mind that little gull is rated as high NCI, it is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional autumn passage population of little gull is of ***minor significance*** under the terms of the EIA Regulations.

Table 4.38 Summary of CIA for the three proposed offshore wind farm in south-east Scotland on the regional autumn passage population of little gull

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF	Predicted impact from Inch Cape pOWF	Predicted impact from Firth of Forth Round 3 Zone pOWF	Cumulative Impact
Displacement: effective loss of foraging habitat during autumn passage period (%)	25%	0.6%.	0.2%	0.9%	1.7%, Low
Collision most adverse designs: % increase in assumed annual adult mortality rate based on 98.0% avoidance rate.	0%	5.5%, moderate	no data, assumed moderate	no data, assumed moderate	High
Collision least adverse designs: % increase in assumed annual adult mortality rate based on 98.0% avoidance rate.	0%	3.9%, moderate	no data, assumed moderate	no data, assumed moderate	High
Collision most adverse designs: % increase in assumed annual adult mortality rate based on 99.5% avoidance rate.	0%	1.4%, low	no data, assumed low	no data, assumed low	Moderate
Collision least adverse designs: % increase in assumed annual adult mortality rate based on 99.5% avoidance rate.	0%	1.0%, low	no data, assumed low	no data, assumed low	Moderate

Conclusion on the cumulative impact of collision mortality on the regional autumn passage population of little gull is sensitive to the wind farm designs evaluated and the level of avoidance rate used for predictive calculations of the number of collision strikes. There is also uncertainty over the cumulative impact of collision mortality on little gulls because of the lack of modelling results for this species for the Inch Cape and Firth of Forth Round 3 Zone developments, and because of the uncertainty over the baseline mortality rate for this species.

Using an avoidance rate of 99.5% the cumulative impact of the three wind farms in terms of the predicted increase to the annual adult mortality rate is provisionally estimated to be an effect of moderate magnitude (Table 4.38), temporally long-term and reversible (Table 3.1 and Table 3.2). On this basis it is concluded that, when assessed using a 99.5% avoidance rate, the cumulative impact of collision mortality caused by the three proposed offshore wind farms and on the regional autumn passage population of little gull is of ***moderate significance*** under the terms of the EIA Regulations.

Using an avoidance rate of 98.0% the cumulative impact of the three wind farms in terms of the predicted increase to the annual adult mortality rate is provisionally estimated to be an effect of high magnitude (Table 4.38), temporally long-term and reversible (Table 3.1 and Table 3.2). On this basis it is concluded that, when assessed using a 98.0% avoidance rate, the cumulative impact of collision mortality caused by the three proposed offshore wind farms and on the regional autumn passage population of little gull is of ***major significance*** under the terms of the EIA Regulations.

10.3.1.2 Mitigation measures for little gull

There are few if any practical mitigation measures that are likely to significantly reduce the potential collision mortality for little gull.

4.4.18 Sabine's gull *Larus sabini*

4.4.18.1 Status

Sabine's gull is a scarce but regular passage migrant off the Scottish coast, with most birds recorded in autumn, primarily off the west coast (Forrester *et al.,* 2007). Off the Fife coast, it is classed as a rare autumn migrant, with low numbers recorded in most years (Dickson 2002).

4.4.18.2 Neart na Gaoithe Study Area

One adult Sabine's gull was recorded in Year 1, in the south-west of the Neart na Gaoithe offshore site in August (Table 4.2) (Figure 4.41). One was also recorded in Year 2, in the buffer area in June. Both birds were flying between 7.5 and 12.5 m in height.

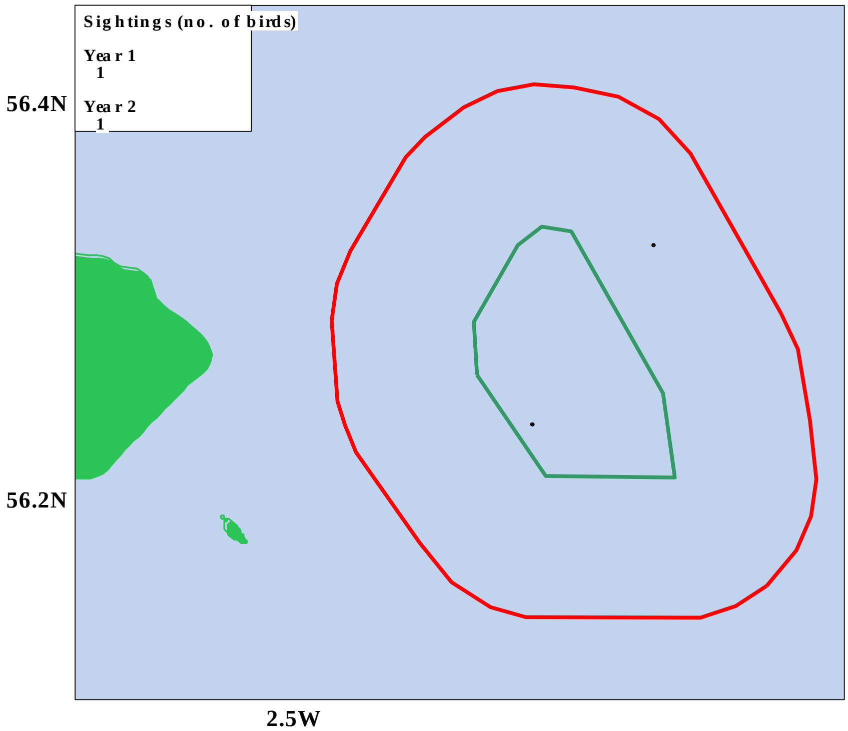


Figure 4.41 Sabine's gull sightings in Years 1 and 2

4.4.18.3 Assessment

The very small number of birds recorded on baseline surveys, with only one individual recorded in the offshore site in August of Year 1, indicate that Sabine's gull is unlikely to be affected by the Development.

4.4.19 Black-headed gull *Larus ridibundus*

4.4.19.1 Status

Black-headed gulls are common and widespread in Britain and occur both inland and on the coast, although they are rarely found far offshore. In summer, birds breed at inland and coastal colonies. Seabird 2000 recorded 127,907 pairs of black-headed gulls breeding in Britain (Mitchell *et al.,* 2004). The nearest major breeding colony to the Neart na Gaoithe site is inland, at Loch Leven in Fife, with a population of 6,832 pairs recorded during Seabird 2000 (Mitchell *et al.,* 2004).

4.4.19.2 Neart na Gaoithe Study Area

No black-headed gulls were recorded in the offshore site on surveys in Years 1 or 2. A total of 27 black-headed gulls were recorded on surveys in the buffer area in Year 1, with a peak count of 25 in November (Table 4.2). Fewer black-headed gulls were seen on surveys in Year 2, with 11 birds recorded between March and October, and a peak of 6 birds in August. The majority of birds were in the west of the study area, over shallower water (Figure 4.42).

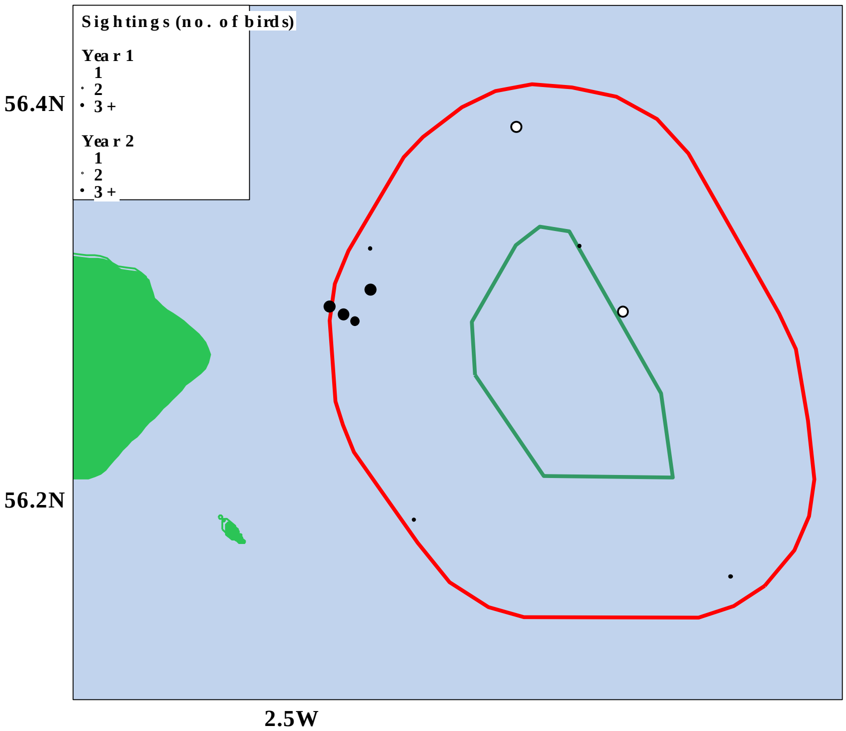


Figure 4.42 Black-headed gull sightings in Years 1 and 2

In Years 1 and 2, 38 black-headed gulls were recorded in flight, with 28 birds (73.7%) flying below 22.5 m and 10 birds (26.3%) flying above 22.5 m, i.e. within the rotor swept zone, at estimated heights of 25 and 30 m. (Table 4.3).

10.3.1.3 Species sensitivity

Black-headed gull is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009). A recent review assessed black-headed gull as being at low risk of collision with turbines, displacement, barrier effects, habitat loss or changes in prey distribution resulting from offshore wind farms. Overall, black-headed gull was assessed as being at low risk from offshore wind developments (Langston 2010) (Table 4.4).

Black-headed gull is listed as a qualifying interest species in the breeding season for one SPA on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development: Coquet Island SPA. This SPA held 1.3% of the UK breeding population, and 0.1% of the biogeographic population at the time of designation (JNCC, 2012). Since designation, the population at Coquet Island has increased to an estimated 3,385 pairs in 2009 (SMP, 2012). The distance between the offshore site and this SPA is greater than the maximum known foraging range of this species (40 km) (Thaxter *et al.*, 2012).

4.4.19.3 Assessment

No black-headed gulls were recorded in the offshore site on baseline surveys. Based on these results and the predominantly coastal distribution of the species, it is considered that black-headed gulls are unlikely to be affected by the Development.

Displacement

It is not known whether black-headed gulls will be displaced by the offshore site. The review undertaken by Langston (2010) suggested that black-headed gulls are at low risk of displacement effects. No black-headed gulls were recorded in the offshore site on baseline surveys. Overall, it is concluded that any effects of displacement on black-headed gulls is ***not significant*** under the terms of the EIA Regulations.

Barrier effect

The nearest large breeding colony is at Loch Leven in Fife, and the offshore site is beyond the maximum known foraging range for breeding birds (Thaxter *et al*., 2012). Therefore no barrier effects on breeding birds are predicted to occur during the breeding period. Outwith the breeding season black-headed gulls are predominantly found in coastal waters or inland (Forrester *et al*., 2007). Based on this, and the absence of black-headed gulls recorded in the offshore site on baseline surveys, it is concluded that the effects of barrier effect on black-headed gull is ***not significant*** under the terms of the EIA Regulations.

Collision Mortality

Collision risk modelling was not undertaken for black-headed gulls, due to the low numbers of birds recorded in flight on baseline surveys (38 birds). Although 26.3% of all flying birds were recorded flying above 22.5 m, i.e. within the rotor swept zone, there were no black-headed gulls recorded within the offshore site on baseline surveys. Based on this, it is predicted that black-headed gulls will not experience significant mortality from collision with turbine rotors.

Overall, the potential effect of the collision mortality of black-headed gulls on the baseline mortality rate is rated as negligible in magnitude, temporally long-term and reversible. It is concluded that the effects of collision mortality on black-headed gulls is ***not significant*** under the terms of the EIA Regulations.

4.4.20 Common gull *Larus canus*

4.4.20.1 Status

Common gulls are common and widespread in lowland, urban and coastal areas in winter, and breed in colonies in coastal and inland locations in summer. Seabird 2000 recorded 48,163 pairs of common gulls in Britain (Mitchell *et al.,* 2004). Common gulls typically feed on farmland, playing fields, estuaries and in coastal waters, and are comparatively uncommon offshore (Forrester *et al.,* 2007, Stone *et al.,* 1995).

4.4.20.2 Neart na Gaoithe Study Area

In Year 1, 78 common gulls were recorded on surveys in the Neart na Gaoithe study area, with the majority of birds (92.3%) in the buffer area in the winter months, peaking in October (28 birds) (Table 4.2). Fewer common gulls were seen on surveys in Year 2, with 52 birds recorded, the majority of which (76.9%) occurred in the buffer area in the winter months, peaking in December (15 birds).

Due to the low sample size of common gulls recorded on baseline surveys, it was not possible to conduct Distance analysis on the data. Abundance rates (birds/km) were calculated instead.

Mean monthly common gull abundance was generally low in the offshore site and the buffer area in Years 1 and 2, with a peak of 0.2 birds/km in the buffer area in December of Year 2 (Figure 4.43). ESAS abundance data from the surrounding ICES rectangles and across Regional Sea 1 was low.

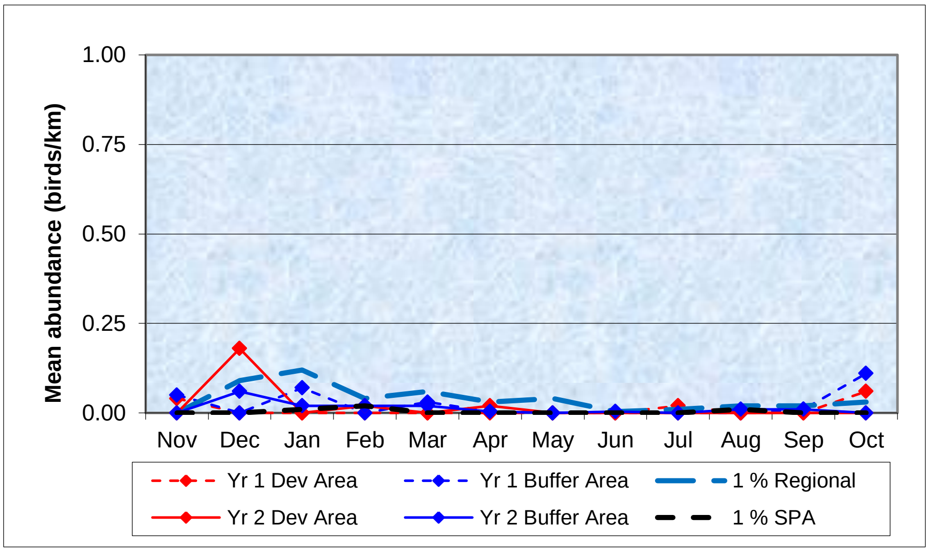


Figure 4.43 Comparison of common gull monthly mean abundance in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

A total of 104 common gulls were recorded in flight in Years 1 and 2, with 72.1% of birds flying below 22.5 m, and 29 birds (27.9%) flying above 22.5 m, i.e. within the rotor swept zone, at estimated heights of between 25 and 50 m. (Table 4.3).

Common gulls were scattered sporadically in the western half of the Neart na Gaoithe study area at low abundances in Year 1 (Figure 4.44). Few birds were recorded in the offshore site in Year 1.

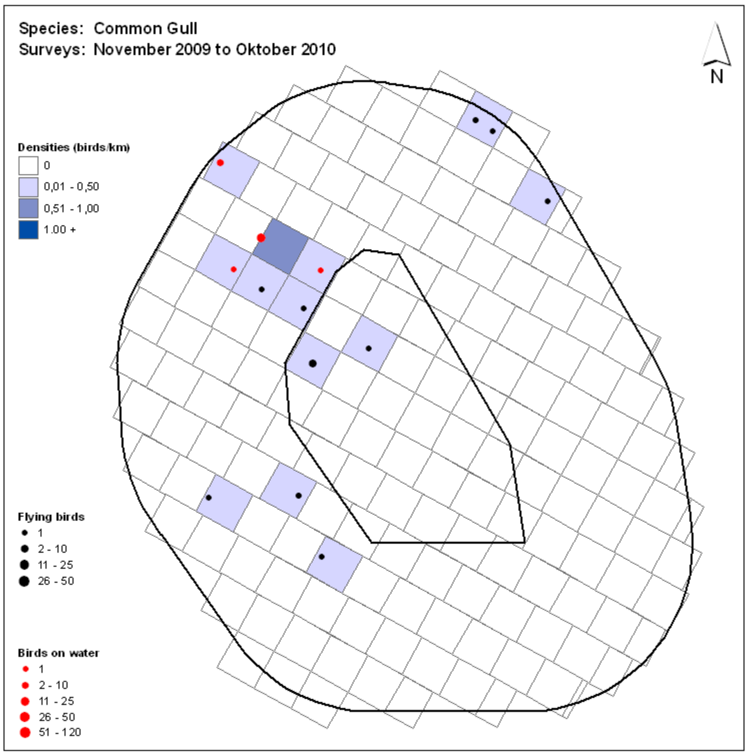


Figure 4.44 Common gull abundance all months combined, Year 1

A similar distribution pattern was recorded in Year 2, with common gulls scattered at low abundance predominantly in the western half of the study area (Figure 4.45). As in Year 1, few common gulls were recorded in the offshore site over the period.

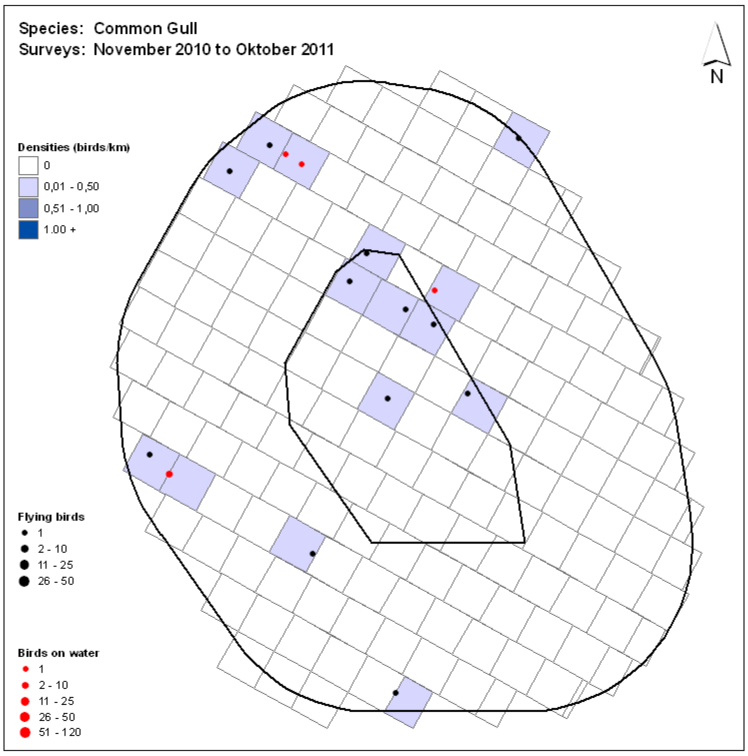


Figure 4.45 Common gull abundance all months combined, Year 2

4.4.20.3 Species sensitivity

Common gull is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

A recent review assessed common gull as being at low risk of collision with turbines, displacement, barrier effects, habitat loss or changes in prey distribution resulting from offshore wind farms. Overall, common gull was assessed as being at moderate risk from offshore wind developments Langston (2010) (Table 4.4).

Common gull is listed as a qualifying interest species in the breeding season for one inland SPA that could potentially be affected by the Neart na Gaoithe development; Tips of Corsemaul and Tom Mor SPA. This SPA held 18,000 pairs or 26.5% of the UK breeding population, and 14.5% of the biogeographic population at the time of designation (JNCC, 2012). A similar figure (18,136 pairs) was recorded in 1998, however no more recent data was available (SMP, 2012). The distance between the offshore site and this SPA is greater than the known maximum foraging range (50 km) for this species (Thaxter *et al*., 2012).

4.4.20.4 Assessment

Numbers of common gulls recorded in the offshore site on baseline surveys were low. Based on this, and the predominantly coastal distribution of the species, it is considered that common gulls are unlikely to be affected by the Development.

Displacement

It is not known whether common gulls will be displaced by the offshore site. The review undertaken by Langston (2010) suggested that common gulls are at low risk of displacement effects. Low numbers of common gulls were recorded in the offshore site on baseline surveys. Overall, it is concluded that any effects of displacement on common gulls is ***not significant*** under the terms of the EIA Regulations.

Barrier effect

There are no breeding colonies within 50 km of the offshore site, which is the maximum known foraging range for breeding birds (Thaxter *et al*., 2012). Therefore no barrier effects on breeding birds are predicted to occur during the breeding period. Outwith the breeding season black-headed gulls are predominantly found in coastal waters or inland (Forrester *et al*., 2007). Based on this, and the low numbers of common gulls recorded in the offshore site on baseline surveys, it is concluded that the effects of barrier effect on common gull is ***not significant*** under the terms of the EIA Regulations.

Collision Mortality

Collision risk modelling was not undertaken for common gulls, due to the low numbers of birds recorded in flight on baseline surveys (104 birds). Although 27.9% of all flying birds were recorded flying above 22.5 m, i.e. within the rotor swept zone, numbers recorded within the offshore site on baseline surveys were very low. Based on this, it is predicted that common gulls will not experience significant mortality from collision with turbine rotors.

Overall, the potential effect of the collision mortality of common gulls on the baseline mortality rate is rated as negligible in magnitude, temporally long-term and reversible. It is concluded that the effects of collision mortality on common gulls is ***not significant*** under the terms of the EIA Regulations.

4.4.21 Lesser black-backed gull *Larus fuscus*

4.4.21.1 Status

Lesser black-backed gulls are common and widespread in summer, and breed in colonies in coastal and inland locations. In winter, many birds leave Scotland between November and March, although some remain all year, particularly in the south-west (Forrester *et al*., 2007). Seabird 2000 recorded 111,835 breeding pairs in Britain (Mitchell *et al*., 2004). The nearest large colonies to the Neart na Gaoithe development are on the islands in the Firth of Forth, and the Isle of May. Lesser black backed gulls take a wide variety of prey and scavenged food, both at sea, and on farmland and refuse sites (Forrester *et al*., 2007).

4.4.21.2 Neart na Gaoithe Study Area

In Year 1, 66 lesser black-backed gulls were recorded on surveys in the Neart na Gaoithe study area. Low numbers (10 birds) were recorded in the offshore site, with the majority of birds (84.8%) in the buffer area in the summer months, peaking in June (22 birds) (Table 4.2). Numbers on surveys in Year 2 were higher, with 195 birds recorded, although low numbers (11 birds) were again recorded in the offshore site. The majority of birds (94.4%) were seen in the buffer area in the summer months, peaking in September (120 birds).

Due to the low sample size of lesser black-backed gulls recorded in Years 1 and 2, it was not possible to conduct Distance analysis on the data. Abundance rates (birds/km) were calculated instead.

Mean monthly lesser black-backed gull abundance was generally low in the offshore site and the buffer area in Years 1 and 2, with a peak of 0.4 birds/km in the buffer area in September of Year 2 (Figure 4.46). ESAS abundance data from the surrounding ICES rectangles and across Regional Sea 1 was low.

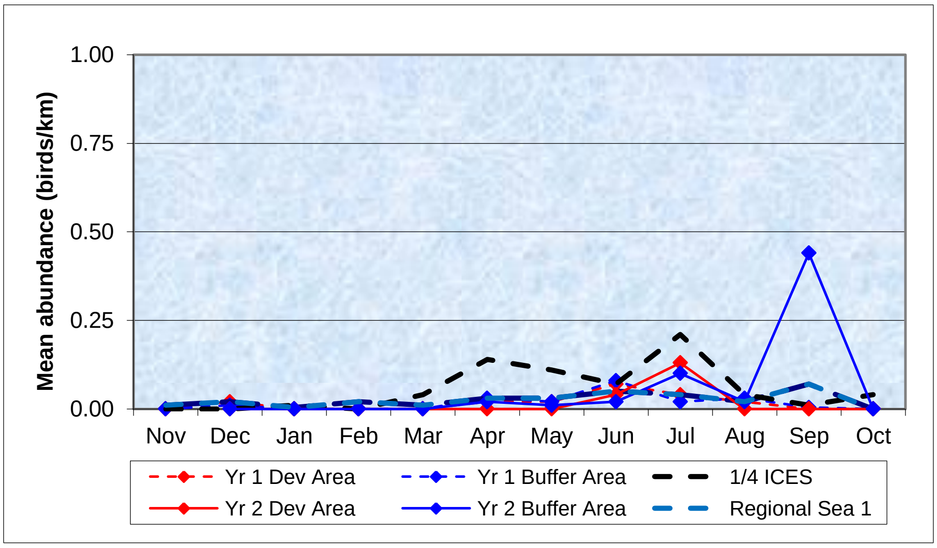


Figure 4.46 Comparison of lesser black-backed gull monthly mean abundance in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

Lesser black-backed gulls were scattered sporadically throughout the southern half of the Neart na Gaoithe study area at low abundances between April and September of Year 1 (Figure 4.47). Few birds were recorded in the offshore site over the period.

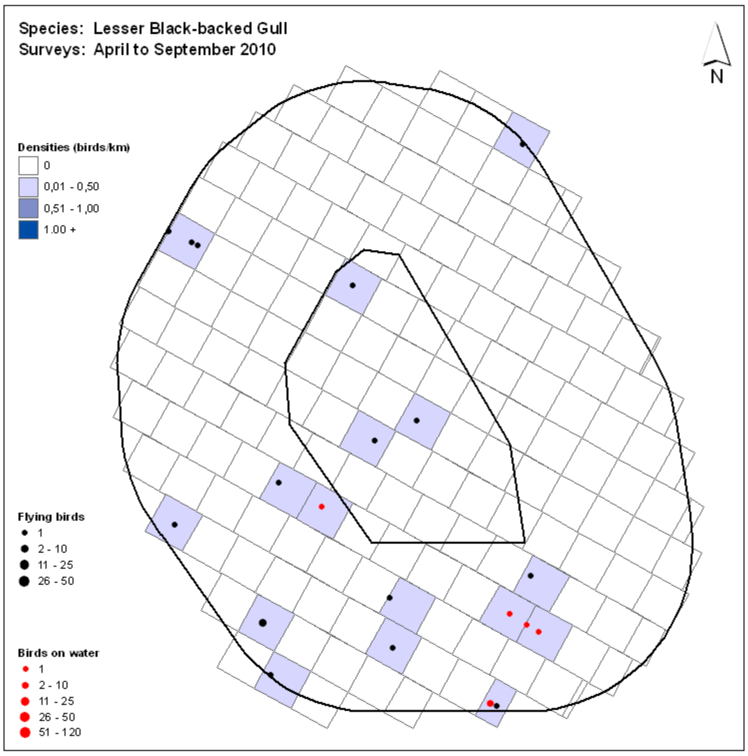


Figure 4.47 Lesser black-backed gull abundance between April and September, Year 1

A similar distribution pattern was recorded in Year 2 between April and September, with highest abundance of lesser black-backed gulls recorded in the south-west of the buffer area (Figure 4.48). Fewer lesser black-backed gulls were recorded in the offshore site over the period than in Year 1.

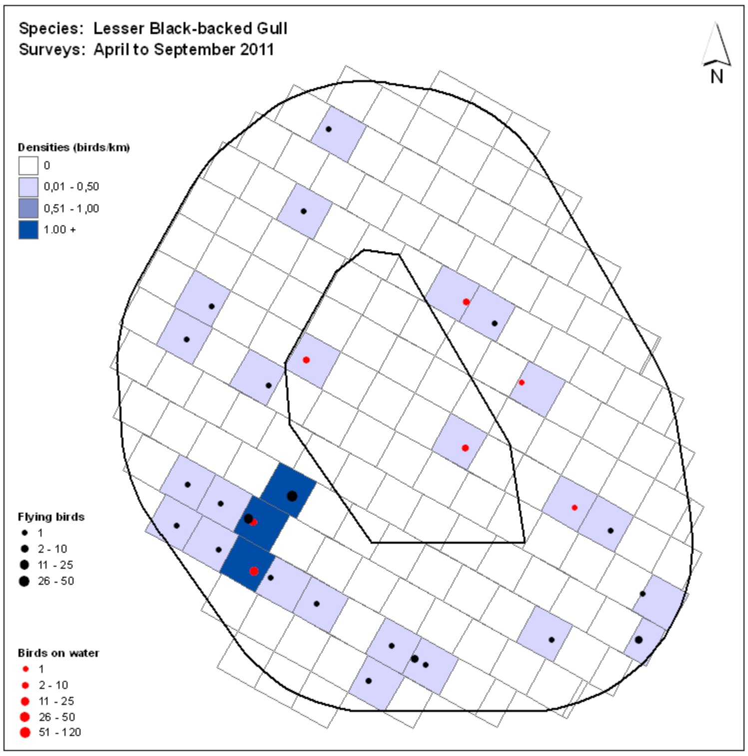


Figure 4.48 Lesser black-backed gull abundance between April and September, Year 2

A total of 216 lesser black-backed gulls were recorded in flight in Years 1 and 2, with 86.1% of birds flying below 22.5 m i.e. below the turbine swept zone (Table 4.3). A total of 30 birds (13.9%) were recorded flying above 22.5 m, i.e. within the rotor swept zone, at estimated heights of 25 to 50 m.

10.3.1.4 Species sensitivity

Lesser black-backed gull is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

A recent review assessed lesser black-backed gull as being at moderate risk of collision. Displacement, barrier effects, habitat loss or changes in prey distribution resulting from offshore wind farms were rated as low risk. Overall, lesser black-backed gull was assessed as being at high risk from offshore wind developments, when the importance of its UK breeding population was taken into account Langston (2010) (Table 4.4).

Lesser black-backed gull is listed as a qualifying interest species in the breeding season for one SPA on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development: Forth Islands SPA. This SPA held 3.5% of the UK breeding population, and 2.4% of the biogeographic population at the time of designation (JNCC, 2012). Since designation, the breeding population at the SPAs has decreased from 2,920 pairs to 2,061 pairs (SMP, 2012). The distance between the offshore site and the Forth Islands SPA is within the mean maximum foraging range of lesser black-backed gull (132.1 km) (Thaxter *et al*., 2012). Lesser black-backed gull mean maximum foraging range from the Firth of Forth breeding SPA in relation to the offshore site is shown in Figure 4.49.

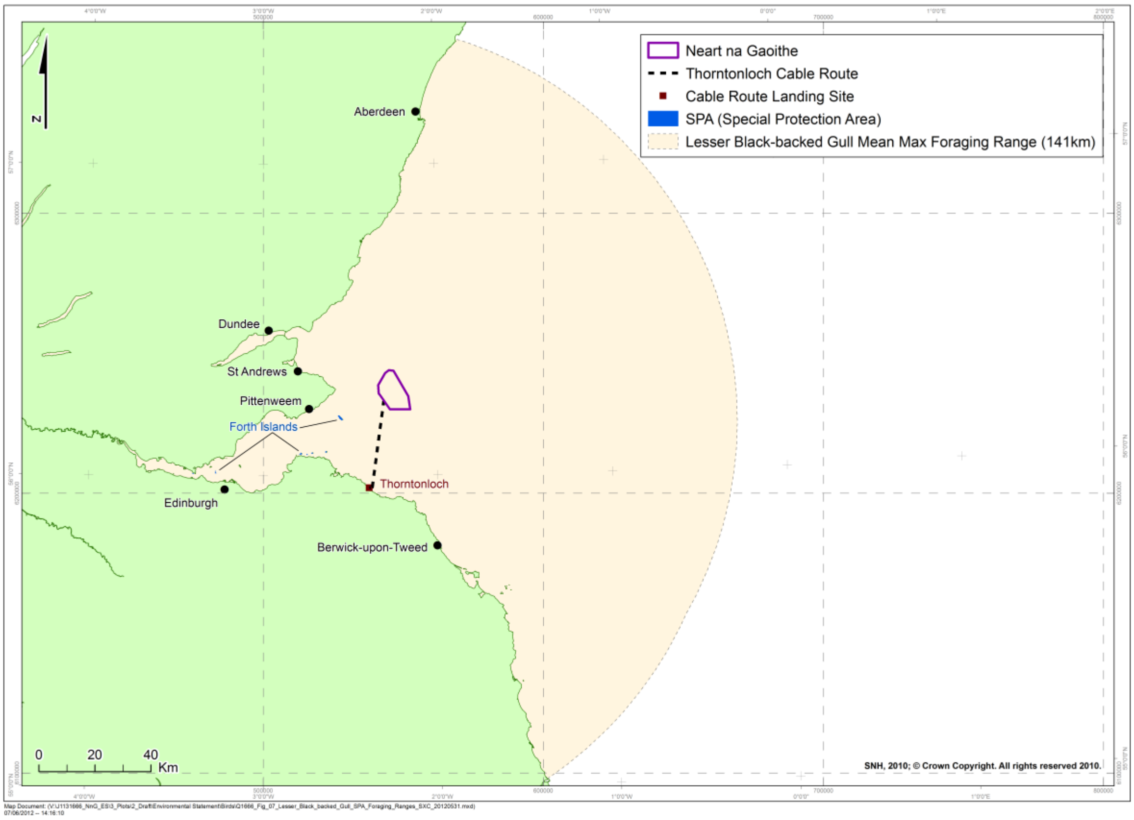


Figure 4.49 Lesser black-backed gull mean maximum foraging range from the Firth of Forth breeding SPA in relation to the Development

4.4.21.3 Assessment

Definition of seasons

Lesser black-backed gulls are predominantly a summer visitor and were mainly recorded in the offshore site during the breeding season, although numbers were low. This was defined as April to August. During this period the majority of birds present in the offshore site will be from relatively local breeding colonies, i.e., colonies that are less than the mean maximum foraging range distance from the offshore site.

Baseline conditions

In the breeding season the mean estimated number of lesser black-backed gulls present in the offshore site was 11, and the mean estimated number in the offshore site buffered to 1 km was 14.

Based on survey results for the whole study area in the breeding season, 85% of lesser black-backed gulls were in adult plumage and the rest were in immature plumage. For the purposes of assessment it was assumed that all birds in adult plumage were breeding birds. It was assumed that the mean colony attendance rate of lesser black-backed gull during the breeding season was 50% and this figure is used to estimate the importance of the offshore site in the breeding season.

Populations

Recent colony counts indicate the breeding population of lesser black-backed gulls in south-east Scotland has undergone long term increase (Mitchel *et al.,* 2004). Assuming that the sub-set of colonies that have been recently counted are representative, the regional population increase since the Seabird 2000 counts amounts to approximately 26%. The increase in breeding numbers mean that published figures on population size for this species (e.g., Mitchell *et al*., 2004) no longer accurately reflect the current regional population size. To prevent this causing assessment of effects to be biased high, the regional total derived from Seabird 2000 results has been adjusted upwards by 26%. On this basis, the regional breeding population is assumed to be 22,034 breeding adults (i.e. 11,017 breeding pairs).

The regional population size in the non-breeding period is assumed to be approximately 6,000 birds based on Skov *et al.* (1995).

Nature conservation importance

For EIA assessment purposes, the nature conservation importance of lesser black-backed gulls using the offshore site is rated as moderate during the breeding season. Although, a high proportion of lesser black-backed gulls using the offshore site are likely to be from the breeding colonies within the Forth Islands SPA, this species is classed as moderate NCI because the numbers typically present in the offshore site buffered to 1 km form well below 1% of this SPA population. The mean number present in the offshore site buffered to 1 km in the breeding season is also well below 1% of the (at-sea) regional population, and the species is not subject to any special legislative protection , nor is it on any conservation priority lists.

Offshore wind farm studies of lesser black-backed gull

Results from bird monitoring at operational offshore wind farms indicate that a small proportion of lesser black-backed gulls may be displaced from the footprint of offshore wind farms; however most studies show no significant change in abundance of lesser black-backed gulls between pre-and post-construction surveys. At Egmond aan Zee, the Netherlands, lesser black-backed gulls showed statistically significant avoidance of the wind farm on one survey, and no statistically significant attraction and non-significant results on a further 11 surveys. Lesser Black-backed Gulls were often seen within the perimeters of the Egmond aan Zee and adjacent Princess Amalia wind farms, sometimes resting on the sea or on the foundation structures, sometimes feeding in the tidal wake of the monopoles. These results led the authors to conclude that the Egmond aan Zee wind farm had little effect on lesser black-backed gull distribution (Leopold *et al*., 2011). At Horns Rev, Denmark, changes in lesser-black backed gull distribution between pre- and post-construction were not assessed, however, visual monitoring from an observation platform positioned at the edge of the wind farm found that 32% (sample size not given) of lesser black backed gulls recorded, were either within or flying into the wind farm (Diersche and Garthe, 2006).

Visual and radar studies suggest that operational wind farms present only a partial barrier to lesser black backed gulls. Of 81 lesser black-backed gulls recorded at Zeebrugge Harbour, Belgium, 75% of birds crossing the wind farm did not react to the turbines. The remaining

25% showed a reaction to the turbines but nearly all these birds flew through the wind farm once they had changed direction (Everaert, 2003). At two sets of turbine arrays at Maasvlakte, the Netherlands, sited between breeding gull colonies (including lesser black-backed gull) and their offshore foraging areas, only 3.1% of 751 gulls recorded showed a behavioural reaction to the turbines, and of these only one bird was recorded to turn back (van den Bergh *et al*., 2002). The authors highlight the contrast between this result and the strong avoidance responses by gulls observed at Maasvlakte outside the breeding period, explaining the former as rapid habituation to the turbines during the breeding season or reduced sensitivity to the turbines by the breeding birds (van den Bergh *et al*., 2002). Studies of wind farms as barriers to migration or regular bird flights, reviewed in Hötker *et al*.(2006), found no studies where wind farms acted as a barrier to lesser black-backed gulls and three studies where they were shown not to act as a barrier.

Evidence from operational wind farms suggests that the risk of lesser black-backed gulls colliding with wind turbines is likely to be moderate, based on reported flying height and recorded fatalities. At two turbine arrays at Maasvlakte *ca.* 21% of 92 and 42% of 1,828 lesser black-backed gulls passed through the wind farm at rotor height (van den Bergh *et al*., 2002). Results from the Zeebrugge Harbour coastal wind farm reported a third (32%) of 136 lesser black backed gulls flying at rotor height (16-50 m) (Everaert, 2003). The review of offshore wind farm effects on birds (Diersche and Garthe, 2006) highlight lesser black-backed gull collision fatalities at coastal wind farms, and 45 lesser black-backed gull fatalities were reported in a review of the number of collision victims at wind farms in eight European countries (Hötker *et al*., 2006).

EIA Construction Phase assessment for lesser black-backed gull

The construction phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

Lesser black-backed gulls are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during construction operations on the regional population of lesser black-backed gulls in the breeding season is ***not significant*** under the EIA Regulations.

EIA Operational Phase assessment for lesser black-backed gull

Potential for lesser black-backed gull to be affected by displacement

The value of the offshore site buffered to 1 km as a foraging site to lesser black-backed gulls in the breeding season was estimated from the proportion of the adults likely to be at sea (not attending a colony) that were on average present (Table 4.39). This method is likely to overestimate the actual importance of the area for foraging (and thus leads to cautious assessment conclusions) because it is likely that some birds seen on baseline surveys were not foraging in the area but merely flying through it.

Table 4.39 The mean estimated number of lesser black-backed gull present during the breeding season, and this value as a percentage of the (at-sea) receptor population potentially at risk of displacement

Receptor population (Source)	Population size (adults)	No. assumed at sea	Dev.Area		Dev.Area + 1km	
			Mean no. at risk	%	Mean no. at risk	%
National, breeding period (Seabird 2000)	176,646	88,323	11	<0.1	14	<0.1
Regional, breeding period (Seabird 2000 x 26% increase)	22,034	11,017	11	0.1	14	0.1

On this basis, it is estimated that the offshore site buffered to 1 km provides 0.06% of the foraging resources of the regional population of lesser black-backed gulls in the breeding season (Table 4.39).

Likely impacts of displacement on lesser black-backed gull population

Studies at operational wind farms show that lesser black-backed gulls exhibit only low levels of displacement. Therefore, it is likely that little if any of the potential displacement would be realised. For the purpose of assessing displacement it is assumed that 25% of lesser black-backed gulls will be displaced from the proposed wind farm footprint and a surrounding buffer of 1 km. Based on the modelled evidence from Egmond aan Zee and observations from Horns Rev this is likely to be a cautious conclusion as these studies suggest little effect of wind farms on lesser black backed gull distribution.

If 25% of lesser black-backed gulls were to be displaced from the offshore site buffered to 1 km the impact of this would be the effective loss of <0.1% of the foraging habitat of the regional breeding population (25% of the values in Table 4.39). This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of displacement on the regional lesser black-backed gull population in the breeding season are ***not significant*** under the EIA Regulations.

Barrier effect

Lesser black-backed gulls are considered to have low sensitivity to barrier effects on account of a low wing loading (Maclean *et al.*, 2009). The potential effects on lesser black-backed gull of the proposed wind farm acting as a barrier are assessed for the breeding season, when birds are attending colonies.

The only breeding colonies in the region where lesser black-backed gulls are potentially affected by the proposed development acting as a barrier are the Isle of May and Craigleith.

For the purposes of assessment the width of the barrier is assumed to extend 1 km either side of the maximum width of the proposed wind farm. Observations from operational offshore wind farms show no evidence that wind farms pose a barrier to lesser black-backed gulls. It is therefore likely that only a small percentage, if any, of foraging flights potentially intercepted by the barrier would be affected. For the reasons described earlier when discussing displacement, it is assumed that only 25% of birds reaching the barrier will respond by detouring around the wind farm. Barrier effects as assessed here concern birds which would otherwise fly over the offshore site to access feeding resources beyond it. The effect on birds which pass the front edge of the offshore site but would intend to forage within it is considered through the assessment of displacement. Therefore, the only birds that barrier effects concern for the purposes of assessment are those whose flights lie in the direction of the offshore site and for which the intended destination is beyond the offshore site. The mean foraging range of lesser black-backed gull is 71.9 km and the mean maximum foraging range is 141 km (Thaxter *et al*., 2012). The far edge of the potential barrier that would be formed by the development lies approximately 25 km from the Isle of May and 42 km from Craigleith, thus the majority of flights in the direction of the development would potentially be affected.

For the Isle of May colony, the proposed wind farm would present a barrier 17.9 km wide and located 16 km to the NE. This barrier would potentially block approximately 33% of the possible flight directions available to lesser black-backed gulls flying out to distances in excess of 16 km from the Isle of May (Table 3.6). If 25% of birds reaching the barrier respond by detouring around the wind farm, this suggests that approximately 8% of flights from this colony would be affected. On the assumption that the destinations of affected flights lies on average 70 km from the breeding colony (the mean foraging distance is 71.9 km, Thaxter *et al.,* 2012), the mean increase in the length of barrier-affected flights is estimated at 4.7% (Table 3.7).

For birds breeding on Craigleith, the wind farm acting as a barrier would potentially block approximately 28% of the possible flight directions (Table 3.6). If 25% of birds reaching the barrier respond by detouring around the wind farm, this suggests that approximately 7% of flights from this colony would be affected. Assuming a mean destination distance of 70 km, the mean increase in the length of barrier-affected flights is estimated at 3.7% (Table 3.7).

The potential impact of the wind farm to act as a barrier and increase the length and duration of foraging trips for bird of the regional population in the breeding season is an effect that is negligible in magnitude and temporally long-term and reversible (Table 3.1 and

Table 3.2). It is concluded that the impact of any barrier effect on the regional lesser black-backed gull population in the breeding season is ***not significant*** under the terms of the Electricity Regulations

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

Lesser black-backed gulls are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional population of lesser black-backed gulls in the breeding season is ***not significant*** under the EIA Regulations.

Collision mortality

For the purposes of assessment it is assumed that there will be no far-field avoidance of the proposed wind farm by lesser black-backed gulls. Therefore, the mortality that is considered likely to occur for wind farm designs and assumed avoidance rate scenarios illustrated is that presented in Table 4.40 and Table 4.41 (see Appendix 12.2 for further details). It is assumed that all birds present during the breeding season are from the regional breeding population. The most recent information indicates that the regional breeding population is 22,035 adults. Approximately 1,983 adults die each year as a consequence of baseline mortality, calculated as 9.0% per annum (BTO, 2012).

COWRIE guidance (Maclean *et al.,* 2009) is to use a default avoidance rate of 99.5 % for gull species. There is no specific SNH guidance on avoidance rates for seabirds, and therefore the default value of 98.0% recommended by SNH is applicable.

An estimated 13.9% of flying lesser black-backed gulls observed in baseline surveys were within rotor height (Table 4.40).

Table 4.40 Proportion of lesser black-backed gull estimated to be at rotor height and the number of collisions predicted by collision rate modelling (CRM) using avoidance rates (AR) of 98.0% and 99.5% for least adverse and most adverse wind farm designs evaluated. LAT is sea level at lowest astronomical tide, and this is approximately 2.0 m below the average sea level.

Metric	Design 1, 64 x 7MW		Design 4, 128 x 3.6 MW	
	98.0% AR	99.5% AR	98.0% AR	99.5% AR
Rotor swept height range (metres above LAT)	24 - 188		24 - 144	
Flight activity at rotor height	13.9%		13.9%	
Flight activity below rotor height (<22.5 m above sea)	86.1%		86.1%	
Collisions in breeding season (April to August), all ages	4.5	1.1	6.8	1.7

Using avoidance rates of 98.0% and 99.5%, the increase in the adult mortality rate of the regional population would be <0.1% (Table 4.41). In both cases the changes are well below the default guidance threshold of 1%, which is considered the minimum change likely to have potential for adverse impacts on a population (King *et al.,* 2009).

Table 4.41 The effect of CRM predicted collision mortality on the adult mortality rate (AMR) of the regional population of lesser black-backed gulls in the breeding season. Results are presented for least adverse (64 x 7MW turbines) and most adverse (128 x 3.6MW turbines) wind farm designs evaluated, and for two values of overall avoidance rate (OAR)

Wind farm Design Period of year	% birds present assumed to be from regional breeding popn	98.0% OAR		99.5% OAR	
		CRM deaths	Change AMR	CRM deaths	Change AMR
Design 1, 64 x 7MW turbines Whole year	100%	4	0.02%	1	<0.01%
Design 4, 128 x 3.6MW turbines Whole year	100%	6	0.03%	1	0.01%

The potential impact of the predicted collision mortality on the regional population of lesser black-backed gulls in the breeding season is an effect of negligible magnitude (<1%) and temporally long-term and reversible. It is concluded that the effects of collision mortality on the regional population of lesser black-backed gulls in the breeding season are ***not significant*** under the terms of the EIA Regulations.

EIA Decommissioning Phase assessment for lesser black-backed gull

The decommissioning phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration.

Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

Lesser black-backed gulls are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during decommissioning operations on the regional population of lesser black-backed gulls in the breeding season is ***not significant*** under the EIA Regulations.

EIA summary of effects combined for lesser black-backed gull

The adverse impacts of the effects assessed will act in a broadly additive manner. In combination it is judged that the magnitude of the three effects on the population is negligible. Furthermore, the population has low sensitivity to all effects. It is concluded that the overall impact on the population is ***not significant*** under the EIA regulations (Table 4.42).

Table 4.42 Summary of effects on the regional population of lesser black-backed gulls in the breeding season

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Low	***Not significant***
Barrier Effect	Negligible	Long term	Low	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality	Negligible	Long term	Low	***Not significant***
All effects combined	Negligible	Long term	Low	***Not significant***

4.4.21.4 Cumulative Impact Assessment for lesser black-backed gull

Combining the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts on the regional breeding population of lesser black-backed gull will be as shown in Table 4.43. The individual impacts for displacement and collision are combined by addition to give the cumulative impact. Flight directions from the various colonies that are potentially affected by each wind farm acting as a barrier substantially overlap and therefore the cumulative barrier impact from the three developments is not the sum of the individual predicted impacts. The cumulative barrier impact is derived from the overall spread of flight directions at each colony potentially affected by the three developments forming barriers.

The predicted potential effective loss of <1.0% of the foraging habitat of the regional breeding lesser black-backed gull population (Table 4.43) is rated as an effect of negligible magnitude, temporally long-term and reversible. It is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional population of lesser black-backed gulls in the breeding season is ***not significant*** under the terms of the EIA Regulations.

Barrier effects as assessed here concern birds which would otherwise fly through the offshore site to access feeding resources beyond it. The mean foraging distance of lesser black-backed gull is 71.9 km and the mean maximum distance is 141 km (Thaxter *et al.*, 2012). This means that approximately 50% of lesser black-backed gull flights will be to areas beyond the barrier formed by the three proposed wind farms.

Table 4.43 Summary of CIA for the three proposed offshore wind farms in south-east Scotland on the regional population of breeding lesser black-backed gull

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF	Predicted impact from Inch Cape pOWF	Predicted impact from Firth of Forth Round 3 Zone pOWF	Cumulative Impact
Displacement: effective loss of marine foraging habitat during the breeding period (%)	25%	<0.1%, Negligible	<0.1%, Negligible	0.8%, Negligible	<1.0%, Negligible
Barrier Effect:	25%	Foraging trips of ca. 8% of Isle of May & Craigleith birds increase by ca.5%. Negligible	Foraging trips of ca. 5% of Isle of May & Craigleith birds increase by ca. 5%. Negligible	Foraging trips of ca. 5% of Isle of May & Craigleith birds increase by ca. 20%. Low	Foraging trips ca. 5% of Isle of May & Craigleith birds (ca. 3% of regional population) increase by 20%. Low
Collision most	0%	0.03%,	43.9%	Approx. 0.6%	44.5%

adverse designs: % increase in assumed annual adult mortality rate based on 98.0% avoidance rate.		negligible	Severe	negligible (based on NNG scaled by abundance)	moderate
Collision least adverse designs: % increase in assumed annual adult mortality rate based on 98.0% avoidance rate.	0%	0.02%, negligible	2.7% Low	Approx. 0.4%, negligible (based on NNG scaled by abundance)	3.1% low
Collision most adverse designs: % increase in assumed annual adult mortality rate based on 99.5% avoidance rate.	0%	<0.01%, negligible	11.0% Moderate	Approx. 0.6% negligible (based on NNG scaled by abundance)	11.6% moderate
Collision least adverse designs: % increase in assumed annual adult mortality rate based on 99.5% avoidance rate.	0%	<0.01%, negligible	0.7% Negligible	Approx. 0.4%, negligible (based on NNG scaled by abundance)	1.1% low

It is estimated that about 5% of foraging flights of breeding birds from the Isle of May and Craigleith (ca. 4% of the regional total) would be affected by the three wind farms acting as barriers. Affected flights would undergo detours that would on average amount to an increase in length of approximately 20%, compared to direct flights to the same destination beyond the wind farm. This estimate assumes, on the basis of observations at operational wind farms, that only 25% of birds reaching the barrier will respond by detouring around the wind farms. The cumulative barrier effect is rated as low magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of a barrier effect caused by the three proposed offshore wind farms on the regional population of lesser black-backed gulls in the breeding season is of ***minor significance*** under the terms of the EIA Regulations.

The cumulative assessment of barrier effects above assumes that all areas of the wind farm lease sites will be developed. In the case of the Firth of Forth Round 3 Zone there will be large areas without turbines and the eventual design may therefore effectively present birds several smaller barriers with gaps between through which birds can pass, rather than the single large barrier assumed above. Were this to be the case, the magnitude of the cumulative barrier effect could be substantially less than suggested above.

The Year 1 report of survey work for the Firth of Forth Round 3 Zone (SWEL, 2011) does not give collision mortality estimates for lesser back-backed gull despite moderate numbers being recorded there. The estimated numbers of lesser black-backed gulls in the Firth of Forth Round 3 Zone development sites (Phase 1, 2 and 3) was approximately twenty times greater than that recorded in the Neart na Gaoithe offshore site. On this basis, for the purpose of this CIA, it is assumed that collision mortality caused by the Firth of Forth Round 3 Zone development will be twenty times greater than the estimates for Neart na Gaoithe.

Conclusions on the cumulative impact of collision mortality are sensitive to the wind farm designs evaluated and the level of avoidance rate used for predictive calculations of the number of collision strikes.

Using an avoidance rate of 98.0% the cumulative effect of the three wind farms is to increase the annual adult mortality rate by 3.1% for the least adverse wind farm designs and by 44.5% for the most adverse designs. Assuming the least adverse designs would be chosen, the potential cumulative impact of this additional mortality on the regional breeding lesser black-backed gull population is rated as moderate magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that, when assessed using a 98.0% avoidance rate, the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional population of lesser black-backed gull in the breeding season is of ***moderate significance*** under the terms of the EIA Regulations.

Using an avoidance rate of 99.5% the cumulative effect of the three wind farms is to increase the annual adult mortality rate by 1.1% for the least adverse wind farm designs, and by 11.6% for the most adverse designs. Assuming the least adverse designs would be chosen, the potential cumulative impact of this additional mortality on the regional breeding lesser black-backed gull population is rated as low magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that, when assessed using a 99.5% avoidance rate, the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional population of lesser black-backed gull in the breeding season is of ***minor significance*** under the terms of the EIA Regulations.

10.3.1.5 Mitigation measures for lesser black-backed gull

The assessment does not identify any significant adverse effects on the regional population of lesser black-backed gulls in the breeding season. Therefore no mitigation measures are required for this species.

4.4.22 Herring gull *Larus argentatus*

4.4.22.1 Status

Herring gulls are resident, common and widespread, breeding in colonies in coastal and inland locations. There is a general movement southwards in winter months (Forrester et al 2007). Seabird 2000 recorded 142,942 breeding pairs in Britain (Mitchell *et al.,* 2004). The closest large breeding colonies to the Neart na Gaoithe development are on the islands in the Firth of Forth and the Isle of May. Herring gulls exploit a wide range of food sources, including scraps and offal from trawlers, as well as on land at refuse dumps and farm land (Forrester *et al.,* 2007).

4.4.22.2 3.7.26.2 Neart na Gaoithe Study Area

A total of 1,723 herring gulls were recorded on surveys in the Neart na Gaoithe study area in Year 1, however only 50 birds were recorded in the offshore site (Table 4.2). The majority of birds (97.1%) were recorded in the buffer area between November and January, with a peak of 644 birds in January. In Year 2, a total of 1,433 birds were recorded on surveys, however only 58 birds were seen in the offshore site (Table 4.2). The majority of birds (96.0%) were recorded in the buffer area between December and April, with a peak of 714 birds in April.

During the Year 1 breeding season (April to August), the peak estimated number of herring gulls in the offshore site occurred in June (14 birds), with an estimated 484 birds in the buffer area in April (Table 4.44) (Figure 4.50). In the Year 2 breeding season, peak estimated numbers of herring gulls in the offshore site were recorded in July (48 birds), with an estimated 2,015 birds in the buffer area in April.

In the non-breeding season (September to March), the peak estimated number of herring gulls in the offshore site occurred in March of Year 1 (39 birds) (Table 4.44) (Figure 4.50). In the buffer area, estimated numbers peaked at 9,061 birds in January, exceeding the 1% thresholds of national (4,500 birds) and international importance (5,900 birds) (Holt *et al.,* 2011). However, this estimate (and the November figure) was probably inflated by the presence of fishing vessels in the study area with large numbers of herring gulls associating with them, and should therefore be treated with caution as it may not reflect typical conditions.

Table 4.44 Estimated numbers of herring gulls in the offshore site and 8 km buffer in Years 1 and 2

Month	Estimated nos on water Offshore Site	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying Offshore Site	Estimated total Offshore Site	Estimated total 8 km buffer	Estimated total
Yr1 Nov	0	0	0	20	20	3,452	3,472
Yr1 Dec	0	0	0	27	27	921	948
Yr1 Jan	0	0	0	20	20	9,061	9,081
Yr1 Feb	16	4	62	7	23	130	153
Yr1 Mar	12	4	43	27	39	59	98
Yr1 Apr	0	0	0	7	7	484	491
Yr1 May	0	0	0	0	0	14	14
Yr1 Jun	7	2	23	7	14	260	274
Yr1 Jul	0	0	0	0	0	7	7
Yr1 Aug	0	0	0	0	0	37	37
Yr1 Sep	0	0	0	0	0	7	7
Yr1 Oct	0	0	0	0	0	19	19
Yr2 Nov	-	-	-	-	-	-	-
Yr2 Dec	1	0	4	7	8	268	276
Yr2 Jan	0	0	0	41	41	537	578
Yr2 Feb	0	0	0	14	14	118	132
Yr2 Mar	2	0	12	13	15	111	126
Yr2 Apr	0	0	0	0	0	2,015	2,015
Yr2 May	0	0	0	0	0	13	13
Yr2 Jun	0	0	0	20	20	115	135
Yr2 Jul	48	8	281	0	48	175	223
Yr2 Aug	0	0	0	0	0	7	7
Yr2 Sep	0	0	0	0	0	27	27
Yr2 Oct	0	0	0	0	0	0	0

In Year 2, peak estimated numbers of herring gulls also occurred in January although numbers were much lower, with an estimated 41 birds in the offshore site, and 587 birds in the buffer area (Table 4.44) (Figure 4.50).

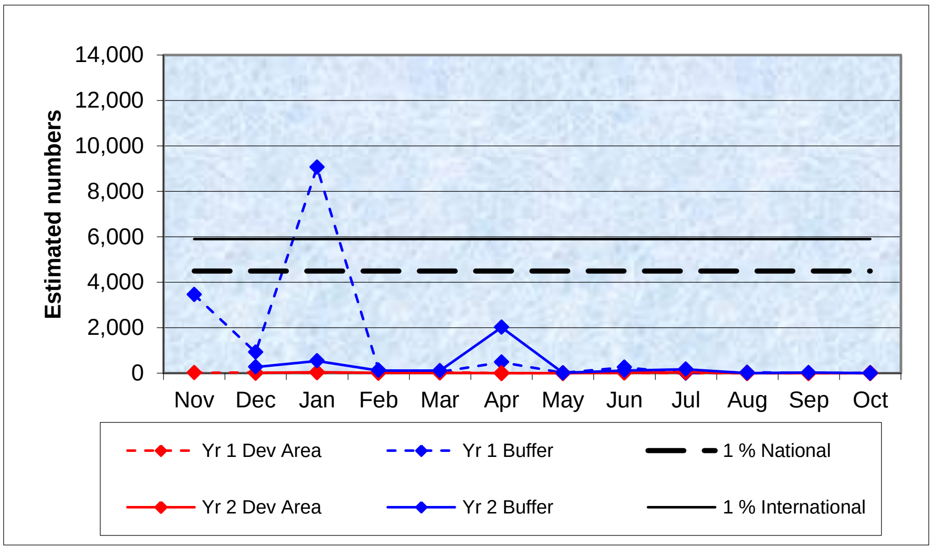


Figure 4.50 Monthly estimated numbers of herring gulls in the Neart na Gaoithe Development & buffer areas in Years 1 and 2

Mean monthly herring gull density in the offshore site was very low throughout both Years 1 and 2 (Figure 4.51). In the buffer area, mean density of herring gulls peaked in November and January of Year 1, and in April of Year 2, most likely connected with fishing vessel activity in the area. ESAS mean density data for the surrounding ICES rectangles and Regional Sea 1 matched this pattern of greater winter activity, with peaks in December, and low mean densities recorded in the breeding season.

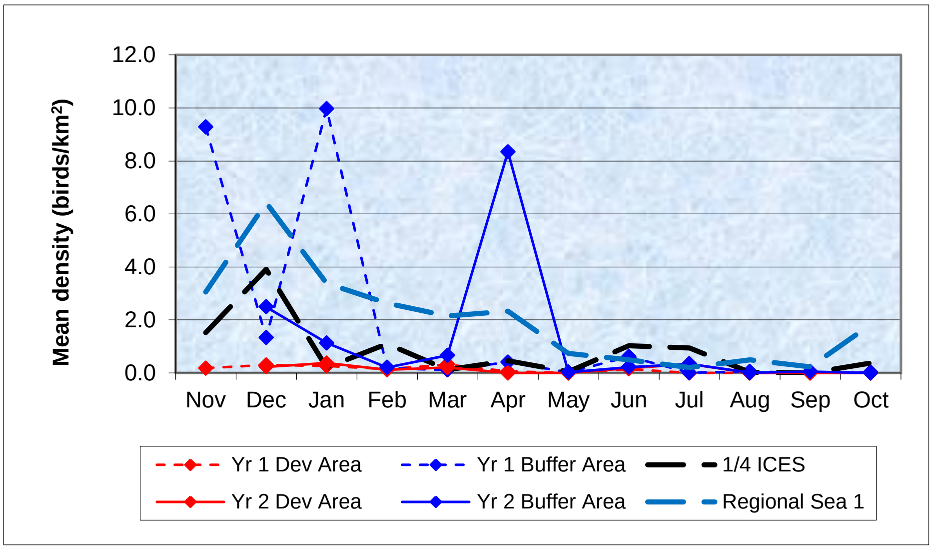


Figure 4.51 Comparison of monthly mean densities for herring gull in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

In the Year 1 non-breeding period (November to March, September and October), highest densities of herring gulls were recorded in the south of the buffer area (Figure 4.52). Densities in the offshore site were low at this time of year. Fewer birds were recorded in the north and east of the study area.

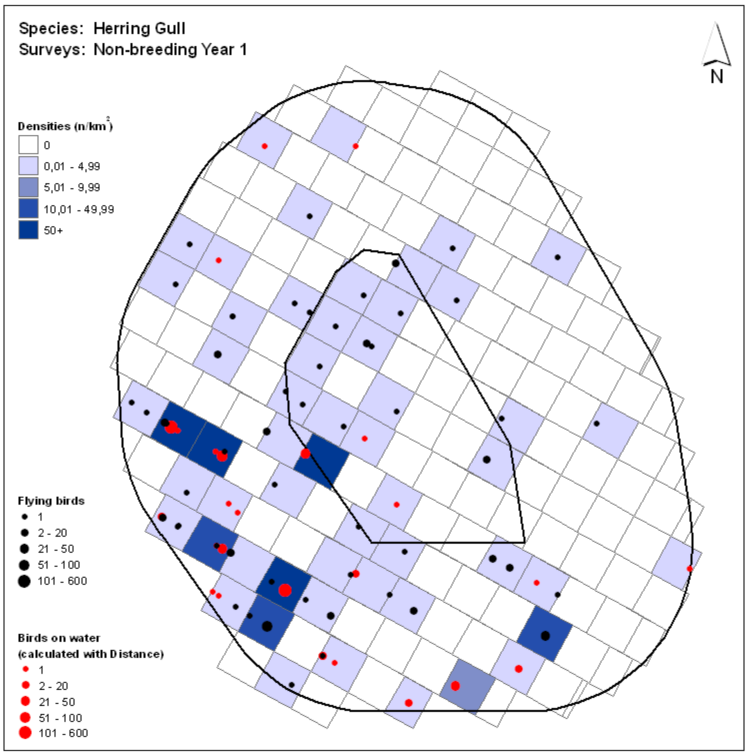


Figure 4.52 Herring gull density in the non-breeding season, Year 1

A similar distribution pattern was recorded in the Year 2 non-breeding period, with generally low densities of herring gulls recorded in the southern half of the study area and fewer birds in the north (Figure 4.53). As in Year 1, herring gull densities in the offshore site at this time were low.

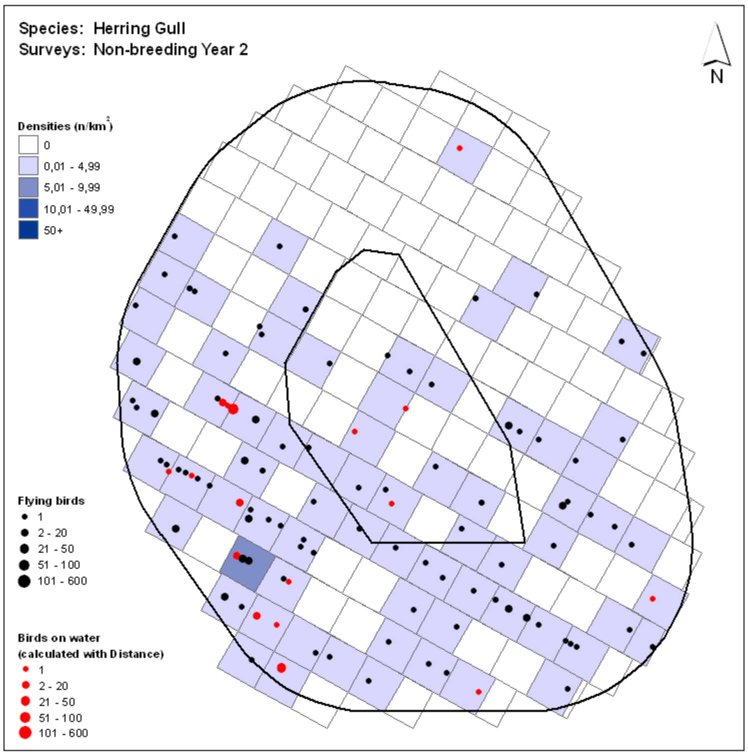


Figure 4.53 Herring gull density in the non-breeding season, Year 2

Herring gull density in the Year 1 breeding season (April to August) was very low in the offshore site, with few birds recorded at over the period (Figure 4.54). Herring gulls were slightly more widespread in the buffer area at this time, with highest density recorded in the south of the buffer area.

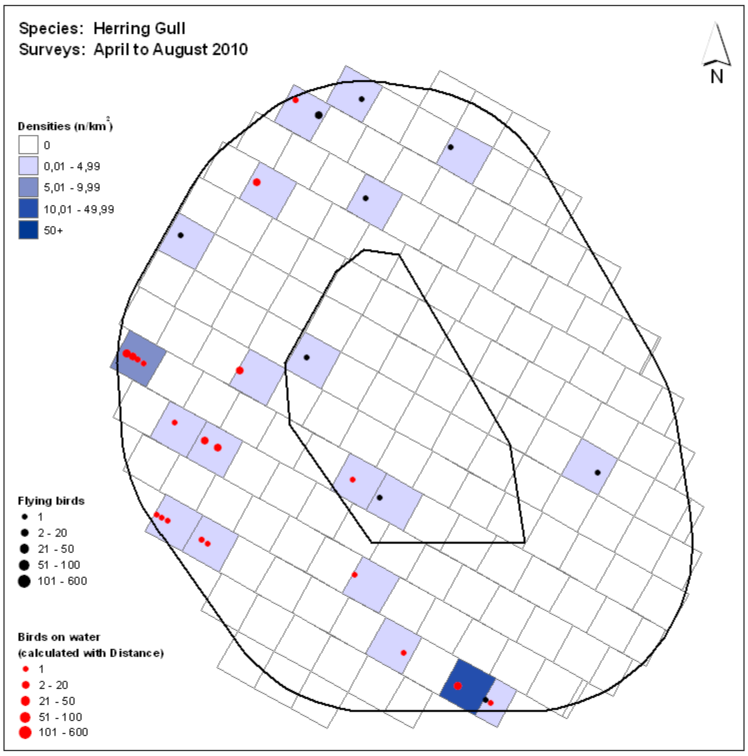


Figure 4.54 Herring gull density in the breeding season, Year 1

Herring gull distribution was similar between April and August of Year 2 and the same period in Year 1, with low densities and few birds recorded within the offshore site at this time. Highest densities were recorded in the west of the buffer area at this time (Figure 4.55).

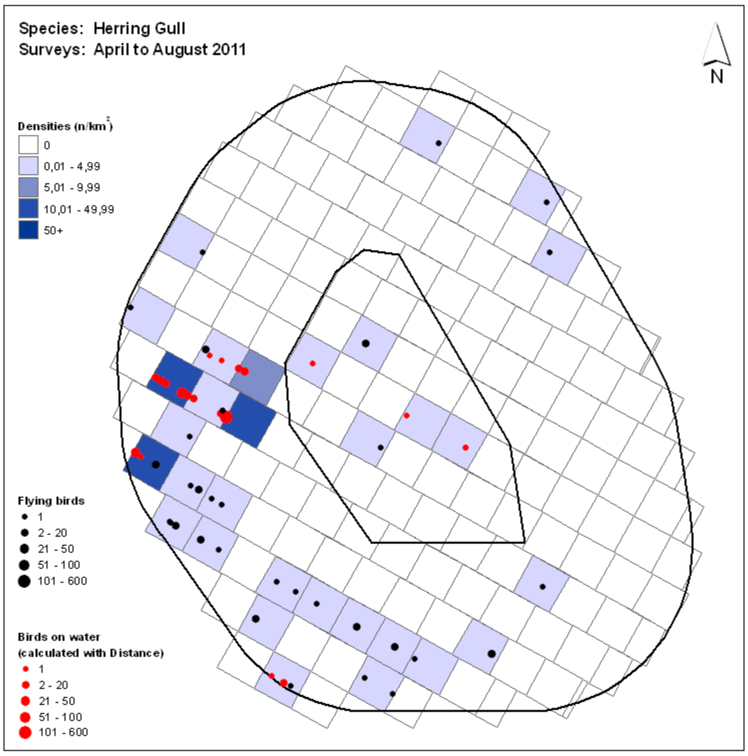


Figure 4.55 Herring gull density in the breeding season, Year 2

In Years 1 and 2, a total of 1,284 herring gulls were recorded in flight, with 70.0% of birds flying below 22.5 m (Table 4.3). A total of 383 birds (30.0%) were recorded flying above 22.5 m, i.e. within the rotor swept zone, at estimated heights of between 25 m and 60 m.

Foraging behaviour was recorded for 953 herring gulls in the Neart na Gaoithe study area in Years 1 and 2, with four types of foraging behaviour recorded, and unspecified feeding behaviour recorded for a further 21 birds (Table 4.45). Scavenging at fishing vessels was the most frequently recorded foraging behaviour (92.8%).

Table 4.45 Herring gull foraging behaviour in the Neart na Gaoithe Study Area in Years 1 and 2

Behaviour	Number of birds
Actively searching	45
Dipping	1
Scavenging	904
Surface pecking	3
Feeding method unspecified	21
Total	**974**

4.4.22.3 Species sensitivity

Herring gull is currently red-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009).

A recent review assessed herring gull as being at moderate risk of collision. Displacement, barrier effects, habitat loss or changes in prey distribution resulting from offshore wind farms were rated as low risk. Overall, herring gull was assessed as being at moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

Herring gull is listed as a qualifying interest species in the breeding season for four SPAs on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development (Table 4.46). These SPAs held 9.5% of the UK breeding population and 1.6% of the biogeographic population at the time of designation (JNCC, 2012). Since designation, the populations at these SPAs have decreased (SMP, 2012). The distance between the offshore site and two SPAs (Forth Islands SPA and St Abb's Head to Fast Castle SPA) is within the mean maximum foraging range (61.1 km), while the distance to Fowlsheugh SPA lies just outside this range, but is within the maximum known foraging range of this species (92 km) (Thaxter *et al.*, 2012). Herring gull mean maximum foraging range from breeding SPAs in relation to the offshore site are shown in Figure 4.56.

Table 4.46 SPAs for breeding herring gulls between Peterhead and Blyth

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count [2]	Year
Buchan Ness to Collieston Coast	113	4,292	0.5	2.7	3,114	2007
Forth Islands	**16**	**6,600**	**0.7**	**4.1**	**3,934**	**2006**
Fowlsheugh	*62*	*3,190*	*0.3*	*2.0*	*214*	*2009*
St Abb's Head to Fast Castle	**31**	**1,160**	**0.1**	**0.7**	**258**	**2008**
Total	**-**	**15,242**	**1.6**	**9.5**	**7,520**	**-**

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. Sites in italics lie within the maximum known foraging range of 92 km. Sites in bold lie within the mean maximum foraging range of 61.1 km (Thaxter *et al.* 2012).

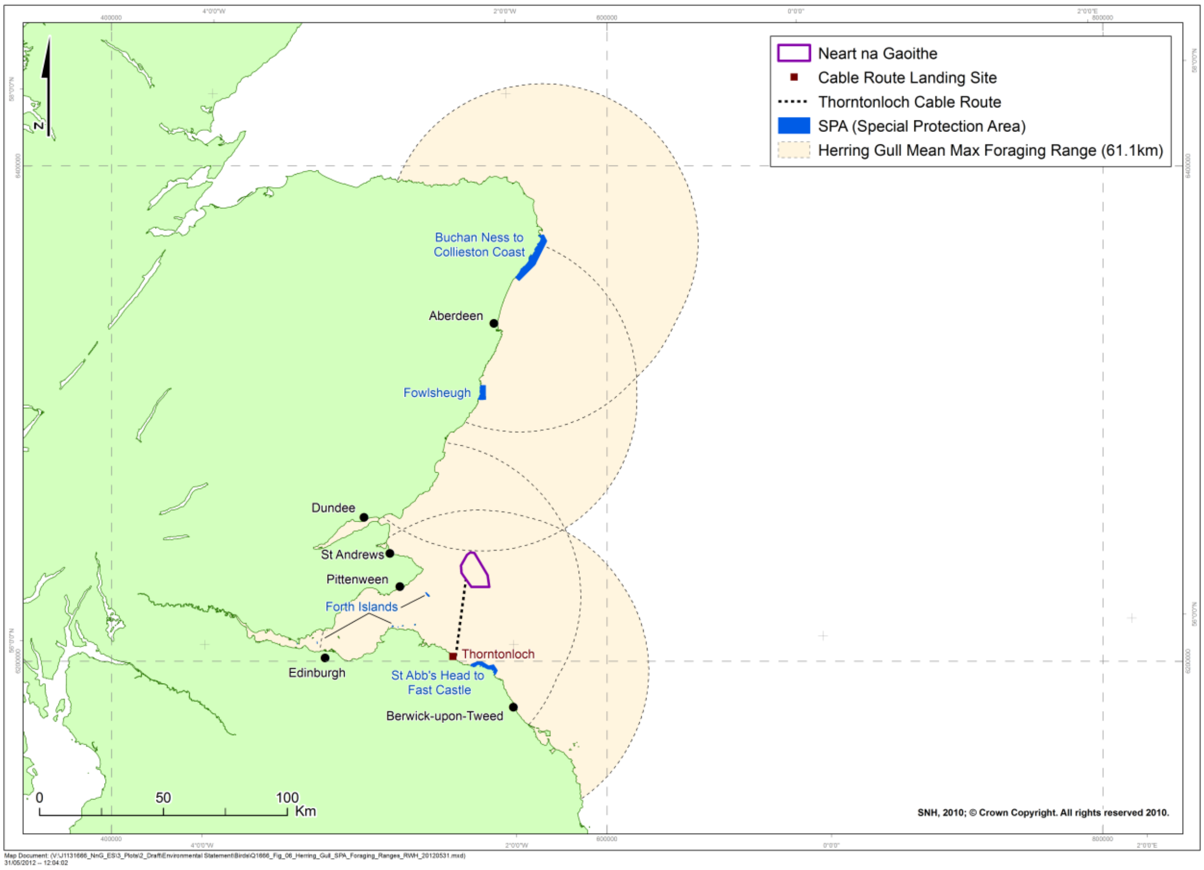


Figure 4.56 Herring gull mean maximum foraging range from breeding SPAs in relation to the Development

4.4.22.4 Assessment

Definition of seasons

The annual cycle for herring gull is divided into two parts to reflect the biology of the species and the broad pattern of use of the offshore site. The main breeding season, when breeding adults are attending colonies, is defined as April to August. At this time the vast majority of birds present in the offshore site will be from relatively local breeding colonies (e.g., colonies that are closer to the offshore site than the mean maximum foraging distance).

The non-breeding period is defined as September to March and broadly corresponds to the period when herring gulls are in their over-wintering area. In this period, especially after November, it is likely that a high proportion of individuals present in the offshore site are from breeding sites outwith the region, including birds from Scandinavia (Wernham *et al*., 2002), although birds from the regional breeding population will also be present.

Baseline conditions

In the breeding season (April to August) the mean estimated number of herring gulls present in the offshore site was 10, and the mean estimated number in the offshore site buffered to 1 km was also 10. On average 60% of birds present in the breeding season in the offshore site were on the sea and the remainder were in flight.

Based on survey results for the whole Survey Area in this period, 78.3% of herring gulls were in adult plumage and the remainder were in immature plumage. For the purposes of assessment it is assumed that all birds in adult plumage were breeding birds and that the mean colony attendance rate was 50%.

In the non-breeding period (November to March) the mean number of herring gulls present in the offshore site was 18, and the mean number in the offshore site buffered to 1 km was 240 birds.

Populations

The breeding population of herring gulls in Scotland has undergone a prolonged period of decline and recent colony counts indicate that the decline is on-going, though recent declines in the south-east Scotland have been modest (SMP, 2012). Assuming that the sub-set of colonies that have been recently counted is representative, the regional decline since the Seabird 2000 counts amounts to approximately -6%. The decline in breeding numbers mean that published figures on population size for this species (e.g., Mitchell *et al*., 2004) no longer accurately reflect the current regional population size. To prevent this causing assessment of effects to be biased low, the regional total derived from Seabird 2000 results has been adjusted downwards by 6%. On this basis, the regional breeding population is assumed to be 43,302 breeding adults (i.e., 21,651 breeding pairs).

The size of the non-breeding-period regional herring gull population is assumed to be 200,000 birds. This is derived by summing the November to February period estimates for localities 5 and 6 and half the estimates for localities 17, 18 and 19 given in Skov *et al.* (1995). The results presented by Skov *et al.* (1995) are for birds in marine habitats only and the estimate is therefore likely to be an underestimate as very large numbers of herring gulls also use terrestrial habitats such as agricultural fields and refuse tips at this time of year.

A large proportion of the regional breeding population are thought to stay in the region through the non-breeding period but these are joined by many more birds from other breeding areas (Wernham *et al.,* 2002).

Nature conservation importance

Herring gull is rated as high NCI because it is on the BoCC Red List and is a UK BAP species (Eaton *et al.,* 2009). In addition, a high proportion of the birds occurring in the offshore site buffered to 1 km are likely to be from SPA breeding populations.

Offshore wind farm studies of herring gull

Results from bird monitoring at operational wind farms indicate that a small proportion of herring gulls may be displaced from offshore wind farms, however studies typically show either no significant change or an increase in abundance of herring gulls at operational wind farms compared to pre-construction numbers (i.e. an attraction effect). At Horns Rev, Denmark, herring gulls occurred less frequently in the wind farm compared with the wider survey area during the pre-construction period, but were more abundant there during operational, and, especially, in the construction phase. This shift in abundance was attributed to the attractive effect of ship traffic and availability of perches. (Diersche and Garthe, 2006, Petersen *et al*., 2004, Christensen *et al*., 2003). The authors, in their final report interpret the overall response of herring gulls to the Horns Rev wind farm as “a slight (but not statistically significant) increase in use of the general area and surroundings” (Petersen *et al*., 2006). Conversely, at Nysted, Denmark, herring gulls showed the strongest avoidance of the wind farm during the operational phase. At Egmond aan Zee, the Netherlands, herring gulls showed statistically significant avoidance of the wind farm in three surveys, attraction in one survey and non-significant results in a further 10 surveys. These results, combined with an analysis to assess the influence of fishing vessels on herring gull distribution led the authors to conclude that the Egmond aan Zee wind farm had hardly any effect on herring gull distribution, but that fishing vessel distribution had a major effect on herring gull distribution (Leopold *et al*., 2011). At North Hoyle, Wales, reports state that there was no evidence of displacement or a barrier effect for herring gull (RWE Group, PMSS, 2007).

Visual and radar studies suggest that operational wind farms present only a partial barrier to herring gulls, with birds regularly flying amongst turbines. At both Horn Rev, and Egmond aan Zee, herring gulls were regularly seen flying within the wind farm. At Horns Rev, Denmark, visual monitoring from an observation platform positioned at the edge of the wind farm found that 37% (sample size not given) of flying herring gulls were either within or flying into the wind farm (Diersche and Garthe, 2006).

Summarising the barrier effect of wind farms on seabird species in German marine areas, herring gulls were categorised as commonly flying through wind farms (Diersche and Garthe, 2006). Behavioural studies at the coastal wind farm at Zeebrugge Harbour, Belgium, reported that 9%, 38% and 41% of herring gulls flying below, at, and above turbine height respectively showed an avoidance reaction to the turbines, however no strong barrier effect was apparent because most birds soon passed through the wind farm (Everaert, 2003). At two turbine arrays at Maasvlakte, the Netherlands, sited between breeding gull colonies (including herring gull) and their offshore foraging areas, 3.1% of 751 gulls recorded showed a behavioural reaction to the turbines, but only one bird was recorded to turn back. The authors highlight the contrast between this result and the strong avoidance responses by gulls observed outside the breeding period at Maasvlkte, explaining the former as rapid habituation to the turbines during the breeding season or reduced sensitivity to the turbines by the breeding birds (van den Bergh *et al*., 2002). Studies of wind farms as barriers to migration or regular bird flights, reviewed by Hötker *et al*. (2006) identified three studies where wind farms were, and three where they were not concluded to act as barriers to herring gulls; no other details were given.

The risk of herring gull colliding with wind turbines is likely to be low to moderate based on reported flying height and recorded fatalities from operational wind farms. At the turbine arrays at Maasvlakte *ca.* 20% and 50% of herring gulls passed through at rotor height (van den Bergh *et al*., 2002). Of 44 flying herring gulls recorded over two years of post-construction monitoring at the Arklow Bank wind farm and the associated 'bank' area, 12% of birds were recorded flying at a height greater than 20 m above the sea surface (Barton *et al*., 2009, Barton *et al*., 2010). During two years monitoring at North Hoyle, Wales, a total of 100 (19%) of 539 flying herring gulls were recorded higher than 20 m above the sea surface (RWE Group, PMSS, 2006, PMSS, 2007). At Blyth Harbour, 13% of herring gulls crossed the wind farm at rotor height (Diersche and Garthe, 2006). Results from the Zeebrugge Harbour coastal wind farm reported 25% of 136 herring gulls flying at rotor height (16-50 m). In this study the "day and night collision chance" for herring gull, based on the number of collision victims and the number of locally migrating birds, was 1:750, for birds flying at turbine height (Everaert, 2003). The review of offshore wind farm effects on birds (Diersche and Garthe, 2006) highlighted the relatively high numbers of herring gull fatalities reported at coastal wind farms, while 189 herring gull fatalities were reported in a review of the number of collision victims at wind farms in eight European countries (Hötker *et al*., 2006). One study found no signs that herring gulls habituated to wind farms; but no details were given (Hötker *et al*., 2006).

For the purpose of assessing collision mortality it is assumed that no herring gulls will be displaced from the wind farm. A cautious approach is merited because during times of gales, fog and at night herring gull avoidance behaviour (either far-field adjustments to course or last moment evasion) could be less effective than is assumed.

EIA Construction Phase assessment for herring gull

The construction phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

Herring gulls are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during construction operations on the populations of herring gulls in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

EIA Operational Phase assessment for herring gull

Potential for herring gull to be affected by displacement

For the purpose of assessing displacement it is assumed that 25% of herring gulls will be displaced from the proposed wind farm footprint and a surrounding buffer of 1 km. Based on evidence from existing wind farms this is likely to be a cautious assumption because reported declines in herring gull abundance at existing wind farm are below 25% (e.g. Nysted, Denmark), indeed, some sites showed no significant change, while increased numbers were recorded at other sites during the operational phase (e.g. Horns Rev, Egmond aan Zee).

Breeding season

The value of the offshore site buffered to 1 km as a foraging site to herring gulls in the breeding season was estimated from the proportion of the adults likely to be at sea (not attending a colony) that were on average present (Table 4.47). This method is likely to overestimate the actual importance of the area for foraging (and thus leads to cautious assessment conclusions) because it is likely that some birds seen in the baseline surveys were not foraging in the area but merely flying through it.

On this basis, it is inferred that the offshore site buffered to 1 km provides up to 0.04% of the marine foraging resources of the regional breeding population (Table 4.47).

Non-breeding season

The value of the offshore site buffered to 1 km as a foraging site to herring gulls in the non-breeding season was estimated from the proportion of birds that were on average present (Table 4.47). On this basis it is inferred that the offshore site buffered to 1 km provides up to 0.1 % of the marine foraging resources of the regional non-breeding population (Table 4.47).

Table 4.47 The mean estimated number of herring gull present during the breeding period and the non-breeding period, and the value as a percentage of the (at-sea) receptor population potentially at risk of displacement

Receptor population	Population size (adults)	No. assumed at sea	Dev.Area		Dev.Area + 1km	
			Mean no. at risk	%	Mean no. at risk	%
National, breeding period (Seabird 2000)	282,613	141,307	10	<0.1	10	<0.1
North Sea, non-breeding period (Skov *et al*., 1995)	971,700	971,700	18	<0.1	240	<0.1
Regional, breeding period (Seabird 2000 x 6% decline)	43,302	21,651	10	<0.1	10	<0.1
Regional, non-breeding period (Skov *et al*., 1995)	200,000	200,000	18	<0.1	240	0.1

Likely impacts of displacement on herring gull populations

Studies of herring gulls show that this species exhibits little, if any, displacement behaviour in response to operational wind farms.

If 25% of herring gulls were displaced during the breeding season from the offshore site buffered to 1 km the impact of this would be the effective loss of <0.1% of the foraging habitat of the regional breeding population (25% of the value in Table 4.47). This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of displacement on the regional herring gull population in the breeding period are ***not significant*** under the EIA Regulations.

If 25% of herring gulls were displaced during the non-breeding period from the offshore site buffered to 1 km the impact of this would be the effective loss of <0.1 % of the foraging habitat of the non-breeding-period population (25% of the value in Table 4.47). This impact is categorised as negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of displacement on the regional herring gull population in the non-breeding period are ***not significant*** under the EIA Regulations.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

Herring gulls are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional populations of herring gulls in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

Barrier effect

Herring gulls breeding at nearby colonies to the proposed wind farm could be affected by the wind farm acting as a barrier. The greatest potential for such impacts is for birds from the Isle of May, Craigleith and St Abb's Head colonies.

Herring gulls are considered to have low sensitivity to barrier effects (Maclean *et al*, 2009, Langston, 2010). The potential effects on herring gull of the proposed wind farm acting as a barrier are assessed for the breeding season, when birds are attending colonies.

For the purposes of assessment the width of the barrier is assumed to extend 1 km either side of the maximum width of the proposed wind farm. Observations from operational offshore wind farms shows no evidence that wind farms pose a barrier to herring gulls. It is therefore likely that only a small percentage of foraging flights potentially intercepted by the barrier would be affected. It is assumed that only 25% of birds reaching the barrier will respond by detouring around the wind farm. Barrier effects as calculated here concern birds which would otherwise fly through the offshore site to access feeding resources beyond.

For birds breeding on the Isle of May colony, the wind farm acting as a barrier would potentially block approximately 33% of the possible flight directions (Table 3.6). Assuming a mean destination distance of 30 km (immediately beyond the barrier), the mean increase in the length of barrier-affected flights is estimated at 17.7% (Table 3.7). For birds breeding on Craigleith, the wind farm acting as a barrier would potentially block approximately 28% of the possible flight directions (Table 3.6). Assuming a mean destination distance of 45 km (immediately beyond the barrier), the mean increase in the length of barrier-affected flights is estimated at 8.7% (Table 3.7). For birds breeding at St Abb's Head, the wind farm acting as a barrier would potentially block approximately 9% of the possible flight directions (Table 3.6). Assuming a mean destination distance of 45 km (immediately beyond the barrier), the mean increase in the length of barrier-affected flights is estimated at 5.3 % (Table 3.7).

However, less than 50% of herring gull flights potentially affected by the barrier are likely to have an intended destination beyond the wind farm, as the mean distance of foraging flights is 10.5 km (Thaxter *et al*., 2012). In addition, only a minority (assumed to be 25%) of birds that do fly beyond the wind farm, are likely to respond by detouring around the wind farm. It is concluded that <5% of foraging flights from these colonies are likely to be affected by barrier effects and the size of detours is relatively small. The likely impacts arising from the proposed wind farm acting as a barrier to herring gulls breeding at other colonies were not examined in detail because, on the basis of the results for the Isle of May, Craigleith and St Abb's Head, the greater distances and the smaller number of birds involved, it is clear that any impact would be negligible and could not plausibly make barrier effects an issue of significance for the regional breeding population.

The potential for the development to act as a barrier and increase the length and duration of foraging trips for herring gulls of the regional population in the breeding season is an effect that is of low magnitude (<1 %) and temporally long-term and reversible (Table 3.1 and Table 3.2). Bearing in mind the low sensitivity of this species to barrier effects, it is concluded that any barrier effect on the regional breeding herring gull population is ***not significant*** under the terms of the Electricity Act.

Collision mortality

For the purposes of assessment it is assumed that there will be no far-field avoidance of the proposed wind farm by herring gulls. Therefore, the mortality that is considered likely to occur for wind farm designs and assumed avoidance rate scenarios illustrated is that presented in Table 4.48 and Table 4.49 (see Appendix 12.2 for further details). COWRIE guidance (Maclean *et al.,* 2009) is to use a default avoidance rate of 99.5% for gull species. There is no specific SNH guidance on avoidance rates for seabirds, and therefore the default value of 98.0% recommended by SNH is applicable.

30.0% of flying herring gulls observed in baseline surveys were within the height range swept by the proposed turbine rotors (Table 4.48).

During the breeding season it was assumed that all birds present were from the regional breeding population. In the non-breeding season the regional population comprises a mixture of birds originating from the regional breeding population and more distant colonies. Therefore, to estimate the effect of year-round collision mortality on the regional breeding population the proportion of collision deaths that occur in the non-breeding period involving birds from the regional breeding population needs to be estimated. Ringing and colour marking studies show that large numbers of herring gulls from the regional population (Peterhead to Blyth) overwinter outside the region (Wernham *et al*., 2002). These studies also show that large numbers of birds from northern Scotland and Scandinavian breeding grounds overwinter in the region (Wernham *et al*., 2002). The winter population estimate for the region (approximately 200,000 birds, Skov *et al*., 1995) also indicates that there is a large net influx of herring gull into the region in the winter, with around twice as many birds present in winter compared to the breeding season. Based on this evidence, for the purposes of assessment it is assumed that 25% of the adult collision deaths during the non-breeding period involve adults from the regional breeding population. The effect of year-round collision mortality on the annual adult mortality rate for the regional breeding population is calculated by summing the breeding season mortality and the non-breeding period mortality attributed to breeding birds from the region.

Table 4.48 Proportion of herring gull estimated to be at rotor height and the number of collisions predicted by collision rate modelling (CRM) using avoidance rates (AR) of 98.0% and 99.5% for most adverse and least adverse wind farm designs. LAT is sea level at lowest astronomical tide, and this is approximately 2.0 m below the average sea level

Metric	Design 1, 64 x 7MW		Design 4, 128 x 3.6 MW	
	98.0% AR	99.5% AR	98.0% AR	99.5% AR
Rotor swept height range (metres above LAT)	24 - 188		24 - 144	
Flight activity at rotor height	30.0%		30.0%	
Flight activity below rotor height (<22.5 m above sea)	70.0%		70.0%	
Collisions in breeding season (April to August), all ages	15.2	3.8	22.9	5.7
Collisions in non-breeding period (September to March), all ages	69.9	17.5	105.1	26.3
Total collisions per year, all ages	85.1	21.3	128.0	32.0

Table 4.49 The effect of CRM predicted collision mortality on the adult mortality rate (AMR) of the regional breeding herring gull population. Results are presented for least adverse (64 x 7MW turbines) and most adverse (128 x 3.6MW turbines) wind farm designs evaluated, and for two values of overall avoidance rate (OAR).

Wind farm Design Period of year	% birds present assumed to be from regional breeding popn	98.0% OAR		99.5% OAR	
		CRM deaths	Change AMR	CRM deaths	Change AMR
Design 1, 64 x 7MW turbines					
Breeding period (April - August)	100%	12	0.2%	3	0.1%
Non-breeding period (Sept. - March)	25%	12	0.2%	3	<0.1%
Whole year	varies	24	0.4%	6	0.1%
Design 4, 128 x 3.6MW turbines					
Breeding period (April - August)	100%	18	0.3%	5	0.1%
Non-breeding period (Sept. - March)	25%	17	0.3%	4	0.1%
Whole year	varies	35	0.6%	9	0.2%

Using an avoidance rate of 99.5%, the increase in the adult annual mortality rate of the regional population of herring gulls in the breeding season would be between 0.1% and 0.2% depending on wind farm design (Table 4.49). Using an avoidance rate of 98.0% the increase is between 0.4% and 0.6%. For both avoidance rates the changes in mortality rate are in all cases below the default guidance threshold of 1% of the baseline mortality rate, which is considered the minimum likely to have potential for adverse impacts on a population (King et al., 2009).

The potential impact of the predicted collision mortality on the regional breeding population is an effect of negligible magnitude (<1 %), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of collision mortality on the regional herring gull population in the breeding season are ***not significant*** under the terms of the EIA Regulations.

EIA Decommissioning Phase assessment for herring gull

The decommissioning phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

Herring gulls are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during decommissioning operations on the regional populations of herring gulls in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

EIA summary of effects combined for herring gull

The adverse impacts on the herring gull regional population in the breeding season of the effects assessed will act in a broadly additive manner. In combination, and using a collision avoidance rate of either 98.0% or 99.5%, it is judged that the combined magnitude of the three effects is negligible (Table 4.50). Furthermore, the population has low sensitivity to all effects except collision risk. It is concluded that the overall operational phase impact of the development on the regional population of herring gull in the breeding season is ***not significant*** under the EIA Regulations.

The adverse impacts on the regional population of herring gulls in the non-breeding period of the effects assessed will act in a broadly additive manner. In combination, and using a collision avoidance rate of either 98.0% or 99.5%, it is judged that the combined magnitude of the three effects is negligible (Table 4.51). It is concluded that the overall operational phase impact of the development on the regional herring gull population in the non-breeding period is ***not significant*** under the EIA Regulations.

Table 4.50 Summary of effects on the regional population of herring gulls in the breeding season

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Low	***Not significant***
Barrier Effect	Negligible	Long term	Low	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality (using either 98.0% or 99.5% A R)	Negligible	Long term	Moderate	***Not significant***
All effects combined	Negligible	Long term	Low - Moderate	***Not significant***

Table 4.51 Summary of effects on the regional population of herring gulls in the non-breeding season

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Low	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality (using either 98.0% or 99.5% A R)	Negligible	Long term	Moderate	***Not significant***
All effects combined	Negligible	Long term	Low - Moderate	***Not significant***

4.4.22.5 Cumulative Impact Assessment for herring gull

Combining the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts to the regional breeding population of herring gull will be as shown in Table 4.52. The individual impacts for displacement and collision are combined by addition to give the cumulative impact. The flight directions from colonies that are potentially affected by each wind farm acting as a barrier substantially overlap and therefore the cumulative barrier impact from the three developments is not the sum of the individual predicted impacts. The cumulative barrier impact is derived from the overall spread of flight directions at each colony potentially affected by the three developments forming barriers.

The predicted potential effective loss of <0.3% of the foraging habitat of the regional population of herring gulls in the breeding season (Table 4.52) is rated as an effect of negligible magnitude, temporally long-term and reversible. It is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional population of herring gulls in the breeding season is ***not significant*** under the terms of the EIA Regulations.

Table 4.52 Summary of CIA for the three proposed offshore wind farms in south-east Scotland on the regional population of herring gull in the breeding season

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF	Predicted impact from Inch Cape pOWF	Predicted impact from Firth of Forth Round 3 Zone pOWF	Cumulative Impact
Displacement - effective loss of marine foraging habitat during the breeding period (%)	25%	<0.1%, Negligible	<0.1%, Negligible	<0.1%, Negligible	≤0.3%, Negligible
Barrier Effect	25%	Foraging trips of <5% off Forth Islands and St Abb's Head birds potentially increase in length on average by 6%, Negligible	Foraging trips of <5% of Fowlsheugh and Forth Islands potentially increase in length on average by ca. 6%. Negligible	No barrier effect as no are flights likely to go beyond offshore site, Negligible	No barrier effect as no are flights likely to go beyond offshore site, Negligible
Collision most adverse designs: % increase in assumed annual AMR based on 98.0% avoidance	0%	0.6%, Negligible	1.9%, Low	1.0%, Low	3.5%, Low

Collision least adverse designs: % increase in assumed annual AMR based on 98.0% avoidance	0%	0.4%, Negligible	0.2%, Negligible	0.7%, Negligible	1.3%, Low
Collision most adverse designs: % increase in assumed annual AMR based on 99.5% avoidance	0%	0.2%, Negligible	0.5%, Negligible	0.3%, Negligible	0.9%, Negligible
Collision least adverse designs: % increase in assumed annual AMR based on 99.5% avoidance	0%	0.1%, Negligible	0.1%, Negligible	0.2%, Negligible	0.4%, Negligible

Barrier effects as assessed here concern birds which would otherwise fly through the offshore site to access feeding resources beyond it. The mean foraging distance of herring gull is 10.5 km and the mean maximum distance is 61.1 km (Thaxter *et al*., 2012). This means that almost no flights will be to areas beyond the barrier formed by the three proposed wind farms. As a result any barrier effect will be negligible. The cumulative barrier effect is rated as negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). Bearing in mind the low sensitivity of herring gull to barrier effects it is concluded that the cumulative impact of a barrier effect caused by the three proposed offshore wind farms on the regional population of herring gulls in the breeding season is ***not significant*** under the terms of the EIA Regulations.

The cumulative assessment of barrier effects above assumes that all areas of the wind farm lease sites will be developed. In the case of the Firth of Forth Round 3 Zone there will be large areas without turbines and the eventual design may therefore effectively present birds several smaller barriers with gaps between through which birds can pass, rather than the single large barrier assumed above. Were this to be the case, the magnitude of the cumulative barrier effect could be substantially less than suggested above.

Conclusion on the cumulative impact of collision mortality are sensitive to the wind farm designs evaluated and the level of avoidance rate used for predictive calculations of the number of collision strikes.

Using an avoidance rate of 98.0% the cumulative effect of the three wind farms is to increase the annual adult mortality rate by 1.3% for the least adverse wind farm designs and by 3.5% for the most adverse designs. The potential cumulative impact of this additional mortality on the regional population of herring gulls in the breeding season is rated as low magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that, when assessed using a 98.0% avoidance rate, the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional population of herring gull in the breeding season is of ***minor significance*** under the terms of the EIA Regulations.

Using an avoidance rate of 99.5% the cumulative effect of the three wind farms is to increase the annual adult mortality rate by 0.4% for the least adverse wind farm designs, and by 0.9% for the most adverse designs. The potential cumulative impact of this additional mortality on the regional population of herring gulls in the breeding season is rated as negligible magnitude (<1%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that, when assessed using a 99.5% avoidance rate, the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional population of herring gulls in the breeding season is ***not significant*** under the terms of the EIA Regulations.

A separate CIA has not been undertaken for the regional population of herring gulls in the non-breeding-period. This is because the cumulative assessment above for collision mortality on the regional breeding population includes collision deaths for the whole year. It assumes that 25% of the birds killed in the non-breeding period are from the regional breeding population. The predicted displacement of herring gulls due to Neart na Gaoithe wind farm in the non-breeding period is negligible and the sensitivity of the population to displacement at this time of year is also negligible. Therefore it is not plausible that the Development could contribute to a significant cumulative impact for this receptor population.

10.3.1.6 Mitigation measures for herring gull

The assessment does not identify any significant adverse effects on the regional population of herring gull in the breeding or non-breeding periods. Therefore no mitigation measures are required for this species.

4.4.23 Great black-backed gull *Larus marinus*

4.4.23.1 Status

Great black-backed gull is a common resident species, occurring in coastal areas. Largest numbers occur in western coasts, with a British population of 17,394 breeding pairs recorded during Seabird 2000 (Mitchell *et al*., 2004). The Isle of May is the closest colony to the Neart na Gaoithe development, with 37 breeding pairs in 2008 (SMP, 2012). Great black-backed gulls are omnivorous, foraging at sea, estuaries and beaches, and less commonly at rubbish dumps (Forrester *et al*., 2007).

4.4.23.2 Neart na Gaoithe Study Area

A total of 528 great black-backed gulls were recorded on surveys in the Neart na Gaoithe study area in Year 1, however only 25 birds were seen in the offshore site, with the majority of birds (95.3%) in the buffer area (Table 4.2). Most birds were recorded between September and January, with a peak of 307 birds in January. Fewer great black-backed gulls were recorded on surveys in Year 2 (434 birds), however numbers in the offshore site (20 birds) were similar to Year 1. As in Year 1, the majority of birds were in the buffer area (95.4%) between September and April, with a peak of 142 birds in April.

During the breeding season (April to August), the peak estimated number of great black-backed gulls occurred in April of Year 1, with zero birds in the offshore site, and an estimated 21 birds in the buffer area (Table 4.53) (Figure 4.57). In Year 2, peak estimated numbers of great black-backed gulls in the offshore site were recorded in June (seven birds), with an estimated 202 birds in the buffer area in April.

Table 4.53 Estimated numbers of great black-backed gulls in the offshore site and 8 km buffer in Years 1 and 2

Month	Estimated nos on water Offshore Site	Lower 95% C.L.	Upper 95% C.L.	Estimated nos flying Offshore Site	Estimated total Offshore Site	Estimated total 8 km buffer [1]	Estimated total
Yr1 Nov	0	0	0	7	7	270	277
Yr1 Dec	0	0	0	21	21	66	87
Yr1 Jan	0	0	0	14	14	658	672
Yr1 Feb	0	0	0	0	0	17	17
Yr1 Mar	0	0	0	0	0	21	21
Yr1 Apr	0	0	0	0	0	21	21
Yr1 May	0	0	0	0	0	0	0
Yr1 Jun	0	0	0	0	0	0	0
Yr1 Jul	0	0	0	0	0	0	0
Yr1 Aug	0	0	0	0	0	0	0
Yr1 Sep	0	0	0	7	7	62	69
Yr1 Oct	0	0	0	0	0	239	239
Yr2 Nov							
Yr2 Dec	0	0	0	7	7	172	179
Yr2 Jan	0	0	0	7	7	219	226
Yr2 Feb	0	0	0	0	0	7	7
Yr2 Mar	0	0	0	0	0	36	36
Yr2 Apr	3	1	8	0	3	202	205
Yr2 May	0	0	0	0	0	0	0
Yr2 Jun	0	0	0	7	7	0	7
Yr2 Jul	0	0	0	0	0	0	0
Yr2 Aug	0	0	0	0	0	0	0
Yr2 Sep	0	0	0	7	7	542	549
Yr2 Oct	0	0	0	0	0	51	51

In the Year 1 non-breeding season (September to March), the peak estimated number of great black-backed gulls in the offshore site was 21 birds in December, with an estimated 658 birds in the buffer area in January (Table 4.53) (Figure 4.57). In the Year 2 non-breeding season, the peak estimated number of great black-backed gulls in the offshore site was 7 birds in December and January, with an estimated 542 birds in the buffer area in September.

Overall, estimated total numbers in the Neart na Gaoithe study area were below the 1% threshold of national importance (400 birds) (Holt *et al.*, 2011) in all months except for January of Year 1 and September of Year 2 (Figure 4.57).

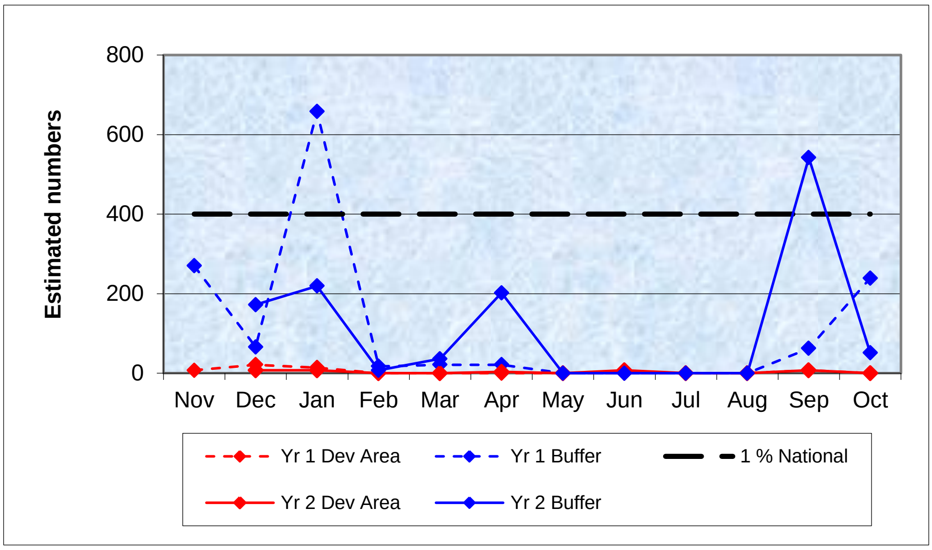


Figure 4.57 Monthly estimated numbers of great black-backed gulls in the Neart na Gaoithe Development & buffer areas in Years 1 and 2

Mean monthly great black-backed gull density in the offshore site was very low throughout both Years 1 and 2 (Figure 4.58). In the buffer area, mean density of great black-backed gulls peaked in January of Year 1, with lower peaks in April and September of Year 2, most likely connected with fishing vessel activity in the area. ESAS mean density data for the surrounding ICES rectangles and Regional Sea 1 peaked in December and September and October, with low mean densities recorded in the breeding season.

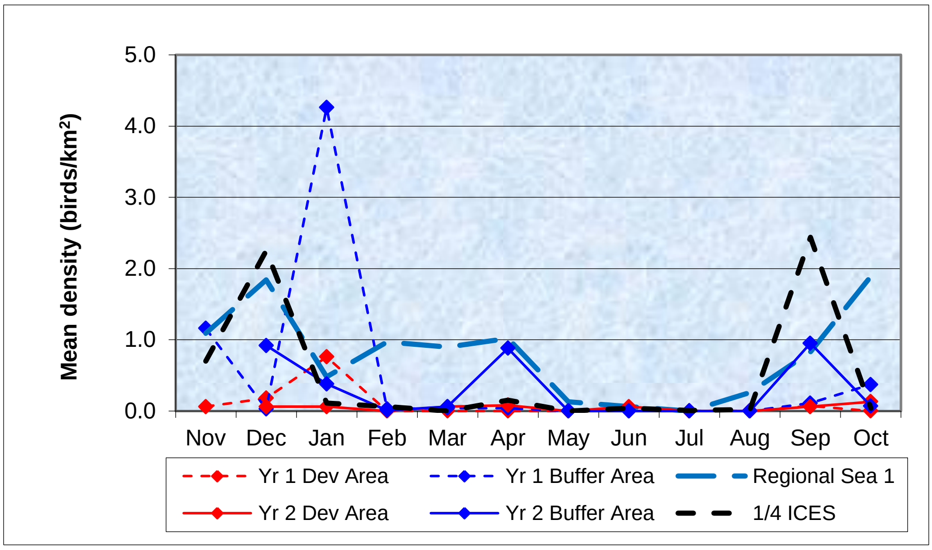


Figure 4.58 Comparison of monthly mean densities for great black-backed gull in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

In the Year 1 non-breeding period (November to March, September and October), great black-backed gulls were predominantly found in mostly low densities in the southern half of the study area, particularly the south-west, with fewer birds elsewhere (Figure 4.59). Generally low densities were recorded sporadically in the offshore site at this time.

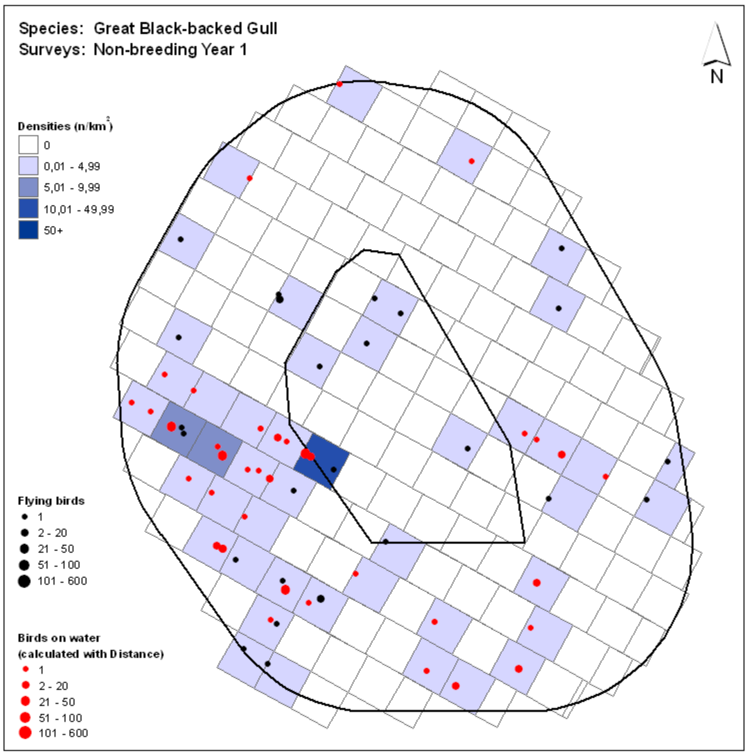


Figure 4.59 Great black-backed gull density in the non-breeding season, Year 1

A similar distribution pattern was recorded in the Year 2 non-breeding period, with the majority of great black-backed gulls recorded at low densities in the south-west of the buffer area (Figure 4.60). Low densities were recorded occasionally in the offshore site at this time.

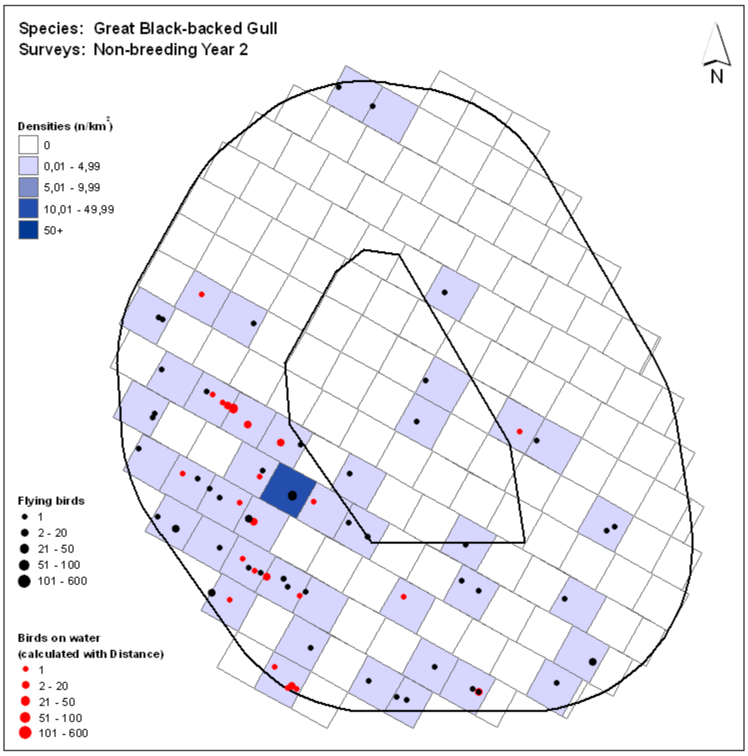


Figure 4.60 Great black-backed gull density in the non-breeding season, Year 2

Great black-backed gull distribution in the Year 1 breeding season (April to August) was very restricted, with no birds recorded in the offshore site, and few birds recorded at low densities in the eastern half of the buffer area over the period (Figure 4.61).

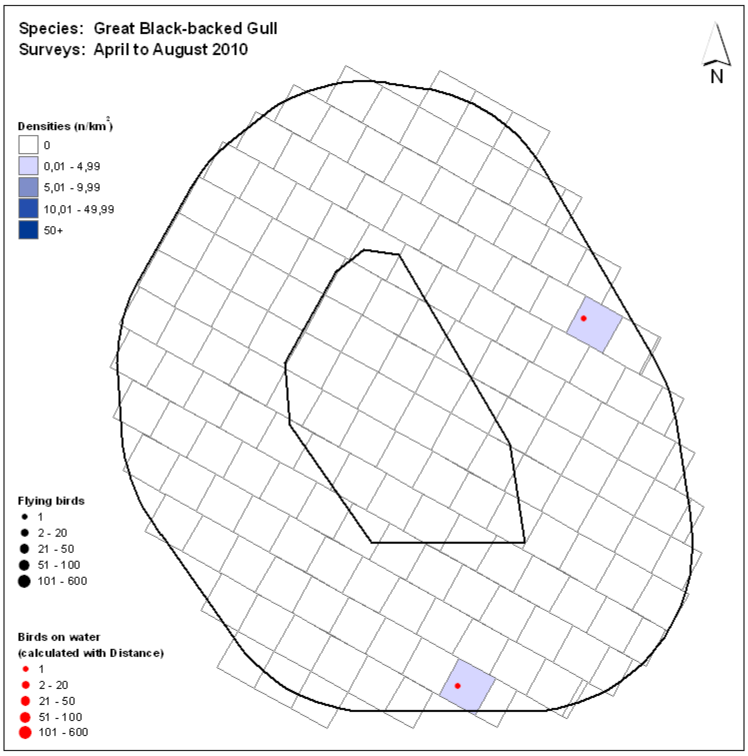


Figure 4.61 Great black-backed gull density in the breeding season, Year 1

Great black-backed gulls were more widely distributed between April and August of Year 2 compared to the same period in Year 1, with low densities and few birds recorded within the offshore site at this time. Highest densities were recorded in the west of the buffer area (Figure 4.62).

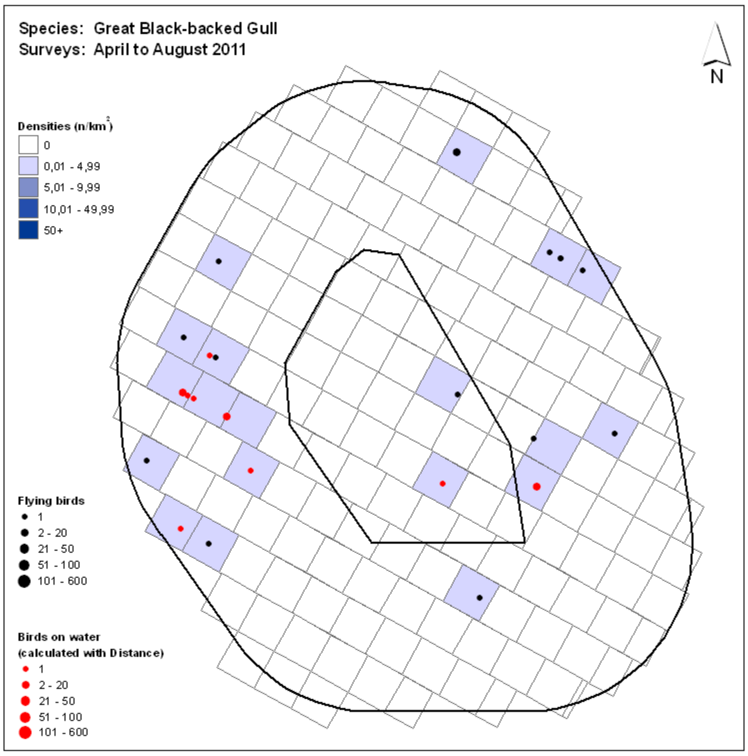


Figure 4.62 Great black-backed gull density in the breeding season, Year 2

In Years 1 and 2, 440 great black-backed gulls were recorded in flight, with 80.7% of birds flying below 22.5 m (Table 4.3). A total of 85 birds (19.3%) were recorded flying above 22.5 m, i.e. within the rotor swept zone, at estimated heights of between 25 m and 60 m.

Foraging behaviour was recorded for 188 great black-backed gulls in the Neart na Gaoithe study area in Year 1, with three types of foraging behaviour recorded, and unspecified feeding behaviour recorded for a further 20 birds (Table 4.54). The majority of all foraging birds were recorded scavenging at fishing vessels (85.6%).

Table 4.54 Great black-backed gull foraging behaviour in the Neart na Gaoithe Study Area in Years 1 and 2

Behaviour	Number of birds
Actively searching	6
Holding fish	1
Scavenging	161
Feeding method unspecified	20
Total	**188**

10.3.1.7 Species sensitivity

Great black-backed gull is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009). A recent review assessed great black-backed gull as being at moderate risk of collision. Displacement, barrier effects, habitat loss or changes in prey distribution resulting from offshore wind farms were rated as low risk. Overall, great black-backed gull was assessed as being at moderate risk from offshore wind developments (Langston, 2010) (Table 4.4).

Great black-backed gull is not listed as a qualifying interest species in the breeding season for any SPAs on the UK east coast between Peterhead and Blyth (JNCC, 2012). The nearest SPA for breeding great blacked-gulls is Copinsay SPA, approximately 297 km from the offshore site. The estimated maximum foraging distance for this species is less than 10 km (Roos *et al*., 2010).

4.4.23.3 Assessment

Definition of seasons

The annual cycle for great black-backed gull is divided into two parts to reflect the biology of the species and the broad pattern of use of the offshore site. The main breeding season, when breeding adults are attending colonies, is defined as April to August. At this time the birds present in the offshore site are likely to be from relatively local breeding colonies (e.g. colonies that are closer to the offshore site than the mean maximum foraging distance) or may be non-breeding immature birds.

The non-breeding period is defined as September to March and broadly corresponds to the period when great black-backed gulls are in their over-wintering area. In this period, especially after October, when numbers in the region show a large increase, it is likely that the vast majority of individuals present in the offshore site are from outwith the region, including birds from Scandinavia and Russia (Skov *et al.,* 1995, Wernham *et al.,* 2002).

Baseline conditions

In the breeding season (April to August), the mean estimated number of great black-backed gulls present in the offshore site was less than one bird (0.9), and the mean estimated number in the offshore site buffered to 1 km was less than two birds (1.5). On average 28% of birds present in the breeding season in the offshore site were on the sea and the remainder were in flight.

Based on survey results for the whole Survey Area in this period, 40.0% of great black-backed gulls were aged as adults and the remainder were immature birds. For the purposes of assessment it is assumed that all birds aged as adults were breeding birds and that the mean colony attendance rate by adults was 50%.

In the non-breeding period (September to March) the mean number of great black-backed gulls present in the offshore site was 5, and the mean number in the offshore site buffered to 1 km was 36 birds. Based on results for the whole Survey Area in this period, 50.5% of great black-backed gulls were aged as adults and the remainder were immature birds.

Nature conservation importance

Great black-backed gull is rated as low NCI during the breeding season and non-breeding season. The mean number present in the offshore site buffered to 1 km in the breeding season are well below 1% of the (at-sea) regional breeding population, and the species is not subject to any special legislative protection.

Offshore wind farm studies of great black-backed gull

Available data from other offshore wind farms indicate that great black-backed gulls may be attracted to offshore wind farms and there are not likely to be any displacement effects (Zucco *et al*., 2006). Results from studies undertaken at Egmond ann Zee reported a significant positive attraction to the offshore wind farm in four surveys, two surveys indicated a significant avoidance effect and eleven surveys indicated no effect on great black-backed gull (Leopold *et al.,* 2011).

Data from post-construction monitoring studies undertaken in Denmark indicate that there is no barrier effect on great black-backed gulls from constructed wind farms (Zucco *et al*., 2006).

Great black-backed gulls fly relatively more frequently at rotor height compared to most other seabird species. Consequently this species is at greater risk of collision than most other seabirds.

EIA Construction Phase assessment for great black-backed gull

The construction phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

Great black-backed gulls are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during construction operations on the regional populations of great black-backed gulls in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

EIA Operational Phase assessment for great black-backed gull

Potential for great black-backed gull to be affected by displacement

For the purpose of assessing displacement it is assumed that 25% of great black-backed gulls will be displaced from the proposed wind farm footprint and a surrounding buffer of 1 km. Based on evidence from existing wind farms this is likely to be a cautious assumption because reported declines in great black-backed gull abundance at existing wind farm are below 25%, indeed, some sites showed no significant change, while increased numbers were recorded at other sites during the operational phase.

Table 4.55 The mean estimated number of great black-backed gull present during the breeding and the non-breeding periods, and the value as a percentage of the (at-sea) receptor population potentially at risk of displacement

Receptor population	Population size (adults)	No. assumed at sea	Dev.Area		Dev.Area + 1km	
			Mean no. at risk	%	Mean no. at risk	%
National, breeding period (Seabird 2000)	39,382	19,691	0.4	<0.1	0.6	<0.1
North Sea, winter period (Skov *et al*., 1995)	299,900	299,900	5.2	<0.1 %	36.4	<0.1
Regional, breeding period (Seabird 2000)	226	113	0.4	0.3	0.6	0.5
Regional, winter period (Skov *et al*., 1995)	21,500	21,500	5.2	<0.1	36.4	<0.2

Likely impacts of displacement on great black-backed gull populations

Studies of great black-backed gulls show that this species exhibits little, if any, displacement behaviour in response to operational wind farms (Zucco *et al*., 2006, Leopold *et al.,* 2011).

If 25% of great black-backed gulls were displaced during the breeding season from the offshore site buffered to 1 km the impact of this would be the effective loss of up to around 0.1% of the foraging habitat of the regional breeding population (Table 4.55). This impact is categorised as negligible magnitude, and temporally long-term and reversible. It is concluded that the effects of displacement on the regional great black-backed gull population in the breeding season are ***not significant*** under the EIA Regulations.

If 25% of great black-backed gulls were displaced during the non-breeding period from the offshore site buffered to 1 km the impact of this would be the effective loss of <0.1% of the foraging habitat of the non-breeding-period population. This impact is categorised as negligible magnitude, and temporally long-term and reversible. It is concluded that the effects of displacement on the regional great black-backed gull population in the non-breeding period are ***not significant*** under the EIA Regulations.

Barrier effect

Great black-backed gulls breeding at nearby colonies to the proposed wind farm could be affected by the wind farm acting as a barrier. The greatest potential for such impacts is for birds from the Isle of May.

Observations from operational offshore wind farms shows no evidence that wind farms pose a barrier to great black-backed gulls. This, together with the limited maximum known foraging range (< 10 km) (Roos *et al.,* 2010) indicates that there will be no barrier effect on great black-backed gull arising from the Development in the breeding season.

Great black-backed gulls are considered to have low sensitivity to barrier effects on account of a low wing loading (Maclean *et al.*, 2009, Langston, 2010). The potential effects on great black-backed gulls of the offshore site acting as a barrier are assessed for the breeding season, when birds are attending colonies.

Barrier effects as calculated here concern birds which would otherwise fly through the offshore site to access feeding resources beyond it. Therefore, the only birds that barrier effects may affect are those whose flights lie in the direction of the offshore site and for which the intended destination is beyond the offshore site. The maximum known foraging distance of great black-backed gulls is only 10 km (Roos *et al.,* 2010). This means that no foraging flights by this species are predicted to be to areas beyond the barrier formed by the offshore site. As a result, any barrier effect will be negligible.

The barrier effect on the breeding population of great black-backed gull is rated as negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). Bearing in mind the low sensitivity of great black-backed gull to barrier effects it is concluded that the impact of a barrier effect on the regional population of great black-backed gull in the breeding season is ***not significant*** under the terms of the EIA Regulations.

Collision mortality

For the purposes of assessment it is assumed that there will be no far-field avoidance of the proposed wind farm by great black-backed gulls. Therefore, the mortality that is considered likely to occur for wind farm designs and assumed avoidance rate scenarios illustrated is that presented in Table 4.56 and Table 4.57 (see Appendix 12.2 for further details). COWRIE guidance (Maclean *et al.,* 2009) is to use a default avoidance rate of 99.5% for gull species. There is no specific SNH guidance on avoidance rates for seabirds, and therefore the default value of 98.0% recommended by SNH is applicable.

19.3% of flying great black-backed gulls observed in baseline surveys were within the height range swept by the proposed turbine rotors (Table 4.56).

There is no published adult survival rate for great black-backed gull, therefore the rate for herring gull (0.88) is used as a surrogate; this is likely to affect the accuracy of the predictions. The most recent information indicates that the regional breeding population is 226 adults (Mitchell *et al.,* 2004). Approximately 27 adults die each year as a consequence of baseline mortality, assuming an adult survival rate of 0.88 per annum (BTO, 2012).

It is assumed that all birds present during the breeding season are from the regional breeding population. In the non-breeding period, the regional population comprises a mixture of birds originating from the region and beyond. Therefore, to estimate the effect of year-round collision mortality on the regional breeding population the proportion of collision deaths that occur in the non-breeding period involving birds from the regional breeding population needs to be estimated. Ringing and colour marking studies show that large numbers of great black-backed gull from northern Scotland and Scandinavian breeding grounds overwinter in the region (Wernham *et al*., 2002). The winter population estimate for the region (approximately 21,500 birds, Skov *et al*., 1995) also indicates that there is a large net influx of great black-backed gull into the region in the winter, with around ten times as many birds present in winter compared to the breeding season. The estimate of regional wintering numbers by Skov *et al.* (1995) is likely to be an underestimate because it is based on surveys of birds at sea and therefore does not include the several hundreds of birds on land, e.g., wintering on coasts and at refuse tips (Forrester and Andrews, 2007). Based on this evidence, for the purposes of assessment it is assumed that 10% of the adult collision deaths during the non-breeding period involve adults from the regional breeding population. The effect of year-round collision mortality on the annual adult mortality rate for the regional breeding population is calculated by summing the breeding season mortality and the non-breeding period mortality attributed to breeding birds from the region.

Table 4.56 Proportion of great blacked-backed gull estimated to be at rotor height and the number of collisions predicted by collision rate modelling (CRM) using avoidance rates (AR) of 98.0% and 99.5% for least adverse (64 x 7MW turbines) and most adverse (128 x 3.6MW turbines) wind farm designs. LAT is sea level at lowest astronomical tide, and this is approximately 2.0 m below the average sea level.

Metric	Design 1, 64 x 7MW		Design 4, 128 x 3.6 MW	
	98.0% AR	99.5% AR	98.0% AR	99.5% AR
Rotor swept height range (metres above LAT)	24 - 188		24 - 144	
Flight activity at rotor height	19.3%		19.3%	
Flight activity below rotor height (<22.5 m above sea)	80.7%		80.7%	
Collisions in breeding season (April to August), all ages	2.2	0.5	3.3	0.8
Collisions in non-breeding period (September to March), all ages	19.0	4.7	28.9	7.2
Total collisions per year, all ages	21.2	5.3	32.3	8.1

Table 4.57 The effect of CRM predicted collision mortality on the adult mortality rate (AMR) of the regional breeding great black-backed gull population. Results are presented for least adverse (64 x 7MW turbines) and most adverse (128 x 3.6MW turbines) wind farm designs, and for two values of overall avoidance rate (OAR)

Wind farm design Period of year	% birds present assumed to be from regional breeding popn	98.0% OAR		99.5% OAR	
		CRM deaths	Change AMR	CRM deaths	Change AMR
Design 1, 64 x 7MW turbines					
Breeding period (April - August)	100%	0.9	3.2%	0.2	0.8%
Non-breeding period (Sept. - March)	10%	1.0	3.5%	0.2	0.9%
Whole year	varies	1.8	6.8%	0.5	1.7%
Design 4, 128 x 3.6MW turbines					
Breeding period (April - August)	100%	1.3	4.9%	0.3	1.2%
Non-breeding period (Sept. - March)	10%	1.5	5.4%	0.4	1.3%
Whole year	varies	2.8	10.3%	0.7	2.6%

Using an avoidance rate of 99.5%, the increase in the assumed adult annual mortality rate of the regional population would be between 1.7% and 2.6% depending on wind farm design. Using an avoidance rate of 98.0%, the increase is between 6.8% and 10.3%. For both avoidance rates the changes in mortality rate are in all cases above the default guidance threshold of 1%, which is considered the minimum likely to have potential for adverse impacts on a population (King *et al.*, 2009).

The potential impact of the predicted collision mortality based on a 99.5% avoidance rate on the regional breeding population is an effect of low magnitude (1-5%) and temporally long-term and reversible. Bearing in mind also that the regional breeding population of great black-backed gull is categorised as having low nature conservation importance, it is concluded that the impact of collision mortality on the regional population of great black-backed gulls in the breeding and non-breeding periods is an effect of ***minor significance*** under the terms of the EIA Regulations.

The potential impact of the predicted collision mortality based on a 98.0% avoidance rate on the regional breeding population is an effect of moderate magnitude (5-20%) and temporally long-term and reversible. Considering the low nature conservation importance for this species, it is concluded that the impact of collision mortality on the regional population of great black-backed gulls in the breeding and non-breeding periods is an effect of ***moderate significance*** under the terms of the EIA Regulations.

EIA Decommissioning Phase assessment for great black-backed gull

The decommissioning phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

Great black-backed gulls are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during decommissioning operations on the regional populations of great black-backed gulls in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

EIA summary of effects combined for great black-backed gull

The three effects assessed will act on the regional breeding population in a broadly additive manner (Table 4.58). In combination it is judged that the magnitude of the three effects is low, using a 99.5% avoidance rate and the overall impact on the regional breeding population is judged of ***minor significance*** under the EIA regulations.

Table 4.58 Summary of effects on the regional breeding population of great black-backed gull. The effects on adult mortality rate due to collision includes collision deaths occurring outside the breeding period and attributed to birds from the regional breeding population

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Low	***Not significant***
Barrier Effect	Negligible	Long term	Low	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality using 98.0% AR	Moderate	Long term	Moderate	***Moderate significance***
Collision mortality using 99.5% AR	Low	Long term	Moderate	***Minor significance***
All effects combined (98.0% AR for collision risk)	Moderate	Long term	Low - Moderate	***Moderate significance***
All effects combined (99.5% AR for collision risk)	Low	Long term	Low - Moderate	***Minor significance***

However, using a 98.0% avoidance rate, the magnitude of the effects combined increases to moderate. Under this assessment scenario the overall impact on the regional breeding population is judged of ***moderate significance*** under the EIA regulations.

In the non-breeding-period, the three effects assessed will act on the regional population in a broadly additive manner (Table 4.58). In combination it is judged that the magnitude of the three effects is negligible using a 99.5% avoidance rate, and the overall impact on the regional non-breeding-period population is judged ***not significant*** under the EIA regulations.

However, using a 98.0% avoidance rate the magnitude of the effects combined increase to low. Under this assessment scenario the overall impact on the regional population in the non-breeding-period is judged of ***minor significance*** under the EIA regulations.

Table 4.59 Summary of effects on the regional population of great black-backed gulls in the non-breeding period

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Low	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality using 98.0% A R	Low	Long term	Moderate	***Minor significance***
Collision mortality using 99.5% A R	Negligible	Long term	Moderate	***Not significant***
All effects combined (98.0% A R for collision risk)	Low	Long term	Low - Moderate	***Minor significance***
All effects combined (99.5% A R for collision risk)	Negligible	Long term	Low - Moderate	***Not significant***

4.4.23.4 Cumulative Impact Assessment for great black-backed gull

Combining the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts to the regional population of great black-backed gull in the breeding season will be as shown in the right most column of Table 4.60. The individual impacts for displacement and collision are combined by addition to give the cumulative impact.

Insufficient data were presented for great black-backed gull in the Year 1 report for the Firth of Forth Round 3 Zone (SWEL, 2011) to quantitatively assess the likely displacement effect arising from this development during the breeding season. On the basis of the very low numbers of this species present in the breeding season recorded at the other two sites, and the relatively large distance of the Firth of Forth Round 3 Zone from the coast, displacement of this species from the Firth of Forth Round 3 Zone is assumed to be of negligible magnitude. Overall, the likely displacement effect caused by the three proposals is the effective loss of <1% of foraging habitat during the breeding season (Table 4.60). This effect is categorised as negligible in magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that displacement caused by the three proposed offshore wind farms on the regional great black-backed gull population in the breeding season is ***not significant*** under the terms of the EIA Regulations.

Table 4.60 Summary of CIA for the three proposed offshore wind farms in south-east Scotland on the regional population of breeding great black-backed gull

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF	Predicted impact from Inch Cape pOWF	Predicted impact from Firth of Forth Round 3 Zone pOWF	Cumulative Impact
Displacement: effective loss of foraging habitat during breeding period (%)	25%	0.1%, Negligible	<0.1%, Negligible	Negligible	<0.2% (+), Negligible
Barrier Effect:	25%	No flights likely to go beyond offshore site, Negligible	No flights likely to go beyond offshore site, Negligible	No flights likely to go beyond offshore site, Negligible	No flights likely to go beyond offshore site, Negligible
Collision most adverse designs: % increase in assumed annual adult mortality rate based on 98.0% avoidance rate.	0%	10.3% 2.8 collisions High	60.9% 16.5 collisions Very High	no data available, likely High	71.2% (+), likely Very High
Collision least adverse designs: % increase in assumed annual adult mortality rate based on 98.0% avoidance rate.	0%	6.8% 1.8 collisions High	8.6% 2.3 collisions High	no data available, likely High	15.4% (+), likely Very High
Collision most adverse designs: % increase in assumed annual adult mortality rate based on 99.5% avoidance rate.	0%	2.6% 0.7 collisions Low	15.2% 4.1 collisions Moderate	no data available, likely Moderate	17.8% (+), likely High
Collision least adverse designs: % increase in assumed annual adult mortality rate based on 99.5% avoidance rate.	0%	1.7% 0.5 collisions Low	2.1% 0.6 collisions Low	no data available, likely Low	3.8% (+), likely Moderate

The cumulative impact of barrier effects on great black-backed gull during the breeding season is categorised as negligible in magnitude, temporally long-term and reversible (Table

3.1 and Table 3.2). It is concluded that the cumulative impact of barrier effects caused by the three proposed offshore wind farms on the regional great black-backed gull population in the breeding season is ***not significant*** under the terms of the EIA Regulations.

Conclusions on the cumulative impact of collision mortality are sensitive to the wind farm designs evaluated and the level of avoidance rate used for predictive calculations of the number of collision strikes. Information on the predicted number of collisions of great black-backed gull are for the Firth of Forth Round 3 Zone was not available (SWEL, 2011). Nevertheless this species is reported to occur in moderate numbers in the Firth of Forth Round 3 Zone, especially in the winter (Skov *et al.,* 1995). Therefore, it is reasonable to assume that collision modelling for this development would predict a moderate number of great black-backed gull collisions, as is the case for the other two sites.

Using an avoidance rate of 99.5% the cumulative effect of the Inch Cape and Neart na Gaoithe wind farms is to increase the assumed annual adult mortality rate by 3.8% for the least adverse wind farm designs, and by 17.8% for the most adverse designs. Assuming that the least adverse designs are chosen, the potential cumulative impact of this additional mortality on the regional great black-backed gull population in the breeding season is rated as low magnitude (1-5%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that, using a 99.5% avoidance rate, the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional population of great black-backed gulls in the breeding season is of ***minor significance*** under the terms of the EIA Regulations. Additional deaths from the Firth of Forth Round 3 Zone development are likely to substantially increase the number of collision strikes annually, possibly increasing the impact on the population to ***moderate significance*** under the terms of the EIA Regulations.

Using an avoidance rate of 98.0% the cumulative effect of the Inch Cape and Neart na Gaoithe wind farms is to increase the assumed annual adult mortality rate by 15.4% for the least adverse wind farm designs and by 71.2% for the most adverse designs. Assuming that the least adverse designs are chosen, the potential cumulative impact of this additional mortality on the regional great black-backed gull population in the breeding season is rated as moderate magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that, using a 98.0% avoidance rate, the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional population of great black-backed gull in the breeding season is of ***moderate significance*** under the terms of the EIA Regulations. Additional collisions from the Firth of Forth Round 3 Zone are likely to substantially increase the cumulative number of collision strikes annually, possibly increasing the impact on the population to ***major significance*** under the terms of the EIA Regulations.

A separate CIA has not been undertaken for the regional great black-backed gull population in the non-breeding-period. This is because the cumulative assessment above for collision mortality on the regional breeding population includes collision deaths for the whole year. It assumes that 10% of the great black-backed gull collisions outside the breeding period are from the regional breeding population. The predicted displacement of great black-backed gulls due to the Neart na Gaoithe wind farm in the non-breeding period is negligible and the sensitivity of the population to displacement at this time of year is low. Therefore it is not plausible that the Neart na Gaoithe wind farm could contribute to a significant cumulative impact for this receptor population.

4.4.23.5 Mitigation measures for great black-backed gull

There are few if any practical mitigation measures that are likely to significantly reduce the potential collision mortality for great black-backed gull.

4.4.24 Black-legged kittiwake *Rissa tridactyla*

4.4.24.1 Status

Kittiwakes are one of the commonest seabird species in the UK, breeding in large colonies on suitable coastal cliff habitat. Largest numbers occur on the east coast, and 366,835 breeding pairs were recorded in Britain during Seabird 2000 (Mitchell *et al.,* 2004). The closest large colonies to the Neart na Gaoithe development are the Isle of May, St Abb's Head and Fowlsheugh. Kittiwakes mostly prey on small fish species such as lesser sandeels and clupeids, as well as fishery discards (Forrester *et al.,* 2007).

4.4.24.2 Neart na Gaoithe Study Area

Kittiwake was the fourth commonest seabird recorded on surveys in the Neart na Gaoithe study area in Year 1, with a total of 3,955 birds. The majority of birds (79.7%) were recorded in the buffer area (Table 4.2). Birds were recorded in all months, with peak numbers in September and October. In Year 2, numbers recorded on survey were similar, with 4,123 birds seen. Again, the majority of birds were recorded in the buffer area. Numbers in the offshore site peaked in July (220 birds), while in the buffer area peak numbers were recorded in October (1,031 birds).

During the Year 1 breeding season (April to August), the peak estimated number of kittiwakes in the offshore site occurred in August (159 birds), with an estimated 2,477 birds in the buffer area in April (Table 4.61) (Figure 4.63). In the Year 2 breeding season, estimated numbers of kittiwakes peaked in July, with 1,616 in the offshore site and 3,578 in the buffer area.

In the post-breeding season (September and October), peak estimated numbers of kittiwakes in the offshore site occurred in September of Year 1 (2,195 birds), with 6,070 birds in the buffer area in October (Table 4.61) (Figure 4.63). In Year 2, peak estimated numbers of kittiwakes occurred in October, with an estimated 88 birds in the offshore site, and 8,380 birds in the buffer area.

In the non-breeding season (November to March), peak estimated numbers of kittiwakes in December of Year 1, with 39 birds in the offshore site and 379 birds in the buffer area (Table 4.61) (Figure 4.63). In Year 2, peak estimated numbers of kittiwakes in the offshore site occurred in December of Year 1 (808 birds), with a peak of 1,091 birds in the buffer area in January.

Table 4.61 Estimated numbers of kittiwakes in the offshore site and 8 km buffer in Years 1 and 2

Month	Estimated nos on water Offshore Site	Lower 95% C.L.	Upper 95% C.L.	Estimated nos flying Offshore Site	Estimated total Offshore Site	Estimated total 8 km buffer [1]	Estimated total
Yr1 Nov	6	1	32	20	26	315	341
Yr1 Dec	25	7	88	14	39	379	418
Yr1 Jan	0	0	0	0	0	0	0
Yr1 Feb	0	0	0	0	0	119	119
Yr1 Mar	0	0	0	0	0	40	40
Yr1 Apr	0	0	0	7	7	2,477	2,484
Yr1 May	0	0	0	41	41	127	168
Yr1 Jun	36	16	80	20	56	464	520
Yr1 Jul	0	0	0	41	41	154	195
Yr1 Aug	152	38	599	7	159	1,274	1,433
Yr1 Sep	2,032	859	4,809	163	2,195	3,772	5,967
Yr1 Oct	1,883	1,032	3,435	136	2,019	6,070	8,089
Yr2 Nov	-	-	-	-	-	-	-
Yr2 Dec	72	32	162	736	808	383	1,191
Yr2 Jan	0	0	0	7	7	1,091	1,098
Yr2 Feb	0	0	0	14	14	223	237
Yr2 Mar	0	0	0	7	7	662	669
Yr2 Apr	239	127	451	48	287	740	1,027
Yr2 May	47	18	122	81	128	365	493
Yr2 Jun	0	0	0	27	27	1,803	1,830
Yr2 Jul	1,393	548	3,537	223	1,616	3,578	5,194
Yr2 Aug	18	7	44	27	45	481	526
Yr2 Sep	0	0	0	20	20	783	803
Yr2 Oct	0	0	0	88	88	8,380	8,468

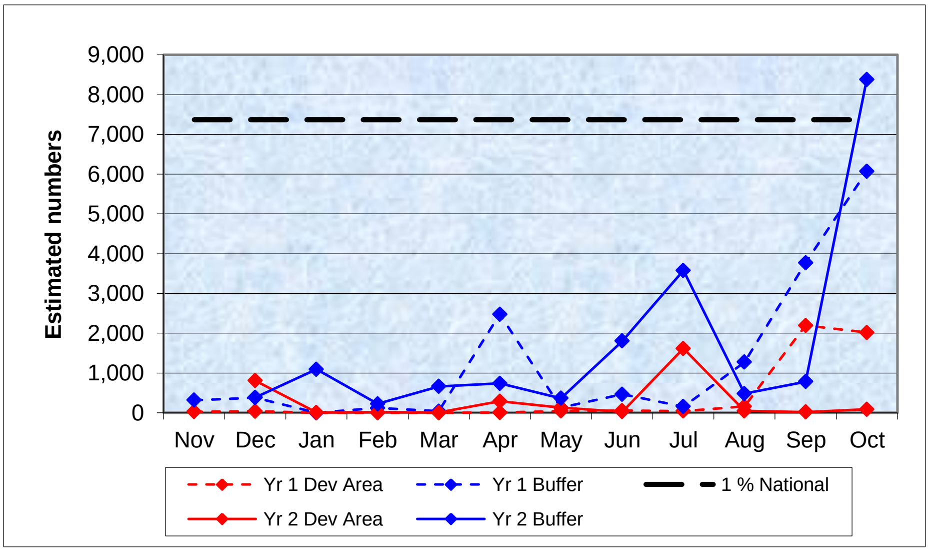


Figure 4.63 Monthly estimated numbers of kittiwakes in the Neart na Gaoithe Development & buffer areas in Years 1 and 2

Mean monthly kittiwake density was generally similar between the offshore site and the buffer area in Years 1 and 2. Mean density in the offshore site peaked in September of Year 1 with 24.1 birds/km^2, compared to a peak of 13.9 birds/km^2 in July of Year 2 (Figure 4.64). In the buffer area, mean density was highest in October, with 11.6 birds/km^2 in Year 1, compared to 8.2 birds/km^2 in Year 2. ESAS mean density data for the surrounding ICES rectangles was similar, with a peak of 10.2 birds/km^2 in September. ESAS mean density data for Regional Sea 1 was generally lower, with a peak of 4.5 birds/km^2 in August.

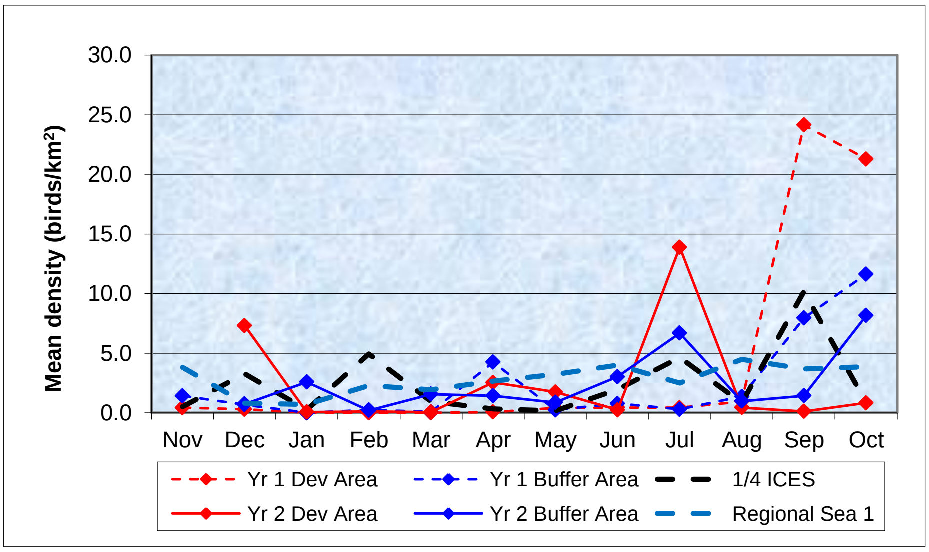


Figure 4.64 Monthly mean densities for kittiwake in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

Between November and March of Year 1, low densities of kittiwakes were recorded sporadically in the offshore site (Figure 4.65). In the buffer area, kittiwakes were scattered at low to moderate densities, with highest densities in the north and south of the buffer area.

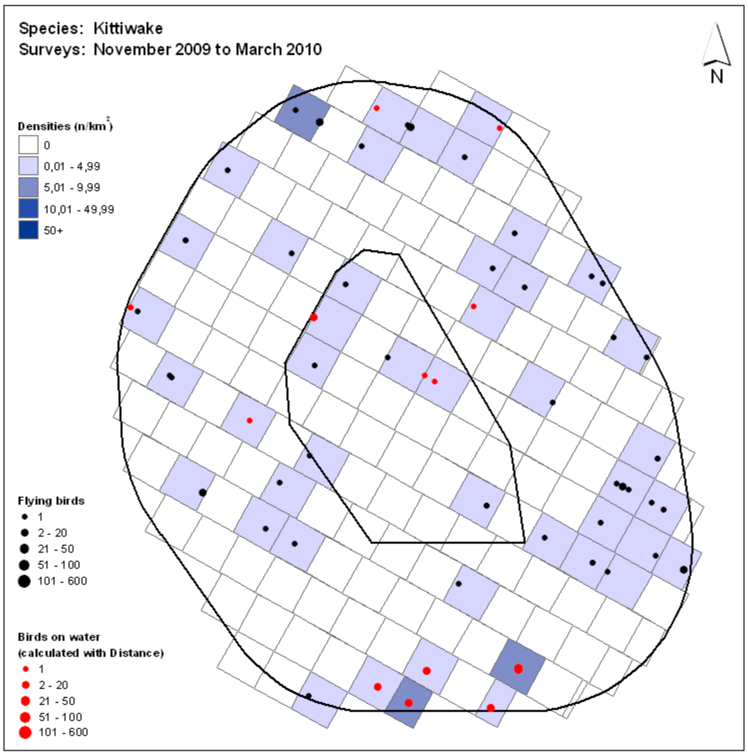


Figure 4.65 Kittiwake density between November and March, Year 1

Over the same period in Year 2, moderate to high densities of kittiwakes were recorded in the north of the offshore site , with low densities elsewhere (Figure 4.66). In the buffer area, kittiwakes were again scattered at mostly low to moderate densities, with highest densities in the south-east of the buffer area.

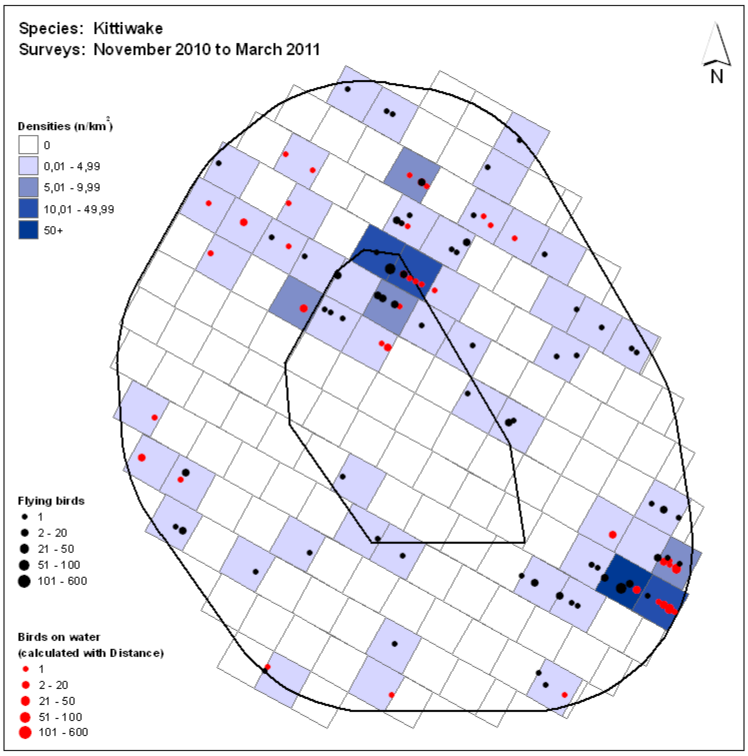


Figure 4.66 Kittiwake density between November and March, Year 2

Between April and August of Year 1, kittiwakes were more widespread across the study area, at mostly low to moderate densities (Figure 4.67). Highest densities were recorded in the south of the offshore site and south-east of the buffer area at this time.

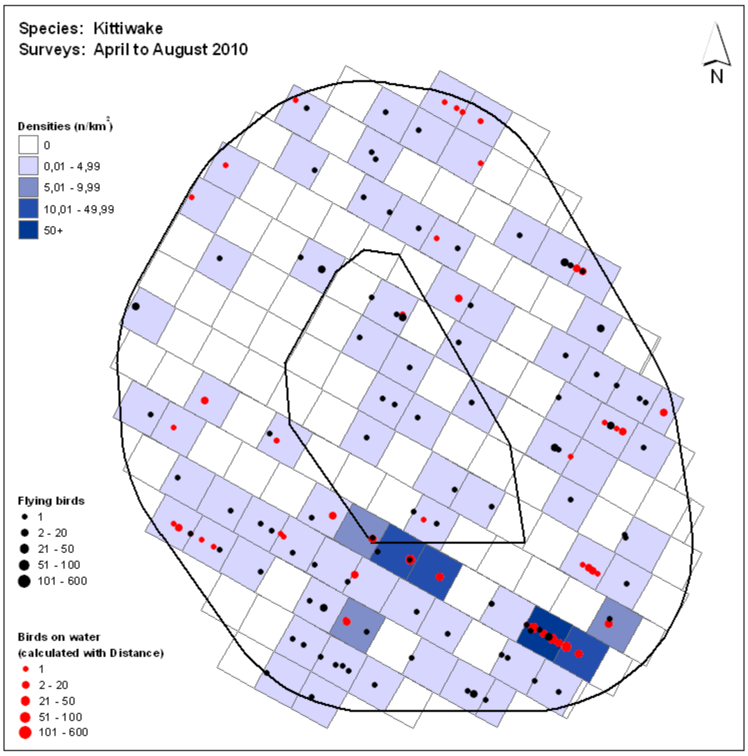


Figure 4.67 Kittiwake density between April and August, Year 1

Kittiwakes were more widespread between April and August of Year 2 than in Year 1, with high densities scattered throughout the offshore site and buffer area (Figure 4.68).

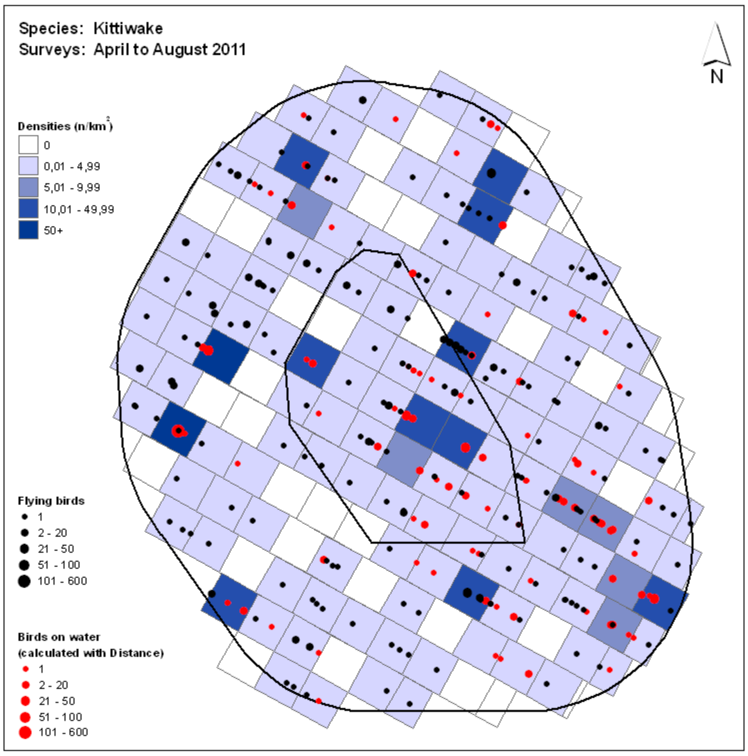


Figure 4.68 Kittiwake density between April and August, Year 2

In Year 1, peak densities of kittiwakes were recorded in September and October (Figure 4.69). Birds were widespread at low to high densities across the study area at this time.

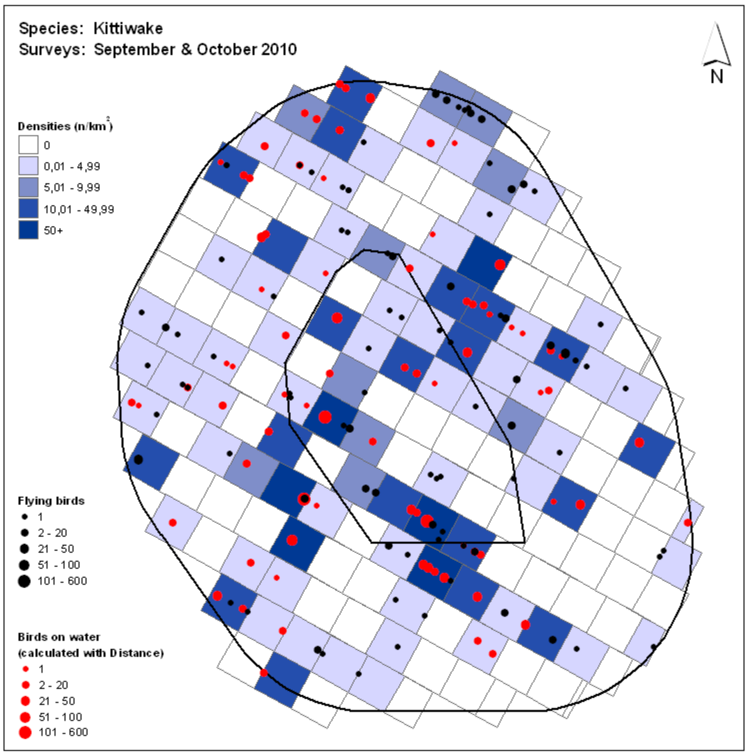


Figure 4.69 Kittiwake density between September and October, Year 1

In September and October of Year 2, densities of kittiwakes in the offshore site were mostly low, and were mainly in the south (Figure 4.70). In the buffer area, few birds were found in the south-east. High densities were scattered across the remainder of the buffer area at this time.

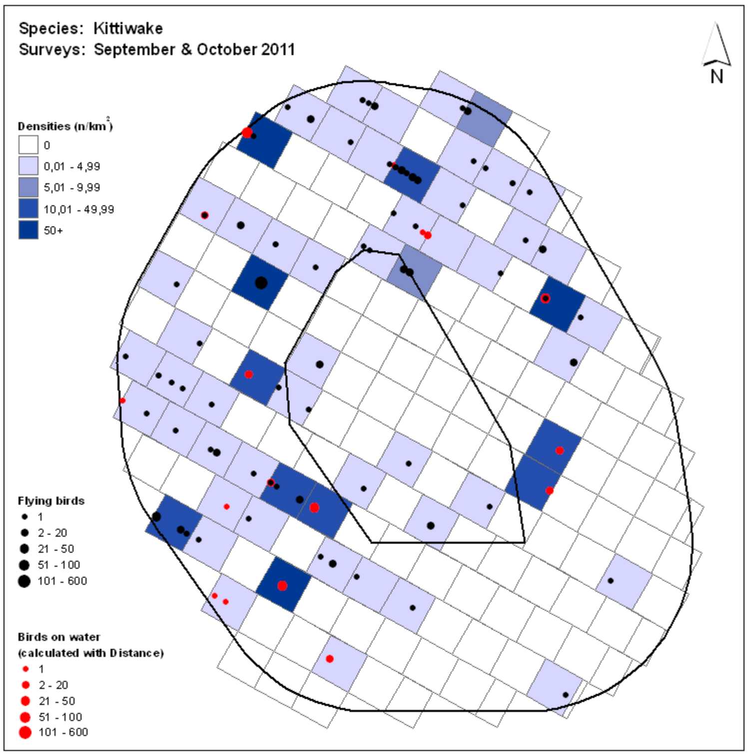


Figure 4.70 Kittiwake density between September and October, Year 2

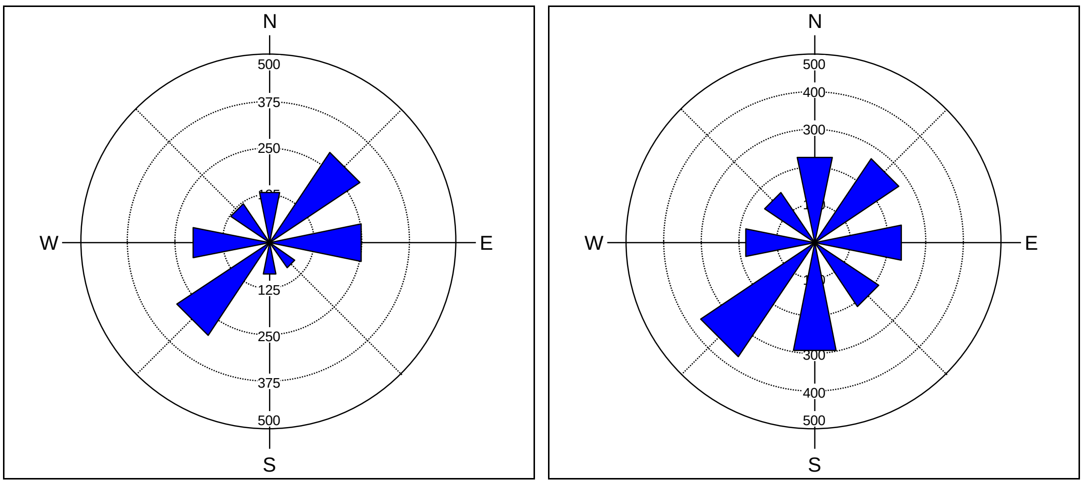
In Years 1 and 2, 4,914 kittiwakes were recorded in flight, with the majority of birds (94.0%) recorded flying below 22.5 m in height (Table 4.3). A total of 295 birds (6.0%) were recorded flying above 22.5 m, i.e. within the rotor swept zone, at estimated heights of between 25 m and 70 m, although the majority of higher flying birds (79.7%) were recorded below 35 m.

Foraging behaviour was recorded for 1,662 kittiwakes in the Neart na Gaoithe study area on baseline surveys, with five types of foraging behaviour recorded, and unspecified feeding behaviour recorded for a further 50 birds (Table 4.62). The majority of all foraging birds were recorded actively searching (27.2%) and dipping (57.4%).

Table 4.62 Kittiwake foraging behaviour in the Neart na Gaoithe Study Area in Years 1 and 2

Behaviour	Number of birds
Actively searching	465
Deep plunging	5
Dipping	982
Holding fish	1
Surface pecking	209
Feeding method unspecified	50
Total	**1,712**

Flight direction was recorded for 1,467 kittiwakes in the breeding season (April to August), and 1,943 kittiwakes in the non-breeding season (September to March) (Figure 4.71). In the breeding season, just over a fifth of all birds recorded were flying south-west (20.1%), with 19.7% of birds flying north-east. In the non-breeding season, 18.7% of birds were recorded flying south-west, with 15.1% flying south. An additional 1,110 birds were recorded as circling (not shown).



April to August (n=1,467 birds) September to March (n=1,943 birds)
Numbers shown on figures are number of birds recorded

Figure 4.71 Flight direction of kittiwakes in the Neart na Gaoithe Study Area in Years 1 and 2

4.4.24.3 Species sensitivity

Kittiwake is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

A recent review assessed kittiwake as being at moderate risk of collision. Displacement, barrier effects, habitat loss or changes in prey distribution resulting from offshore wind farms were rated as low risk. Overall, kittiwake was assessed as being at low risk from offshore wind developments (Langston 2010) (Table 4.4).

Kittiwake is listed as a qualifying interest species in the breeding season for five SPAs on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development (Table 4.63). These SPAs held 20.5% of the UK breeding population and 3.2% of the biogeographic population at the time of designation (JNCC, 2012). Since designation, the populations at these SPAs have decreased (SMP, 2012). The distance between the offshore site and two SPAs is within the mean maximum foraging range of this species (60.0 km) (Forth Islands SPA and St Abb's Head to Fast Castle SPA). The distance to a further two SPAs (Buchan Ness to Collieston Coast SPA, Fowlsheugh SPA and Farne Islands SPA) is within the maximum known foraging range (120 km) (Thaxter *et al.*, 2012). Kittiwake mean maximum foraging range from breeding SPAs in relation to the offshore site are shown in Figure 4.72.

Table 4.63 SPAs for breeding kittiwake between Peterhead and Blyth

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count [2]	Year
Buchan Ness to Collieston Coast	*113*	*30,452*	*0.96*	*6.2*	*14,133*	*2007*
Farne Islands	*72*	*6,236*	*0.2*	*1.3*	*3,699*	*2009*
Forth Islands	**16**	**9,380**	**0.3**	**1.9**	**3,654**	**2009**
Fowlsheugh	*62*	*34,870*	*1.1*	*7.1*	*9,454*	*2009*
St Abb's Head to Fast Castle	**31**	**19,600**	**0.6**	**4.0**	**5,298**	**2008**
Total	**-**	**100,538**	**3.2**	**20.5**	**36,238**	**-**

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. Sites in italics lie within the maximum known foraging range of 120 km. Sites in bold lie within the mean maximum foraging range of 60.0 km (Thaxter *et al.*, 2012).

A recent JNCC statistical analysis of ESAS data investigating possible marine SPAs, identified waters to the south and east of the Neart na Gaoithe study area as an important location for kittiwakes during the breeding season (Kober *et al.,* 2010).

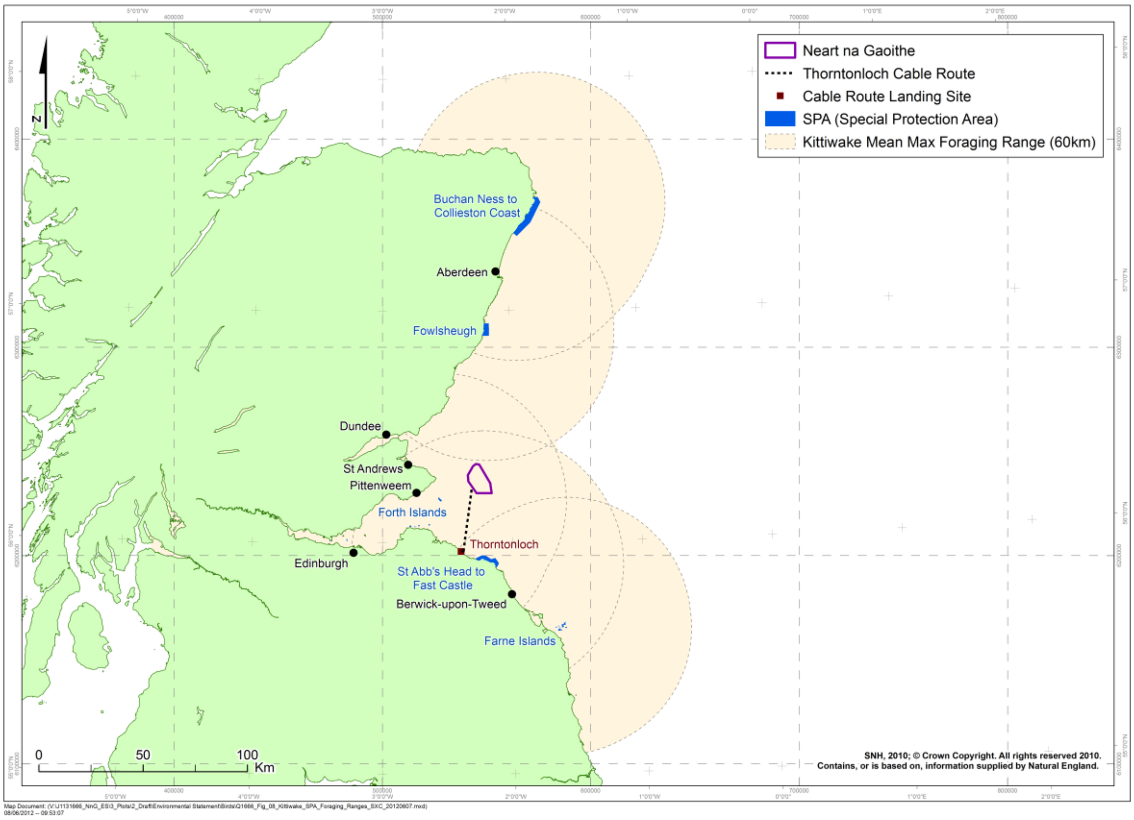


Figure 4.72 Kittiwake mean maximum foraging range from breeding SPAs in relation to the Development

4.4.24.4 Assessment

Definition of seasons

The annual cycle for kittiwake is divided into three parts to reflect the biology of the species and the broad pattern of use of the offshore site.

The breeding season, the period when breeding adults are attending colonies, is defined as April to August. At this time the vast majority of birds present in the offshore site will be from relatively local breeding colonies (i.e. those for which the offshore site is within foraging range).

The post-breeding period is defined as September and October. Although at this time of year birds are no longer breeding they are examined separately from the later part of the non-breeding period (the 'winter') because the numbers present were much greater and it is likely that the majority of individuals present at this time originate from breeding sites within the region. However, it is also likely that birds from more distant breeding colonies (e.g., from further north in Scotland and Scandinavia) may be present at this time as some individuals are known to disperse long distances soon after breeding (Wernham *et al.*, 2002).

The non-breeding period is defined as November to March (the "winter", which broadly corresponds to the period when kittiwakes are in their over-wintering area. In this period it is likely that a high proportion of individuals present in the offshore site are from breeding colonies outwith the region, including birds from other countries (Wernham *et al.*, 2002).

Baseline conditions

In the breeding season (April to August) the mean estimated number of kittiwakes present in the offshore site was 246 birds, and the mean estimated number in the offshore site buffered to 1 km was 342 birds. On average 76% of birds present in the breeding season were on the sea and the remainder were in flight. During the breeding period 94.5% of the birds present were in adult plumage.

The number of kittiwakes recorded in the breeding season during Year 2 of the baseline surveys was approximately twice that recorded in Year 1. The reason for higher numbers in Year 2 is unknown but is likely to be linked to natural variation in prey availability.

For the purposes of assessment it is assumed that all birds in full adult plumage were breeding birds. Studies have estimated that mean colony attendance rate of kittiwake during the breeding season is 37% (Wanless & Harris 1992) and this figure is used in estimates of the breeding season importance of the offshore site.

In the post-breeding period (September and October) the mean estimated number of adult kittiwakes present in the offshore site was 1,043 birds, and the mean estimated number in the offshore site buffered to 1 km was 1,250 birds. During the post-breeding period 64.0% of the birds present were aged as adults and the majority of the remainder were aged as juvenile birds.

In the non-breeding period (November to March) the mean estimated number of kittiwakes present in the offshore site was 100 birds, and the mean estimated number in the offshore site buffered to 1 km was 115 birds. During the non-breeding period 58.7% of the birds present were aged as adult birds.

Populations

Kittiwakes breeding in eastern Scotland are currently experiencing a prolonged period of population decline (SMP, 2012). Assuming that the subset of colonies that have been recently counted are representative, this amounts to a decline of approximately 31.5% since Seabird 2000, and averages nearly 4% per annum over this period. The decline in breeding numbers means that published figures on population size for this species (e.g., Mitchell *et al*., 2004) no longer accurately reflect the current population size in eastern Scotland. To prevent this causing assessment of effects to be biased low, the regional total is based on recent counts (2007 or later) (SMP, 2012), and where these are not available the Seabird 2000 results have been adjusted downwards by 31.5%. On this basis, the regional breeding population is assumed to be 121,101 breeding adults (i.e. 60,550 breeding pairs).

The size of the post-breeding-period regional kittiwake population is assumed to be the same as the regional breeding population, i.e., 121,101 adults.

The size of the regional kittiwake population in the non-breeding-period is assumed to be 68,000 birds. This is derived by summing the October to March period estimates for localities 8, and 9 and half the estimates for locality 11 given in Skov *et al*. (1995).

An unknown proportion of the regional breeding population may remain in the region through the non-breeding period but these are joined by many more birds from colonies outside the region.

Nature conservation importance

For EIA assessment purposes, the nature conservation importance of kittiwakes using the offshore site is rated as high during the breeding season. The species is classed as high NCI because a high proportion of birds using the offshore site are likely to be from the breeding colonies within the Forth Islands SPA and St Abb's Head to Fast Castle SPA, where this species is a qualifying interest. However, the mean number present in the offshore site buffered to 1 km in the breeding season are well below 1% of the (at-sea) regional population, and the species is not subject to any special legislative protection, nor is it currently on any conservation priority lists.

Offshore wind farm studies of kittiwake

Results from bird monitoring at operational wind farms indicate that kittiwakes are not likely to be displaced from offshore wind farms. Typically, studies at existing wind farms show either no significant change or small increases in kittiwake numbers compared to pre-construction numbers at these sites. At Horns Rev, Denmark, selectivity indices were significantly higher for the wind farm area during operation compared with the baseline period (Diersche and Garthe, 2006). By contrast, the compared selectivity indices for the baseline and construction periods showed that kittiwake numbers were significantly lower during the construction phase both in the wind farm and in a zone that comprised the wind farm plus a 4 km area surrounding the wind farm .(Christensen *et al*., 2003).

Post construction monitoring of kittiwakes at Egmond aan Zee, the Netherlands, showed statistically significant attraction to the wind farm during one survey with non-significant results (neither attraction or avoidance) for a further four surveys (Leopold *et al*., 2011). This study also found no behavioural evidence of gulls (including kittiwake) being displaced, with birds regularly seen flying through and sitting on the sea within the wind farm as well as resting on built infrastructure. The authors concluded that "kittiwakes seemed mostly indifferent to the wind farm" and that there was "hardly any effect of the wind farm on their distribution" (Leopold *et al*., 2011). Post-construction monitoring at Arklow Bank, Ireland reported an increase in kittiwake numbers compared to baseline numbers, concentrated within *ca.* 10 km of the turbine array (Barton *et al*., 2009). The overall increase in kittiwake numbers and their proximity to the turbines was positively associated but not significantly so (Barton *et al*., 2009)

Results of radar and visual studies indicate that flying gulls in general are not deflected around or away from wind farms. At Horns Rev, "marked behavioural reactions to the wind farm and single turbines were not observed in gull and tern species" (Christensen and Hounisen, 2005), although the proportion of 15-minute time units that kittiwakes were recorded flying between two turbines was slightly lower when one and both were active compared to when both were inactive, indicating that operational turbines may have insignificant barrier effect on kittiwakes (Petersen *et al*., 2006). Summarising the barrier effect of wind farms on seabird species occurring in German marine areas, kittiwakes were categorised as 'commonly flying through wind farms' (Diersche and Garthe, 2006).

The risk of kittiwakes colliding with wind turbines is likely to be low to moderate based on reported flying heights and recorded fatalities from operational wind farms. Of approximately 15,000 flying kittiwakes monitored over two years in the vicinity of the Arklow Bank wind farm, Ireland, less than 5% of birds were recorded flying at a height over 20 m above the sea surface (Barton *et al*., 2009, Barton *et al*., 2010). During two years monitoring at North Hoyle, Wales, a total of 31 (7%) of 466 flying birds were above 20 m above the sea surface. A single kittiwake fatality was reported in a review of the number of collision victims at wind farms in eight European countries (Hötker *et al*., 2006) although the very low probability of detecting seabird fatalities should be recognised.

EIA Construction Phase assessment for kittiwake

The construction phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

Kittiwakes are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during construction operations on the regional populations of kittiwakes in the breeding, post-breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

EIA Operational Phase assessment for kittiwake

Potential for kittiwake to be affected by displacement

For the purpose of assessing displacement it is assumed that 25% of kittiwakes will be displaced from the proposed wind farm footprint and a surrounding buffer of 1 km.

Breeding season

The value of the offshore site as a foraging site for breeding kittiwakes was estimated for each of the receptor populations from the proportion of the adults likely to be at sea (not attending a colony) that were on average present in the offshore site during the breeding season (Table 4.64). This is not a precise measure of foraging value but it is considered to be a reasonable measure in the absence of more detailed information on behaviour. Indeed because a high proportion of kittiwakes seen on baseline surveys were apparently not actively foraging but merely flying over the offshore site, it is likely that the measure is biased high and thus leads to cautious assessment conclusions.

Table 4.64 The mean estimated number of kittiwakes present during the breeding, post-breeding and non-breeding period, and the value as a percentage of the (at-sea) receptor population potentially at risk of displacement

Receptor population	Population size (adults)	No. assumed at sea	Dev. Area		Dev. Area + 1 km	
			Mean no. at risk	%	Mean no. at risk	%
National, colony-attend period (Seabird 2000)	734,550	462,766	246	<0.1	342	<0.1
National, post-breeding period (Seabird 2000)	734,550	734,550	1,043	0.1	1,250	0.2
North Sea, non-breeding period (Skov *et al*. 1995)	1,032,690	1,032,690	100	<0.1	115	<0.1
Regional, colony-attend period (Seabird 2000 x 32% decline)	121,101	76,293	246	0.3	342	0.4
Regional, post-breeding period (Seabird 2000 x 32% decline)	121,101	121,101	1,043	0.9	1,250	1.0
Regional, non-breeding period (Skov *et al*. 1995)	68,000	68,000	100	0.1	115	0.2

On this basis, it is inferred that the maximum potential for displacement from the offshore site buffered to 1 km during the breeding period would be a 0.4% reduction in access to the foraging resources of the regional breeding population (Table 4.64).

Post-breeding period

The value of the offshore site buffered to 1 km as a foraging site for kittiwakes in the post-breeding period was estimated in the same way as for the breeding period. On this basis, it is inferred that the maximum potential for displacement from the offshore site buffered to 1 km during the post-breeding period would be a 1.0% reduction in access to the foraging resources of the regional post-breeding population (Table 4.64).

Non-breeding period

The value of the offshore site buffered to 1 km as a foraging site to kittiwakes during the non-breeding "winter" period was estimated in the same way as for the breeding period. On this basis, it is inferred that the maximum potential for displacement from the offshore site buffered to 1 km during the non-breeding period would be a 0.2% reduction in access to the foraging resources of the regional population (Table 4.64).

Likely impacts of displacement on kittiwake population

Kittiwakes show low levels of displacement from offshore wind farms and therefore it is very unlikely that the potential displacement effects quantified above will be fully realised.

Breeding season

If 25% of kittiwakes were to be displaced during the breeding season from the offshore site buffered to 1 km the impact of this would be the effective loss of up to 0.1% of the foraging habitat of the regional population. This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of displacement on the regional kittiwake population in the breeding season are ***not significant*** under the EIA Regulations.

Post-breeding period

If 25% of kittiwakes were to be displaced during the post-breeding period from the offshore site buffered to 1 km the impact of this would be the effective loss of up to 0.25% of the foraging habitat of the regional population. This impact is categorised as negligible magnitude**,** and temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of displacement on the regional kittiwake population in the post-breeding period are ***not significant*** under the EIA Regulations.

Non-breeding period

If 25% of kittiwakes were to be displaced during the non-breeding period from the offshore site buffered to 1 km the impact of this would be the effective loss of up to 0.1% of the foraging habitat of the regional population. This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of displacement on the regional kittiwake population in the non-breeding period are ***not significant*** under the EIA Regulations.

Barrier effect

Kittiwakes breeding at nearby colonies to the proposed wind farm could be affected by the wind farm acting as a barrier. The greatest potential for such impacts is to birds from the Isle of May, Craigleith and St Abb's Head colonies.

Kittiwakes are considered to have low sensitivity to barrier effects on account of a low wing loading (Maclean *et al.*, 2009, Langston, 2010). The potential effects on kittiwake of the proposed wind farm acting as a barrier are assessed for the breeding season, when birds are attending colonies.

For the purposes of assessment the width of the barrier is assumed to extend 1 km either side of the maximum width of the proposed wind farm. Observations from operational offshore wind farms shows no evidence that wind farms pose a barrier to kittiwakes. It is therefore likely that only a small percentage, of foraging flights potentially intercepted by the barrier would be affected. It is assumed that only 25% of birds reaching the barrier will respond by detouring around the wind farm. Barrier effects as calculated here concern birds which would otherwise fly through the offshore site to access feeding resources beyond it.

For birds breeding on the Isle of May colony, the wind farm acting as a barrier would potentially block approximately 33% of the possible flight directions (Table 3.6). Assuming a mean destination distance of 30 km (immediately beyond the barrier), the mean increase in the length of barrier-affected flights is estimated at 17.7% (Table 3.7). For birds breeding on Craigleith, the wind farm acting as a barrier would potentially block approximately 28% of the possible flight directions (Table 3.6). Assuming a mean destination distance of 45 km (immediately beyond the barrier), the mean increase in the length of barrier-affected flights is estimated at 8.7 % (Table 3.7). For birds breeding at St Abb's Head, the wind farm acting as a barrier would potentially block approximately 9% of the possible flight directions (Table 3.6). Assuming a mean destination distance of 45 km (immediately beyond the barrier), the mean increase in the length of barrier-affected flights is estimated at 5.3% (Table 3.7).

However, less than 50% of kittiwake flights potentially affected by the barrier are likely to have an intended destination beyond the wind farm as the mean distance of foraging flights is 24.8 km (Thaxter *et al*., 2012), and only a minority of these (assumed to be 25%) are likely to respond by detouring around the wind farm. It is concluded that <5% of foraging flights from these colonies are likely to be affected by barrier effects and the size of detours is relatively small. The likely impacts arising from the proposed wind farm acting as a barrier to kittiwakes breeding at other colonies were not examined in detail because, on the basis of the results for the Isle of May, Craigleith and St Abb's Head, the greater distances and the smaller number of birds involved, it is clear that any impact would be negligible and could not plausibly make barrier effects an issue of significance for the regional breeding population.

The potential for the development to act as a barrier and increase the length and duration of foraging trips for birds of the regional population in the breeding season is an effect that is of low magnitude (<1 %) and temporally long-term and reversible (Table 3.1 and Table 3.2). Bearing in mind the low sensitivity of this species to barrier effects, it is concluded that any barrier effect on the regional kittiwake population in the breeding season is ***not significant*** under the terms of the Electricity Act.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

Kittiwakes are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional populations of kittiwakes in the breeding, post-breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

Collision mortality

For the purposes of assessment it is assumed that there will be no far-field avoidance of the proposed wind farm by kittiwakes. Therefore, the mortality that is considered likely to occur for wind farm designs and assumed avoidance rate scenarios illustrated is that presented in Table 4.65 and Table 4.66*)* (see Appendix 12.2 for further details). COWRIE guidance

(Maclean *et al.,* 2009) is to use a default avoidance rate of 99.5% for gull species. There is no specific SNH guidance on avoidance rates for seabirds, and therefore the default value of 98.0% recommended by SNH is applicable.

6.0% of flying kittiwakes observed in baseline surveys were within the height range swept by the proposed turbine rotors (Table 4.65).

It is assumed that all birds present during the breeding season are from the regional breeding population. In the non-breeding period (including the post-breeding period) the kittiwakes present in the offshore site will comprises a mixture of birds originating from regional breeding colonies and more distant colonies, including overseas colonies (Wernham *et al.*, 2002). Furthermore, many, perhaps most, of the birds breeding in the region overwinter outside the region (Wernham *et al.*, 2002). To estimate the effect of year-round collision mortality on the regional breeding population the collisions that occur in the non-breeding period and that affect birds from the regional breeding population needs to be estimated. To do this it is cautiously assumed on the basis of the ringing information (Wernham *et al.,* 2002) that half the birds present in the non-breeding period originate from the regional breeding population. The effect of collision mortality on the adult mortality rate for the regional breeding population is calculated by summing the breeding season mortality and the non-breeding period mortality attributable to birds from the region.

Given the large and continuing decline of the regional kittiwake population (by approximately 4% per year (SMP, 2012), equivalent to approximately 4,844 breeding adults every year), this population is considered to have high sensitivity to any additional adult mortality.

Table 4.65 The mean estimated proportion of kittiwakes estimated to be at rotor height and the number of collisions predicted by collision rate modelling (CRM) using avoidance rates (AR) of 98.0% and 99.5% for the least adverse (64 x 7 MW turbines) and most adverse (128 x 3.6 MW turbines) wind farm designs evaluated. LAT is sea level at lowest astronomical tide, and this is approximately 2.0 m below the average sea level

Metric	**Design 1, 64 x 7MW**		**Design 4, 128 x 3.6 MW**	
	98.0% AR	**99.5% AR**	**98.0% AR**	**99.5% AR**
Rotor swept height range (metres above LAT)	24 - 188		24 - 144	
Flight activity at rotor height	94.0%		94.0%	
Flight activity below rotor height (<22.5 m above sea)	6.0%		6.0%	
Collisions in breeding season (April to August), all ages	40.2	10.1	60.1	15.0
Collisions in non-breeding period (September to March), all ages	73.5	18.4	109.8	27.5
Total collisions per year, all ages	113.8	28.4	169.9	42.5

Table 4.66 The effect of CRM predicted collision mortality on the adult mortality rate (AMR) of the regional breeding kittiwake population

Wind farm Design Period of year	**% birds present assumed to be from regional breeding popn**	**98.0% OAR**		**99.5% OAR**	
		CRM deaths	**Change AMR**	**CRM deaths**	**Change AMR**
Design 1, 64 x 7MW turbines					
Breeding period (April - August)	100%	38	0.5%	10	0.1%
Non-breeding period (Sept. - March)	50%	22	0.3%	6	0.1%
Whole year	varies	60	0.8%	15	0.2%
Design 4, 128 x 3.6MW turbines					
Breeding period (April - August)	100%	57	0.8%	14	0.2%
Non-breeding period (Sept. - March)	50%	33	0.5%	8	0.1%
Whole year	varies	90	1.3%	23	0.3%

Using an avoidance rate of 99.5% the increase in the adult annual mortality rate of the regional breeding population would be between 0.2% and 0.3% depending on the wind farm design (Table 4.66). Using an avoidance rate of 98.0% the increase is between 0.8% and 1.3%.

Using a 99.5% avoidance rate, the predicted changes to the adult annual mortality rate are in all cases well below the default guidance threshold of 1%, considered the minimum likely to have potential for adverse impacts on a population (King *et al.,* 2009). However, when assessed using a 98% avoidance rate and the most adverse design (Wind Farm Design 4) the predicted change in mortality rate of 1.3% exceeds the guidance threshold of 1%. The prediction of 0.8% increase in the mortality rate for Wind Farm Design 1 approaches the 1% threshold. Considering the baseline population is currently declining, additional mortality of around 1% is likely to exacerbate the rate of population decline.

Assessed for a 99.5% avoidance rate the potential impact of the predicted collision mortality on the regional population in the breeding season is an effect of negligible magnitude (<1%) and temporally long-term and reversible. It is concluded that the effects of collision mortality on the regional kittiwake population in the breeding season is ***not significant*** under the terms of the EIA Regulations.

Assessed for a 98.0% avoidance rate the potential impact of the predicted collision mortality on the regional population in the breeding season is an effect of low magnitude (<1%) and temporally long-term and reversible. Using a 98.0% avoidance rate the impact of collision mortality on the regional population of kittiwakes in the breeding population is an effect of ***minor significance*** under the terms of the EIA Regulations.

EIA Decommissioning Phase assessment for kittiwake

The decommissioning phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

Kittiwakes are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during decommissioning operations on the regional populations of kittiwakes in the breeding, post-breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

EIA summary of effects combined for kittiwake

The adverse impacts on the regional kittiwake population in the breeding season of the effects assessed will act in a broadly additive manner. In combination, and using a collision avoidance rate of 99.5%, it is judged that the combined magnitude of the three effects is negligible (Table 4.67). Furthermore, the population has low sensitivity to all effects except collision risk. It is concluded that the overall operational phase impact of the development on the regional kittiwake population in the breeding season is ***not significant*** under the EIA Regulations.

Using a collision avoidance rate of 98.0%, it is judged that the combined magnitude of the three effects is low (Table 4.67) and it is concluded that the overall operational phase impact of the development on the regional kittiwake population in the breeding season is of ***minor significance*** under the EIA Regulations.

Table 4.67 Summary of effects on the regional breeding population of kittiwakes. The effects on the adult mortality rate due to collision includes collision deaths occurring outside the breeding period and attributed to birds from the regional breeding population

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Low	***Not significant***
Barrier Effect	Negligible	Long term	Low	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality using 98.0% A R	Low	Long term	Moderate	***Minor significance***
Collision mortality using 99.5% A R	Negligible	Long term	Moderate	***Not significant***
All effects combined (for 98.0% AR)	Low	Long term	Moderate	***Minor significance***
All effects combined (for 98.0% AR)	Negligible	Long term	Low/Moderate	***Not significant***

The adverse impacts on the regional kittiwake population in the non-breeding period of the effects assessed will act in a broadly additive manner. Using an avoidance rate either 98.0% or 99.5%, it is judged that the combined magnitude of the three effects is negligible (Table 4.68). It is concluded that the overall operational phase impact of the development on the regional kittiwake population in the non-breeding period is ***not significant*** under the EIA Regulations.

Table 4.68 Summary of effects on the regional population of kittiwakes present in the non-breeding period

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat, post-breeding period	Negligible	Long term	Low	***Not significant***
Displacement from foraging habitat, winter period	Negligible	Long term	Low	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality (using either 98.0% or 99.5% A R)	Negligible	Long term	Moderate	***Not significant***
All effects combined	Negligible	Long term	Low	***Not significant***

4.4.24.5 Cumulative Impact Assessment for kittiwake

Combining the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts on the regional population of kittiwake in the breeding season will be as shown in Table 4.69. The individual impacts for displacement and collision are combined by addition to give the cumulative impact. The flight directions from the various colonies that are potentially affected by each wind farm acting as a barrier substantially overlap and therefore the cumulative barrier impact from the three developments is not the sum of the individual predicted impacts. The cumulative barrier impact is derived from the overall spread of flight directions at each colony potentially affected by the three developments forming barriers.

The predicted potential effective loss of up to 1.7% of the foraging habitat of the regional kittiwake population in the breeding season (Table 4.69) is rated as an effect of low magnitude, temporally long-term and reversible. It is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional population of kittiwake in the breeding season is of ***minor significance*** under the terms of the EIA Regulations.

Table 4.69 Summary of CIA for the three proposed offshore wind farm in south-east Scotland on the regional population of breeding kittiwakes

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF	Predicted impact from Inch Cape pOWF	Predicted impact from Outer Forth pOWF	Cumulative Impact
Displacement: effective loss of foraging habitat during breeding period (%)	25%	0.1%	0.6	1.0	≤1.7
Displacement: effective loss of foraging habitat during post-breeding period (%)	25%	0.3%	0.2	0.4	≤0.9
Barrier Effect	25%	Foraging trips of <5% off Forth Islands & St Abb's Head birds increase by 6%, Negligible	Foraging trips of <5% of Fowlsheugh & Forth Islands increase by ca. 6%. Negligible	No flights likely to go beyond offshore area, Negligible	No flights likely to go beyond offshore site, Negligible
Collision most adverse designs: % increase in assumed annual adult mortality rate with 98.0% avoidance rate.	0%	1.3%, Low	18.3%, Moderate	16.7%, Moderate	36.3%, High
Collision least adverse designs: % increase in assumed annual adult mortality rate with 98.0% avoidance rate.	0%	0.9%, Negligible	0.2%, Negligible	10.4%, Moderate	11.5%, Moderate
Collision most adverse designs: % increase in assumed annual	0%	0.3%, Negligible	4.4%, Negligible	4.2%, Low	8.9%, Moderate

adult mortality rate with 99.5% avoidance rate.					
Collision least adverse designs: % increase in assumed annual adult mortality rate with 99.5% avoidance rate.	0%	0.2%, Negligible	<.1%, Negligible	2.6%, Low	2.8%, Low

Barrier effects as assessed here concern birds which would otherwise fly through the offshore site to access feeding resources beyond it. The mean foraging distance from the colony for kittiwakes is 24.8 km and the mean maximum foraging distance is 60.0 km (Thaxter *et al*., 2012). This means that almost no foraging flights by kittiwake will be to areas beyond the barrier formed by the three proposed wind farms. As a result any barrier effect will be negligible. The cumulative barrier effect is rated as negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). Bearing in mind the low sensitivity of kittiwake to barrier effects, it is concluded that the cumulative impact of a barrier effect caused by the three proposed offshore wind farms on the regional population of kittiwakes in the breeding season is ***not significant*** under the terms of the EIA Regulations.

Conclusion on the cumulative impact of collision mortality are sensitive to the wind farm designs evaluated and the level of avoidance rate used for predictive calculations of the number of collision strikes.

Using an avoidance rate of 98.0% the cumulative effect of the three wind farms is to increase the annual adult mortality rate by 11.5% for the least adverse wind farm designs and by 36.3% for the most adverse designs. Assuming that the least adverse designs are chosen, the potential cumulative impact of this additional mortality on the regional kittiwake population in the breeding season is rated as moderate magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that, using a 98.0% avoidance rate, the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional population of kittiwakes in the breeding season is of ***moderate significance*** under the terms of the EIA Regulations.

Using an avoidance rate of 99.5% the cumulative effect of the three wind farms is to increase the annual adult mortality rate by 2.8% for the least adverse wind farm designs, and by 8.9% for the most adverse designs. Assuming that the least adverse designs are chosen, the potential cumulative impact of this additional mortality on the regional kittiwake population in the breeding season is rated as low magnitude (1-5%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that, when assessed using a 99.5% avoidance rate, the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional population of kittiwakes in the breeding season is of ***minor significance*** under the terms of the EIA Regulations.

A separate CIA has not been undertaken for the regional kittiwake population in the non-breeding-period. This is because the cumulative assessment above for collision mortality on the regional breeding population includes collision deaths for the whole year. It assumes that 50% of the birds killed outside the breeding period are from the regional breeding population. The predicted displacement of kittiwakes due to the Neart na Gaoithe wind farm in the non-breeding period is negligible and the sensitivity of the population to displacement at this time of year is also negligible. Therefore it is not plausible that the Neart na Gaoithe Development could contribute to a significant cumulative impact for this receptor population.

4.4.24.6 Mitigation measures for kittiwake

The assessment does not identify any significant adverse effects on the regional populations of kittiwake in the breeding, post-breeding and non-breeding periods. Therefore no mitigation measures are required for this species.

4.4.25 Common tern *Sterna hirundo*

4.4.25.1 Status

Common terns are summer visitors to Britain, breeding in colonies at coastal sites and also inland. Seabird 2000 recorded 10,308 pairs in Britain (Mitchell *et al.,* 2004). The closest large colonies to the Neart na Gaoithe study area are Leith Docks in Edinburgh (789 pairs in 2008) and the Isle of May (40 pairs in 2009) (SMP, 2012). Common terns have a broad diet compared to other tern species, including sandeels, clupeid and gadoid fish (Mitchell *et al.,* 2004).

4.4.25.2 Neart na Gaoithe Study Area

Low numbers of common terns were recorded in the Neart na Gaoithe study area in Year 1, with a total of 13 birds recorded between July and November. Highest numbers occurred in September, when 10 birds were recorded. Birds were predominantly in the northern half of the study area, with three recorded in the offshore site in September, and the remainder seen in the buffer area (Table 4.2). Numbers of common terns in Year 2 were slightly higher, with a total of 50 birds recorded, again mostly in September. The majority of birds were in the north of the buffer area at this time (35), with 13 birds in the offshore site (Figure 4.73).

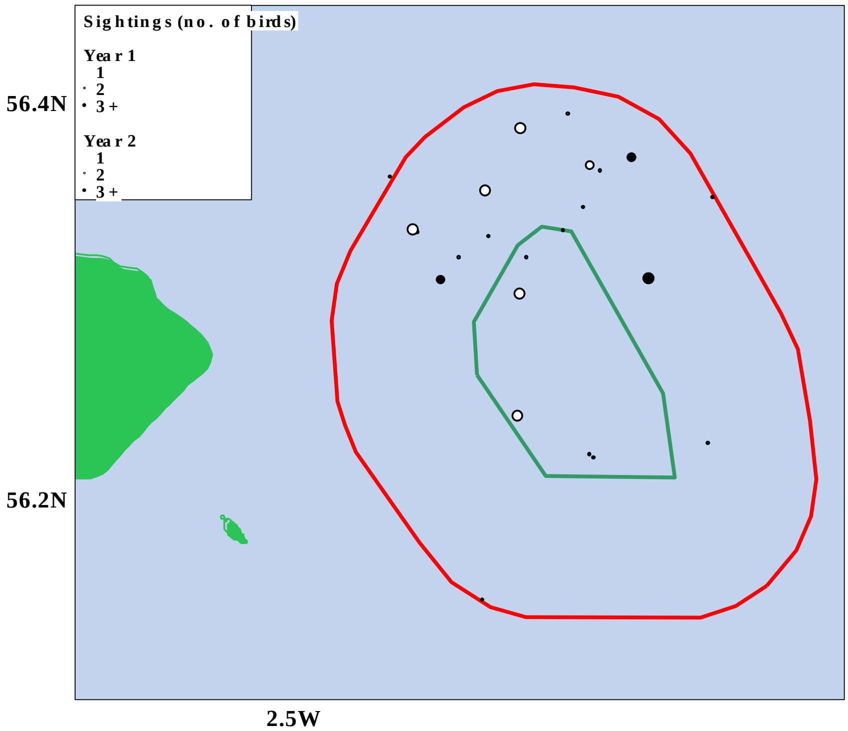


Figure 4.73 Common tern sightings between May and November in Years 1 and 2

In Years 1 and 2, a total of 35 common terns were recorded in flight, with all birds flying below 17.5 m in height (Table 4.3).

4.4.25.3 Species sensitivity

Common tern is listed on Annex I of the EU Birds Directive (2009/147/EEC), and the species is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009).

A recent review assessed common tern as being at moderate risk of collision and habitat loss or changes in prey distribution resulting from offshore wind farms. Displacement and barrier effects were rated as low risk. Overall, common tern was assessed as being at moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

Common tern is listed as a qualifying interest species in the breeding season for five SPAs on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development (Table 4.70). These SPAs held 21.6 % of the UK breeding population and 1.3 % of the biogeographic population at the time of designation (JNCC, 2012). The distance between the offshore site and one SPA (Forth Islands SPA) is within the maximum known foraging range (30 km) but just beyond the mean maximum foraging range of this species (15.2 km) (Thaxter *et al.*, 2012). Common tern mean maximum foraging range from breeding SPAs in relation to the offshore site are shown in Figure 4.74.

Table 4.70 SPAs for common tern in the breeding season between Peterhead and Blyth

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count [2]	Year
Coquet Island	106	740	0.4	6.0	1,228	2009
Farne Islands	72	230	0.1	1.9	98	2009
Firth of Forth Islands	*16*	*800*	*0.4*	*6.5*	*101*	*2008*
Ythan Estuary, Sands of Forvie & Meikle Loch	110	265	0.1	2.2	4	2010
Leith Docks	62	789	0.3	5.0	789	2008
Total	**-**	**2,824**	**1.3**	**21.6**	**2,220**	**-**

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. Sites in italics lie within the maximum known foraging range of 30 km. Sites in bold lie within the mean maximum foraging range of 15.2 km (Thaxter *et al.*, 2012).

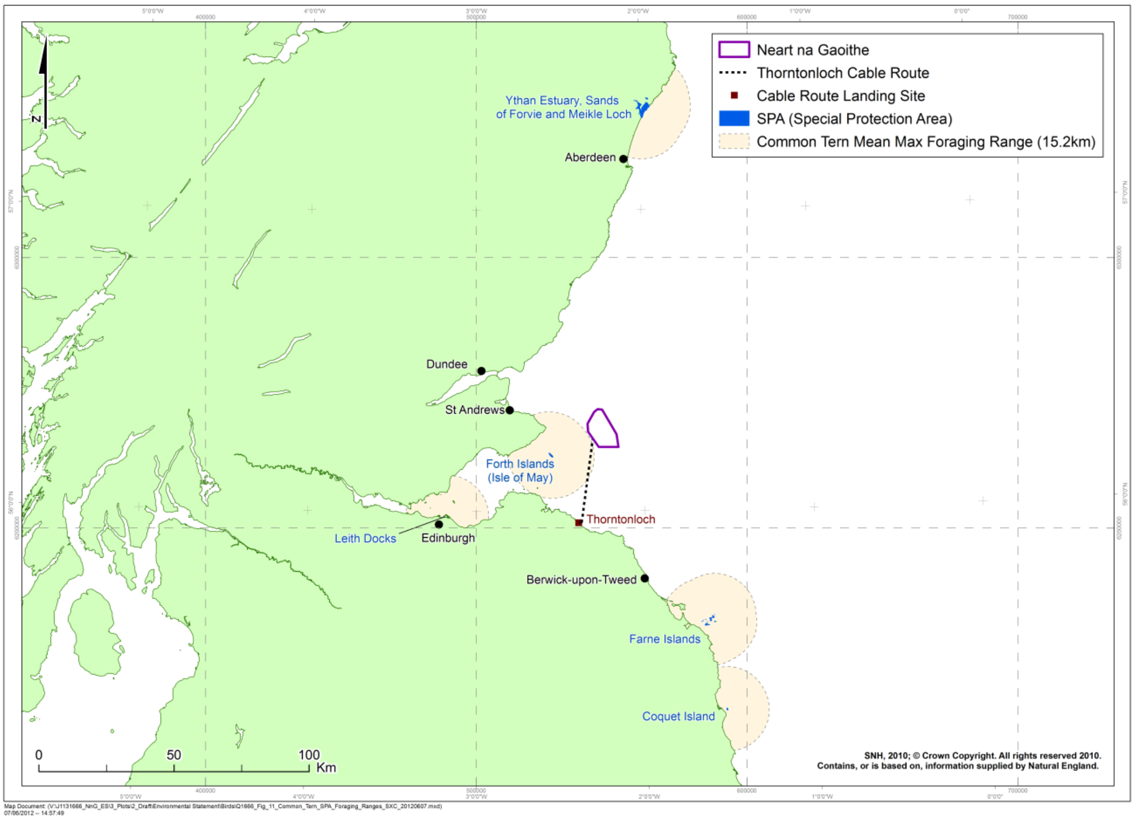


Figure 4.74 Common tern mean maximum foraging range from breeding SPAs in relation to the Development

4.4.25.4 Assessment

Common terns were only recorded on baseline surveys between July and November, with the majority of birds seen in September during autumn passage. Low numbers of common terns were recorded within the offshore site during the baseline surveys, with a total of three birds in Year 1 and 13 birds in Year 2, all recorded in September.

Populations

There is no published estimate of the size of the regional autumn passage population of common tern. The EIA assessment was therefore based on the number breeding in eastern Scotland and north-east England; approximately 4,063 pairs or 8,126 breeding adults based on the Seabird 2000 census (Mitchell *et al*., 2004).

Displacement

It is not known whether common terns will be displaced by the proposed Neart na Gaoithe development. The review undertaken by Langston (2010) suggested that common terns are at low risk of displacement effects, while the very low numbers of birds recorded in the offshore site on baseline surveys suggest that few would be affected were displacement effects to occur. Overall, it is concluded that the effects of displacement on common terns is ***not significant*** under the terms of the EIA Regulations.

Barrier effect

The largest breeding colony of common terns is at Leith Docks in Edinburgh (789 pairs in 2008) and the Isle of May (40 pairs in 2009) (SMP, 2012). The distance between the colony at Leith Docks and the offshore site (62 km) is greater than the maximum known foraging range (30 km) (Thaxter *et al*., 2012). Therefore no barrier effects on breeding birds from this colony are predicted to occur during the breeding period.

The distance between the Isle of May colony and the offshore site (16 km) is just outside the mean maximum foraging range of common tern (15.2 km), but within the maximum recorded foraging range (30 km) (Thaxter *et al.,* 2012). This suggests that some common terns from the Isle of May could forage in the Developent Area in the breeding season, however as common terns were only recorded in September on baseline surveys this is unlikely to be the case. Based on the absence of common terns in the offshore site in the breeding season, it is concluded that the effects of barrier effect on common terns is ***not significant*** under the terms of the EIA Regulations.

Collision Mortality

Baseline surveys recorded a total of 35 common terns in flight, with all birds flying below 17.5 m in height (Table 4.3).

Collision risk modelling was undertaken for common tern based on an assumed population of 1,000 birds passing through the offshore site twice per year, once in spring and autumn, with an estimated 13% of birds at flight height (Cook *et al.*, 2012). This resulted in a total of 0.5 collisions predicted per year, based on an avoidance rate of 98.0%, or 0.1 collisions predicted per year, based on an avoidance rate of 99.5%. Further details are presented in Appendix 12.2.

Based on a baseline annual adult mortality rate of 10.0% (BTO, 2012), the additional collision mortality would lead to an increase of 0.5% in the annual adult mortality rate (i.e., to 10.5% p.a.) for collisions predicted using a 98.0% avoidance rate, and to an increase of 0.12% for collisions predicted for a 99.5% avoidance rate. In both case the effect is of negligible magnitude.

Scaling this up to the size of the common tern population of 8,126 adults that might pass through would give a worst case scenario of 40 and 9 collisions per annum for 98.0% and 99.5% avoidance rates respectively. The effect of this on the population's adult mortality rate remains the same as shown above.

This worst case scenario assumes that all birds in the eastern Scotland and north-east England breeding population pass through the wind farm at rotor height twice a year, which is extremely unrealistic for two reasons. First, in actuality a smaller proportion of common tern flight activity would be at rotor height, indeed, 0% of 35 flying common terns recorded in baseline surveys were above the proposed rotor height, compared to 13% assumed here. Second, common terns migrate on a relatively broad front that is wider than the offshore site. Therefore, only a relatively small proportion (say, <25%) of the population would be expected to pass through the offshore site, although numbers recorded in the offshore site on baseline surveys suggest that 25% would still be a considerable overestimate. It is likely therefore that the actual effects of collision mortality on migrating common terns is considerably lower than the worst case scenario figures presented above.

Based on this, the potential effect of collision mortality of common terns on the baseline mortality rate is rated as negligible in magnitude, temporally long-term and reversible. It is concluded that the effects of collision mortality on common terns is therefore ***not significant*** under the terms of the EIA Regulations.

4.4.26 Arctic tern *Sterna paradisaea*

4.4.26.1 Status

Arctic terns are summer visitors to Britain, breeding in colonies at coastal sites and also inland. Seabird 2000 recorded 56,123 breeding pairs in Britain (Mitchell *et al.,* 2004). The closest large colony to the Neart na Gaoithe study area is the Isle of May, with 365 pairs in 2009 (SMP, 2012). Sandeels are the major prey species (Mitchell *et al.,* 2004). The Firth of Forth and Firth of Tay are known to be important areas for terns on autumn passage (Forrester *et al.,* 2009).

4.4.26.2 Neart na Gaoithe Study Area

In Year 1, a total of 857 Arctic terns were recorded on surveys in the Neart na Gaoithe study area. Approximately three quarters of all birds (76.6%) were recorded in the buffer area in August (Table 4.2). Fewer birds were recorded in Year 2 (329 birds), although most birds (88.8%) were again in the buffer area in August and September. It is likely that these birds were passing through the Neart na Gaoithe study area on southward migration at the end of the breeding season. The majority of aged birds were adults (95.7%).

Due to the low sample size of Arctic terns recorded in Years 1 and 2, it was not possible to conduct Distance analysis on the data. Simple abundance rates (birds/km) were calculated instead.

Mean monthly Arctic tern abundance was generally low in the offshore site and the buffer area in Years 1 and 2, apart from in August (Figure 4.75). Highest mean abundance was recorded in August of Year 1, with a peak of 4.3 birds/km in the offshore site, compared to 2.7 birds/km in the buffer area. Peak mean abundance in August of Year 2 was lower. ESAS abundance data from the surrounding ICES rectangles and across Regional Sea 1 was very low.

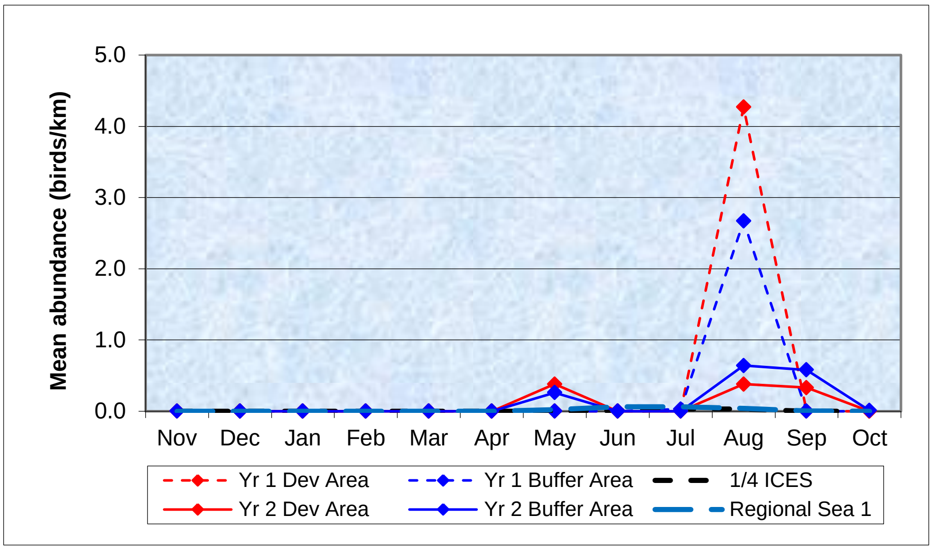


Figure 4.75 Comparison of Arctic tern monthly mean abundance in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

Arctic terns were scattered sporadically in the offshore site and buffer area at low abundances between April and September of Year 1 (Figure 4.76). Few birds were recorded to the south of the offshore site over the period.

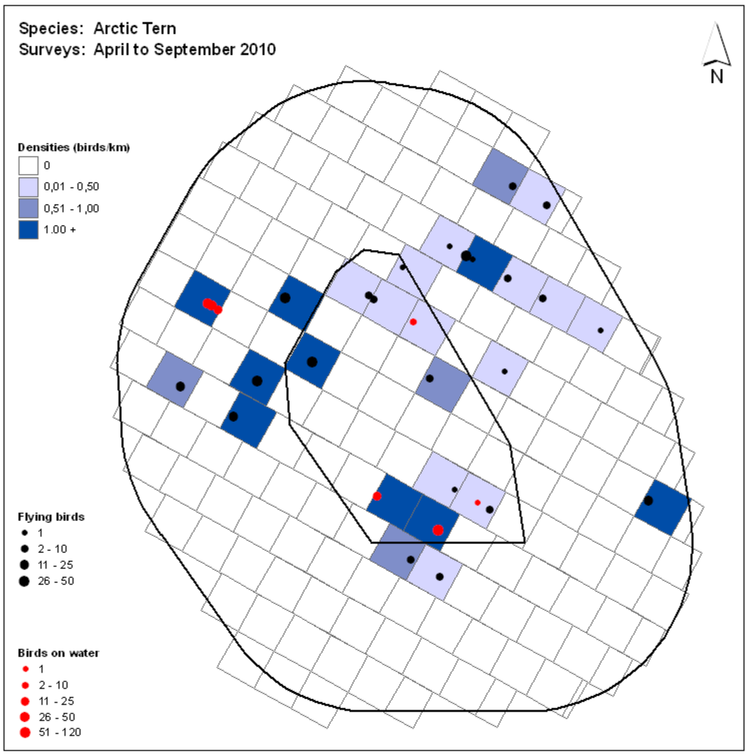


Figure 4.76 Arctic tern abundance between April and September, Year 1

Over the same period in Year 2, Arctic tern distribution was broadly similar to Year 1, although fewer birds were recorded in the offshore site over the period (Figure 4.77).

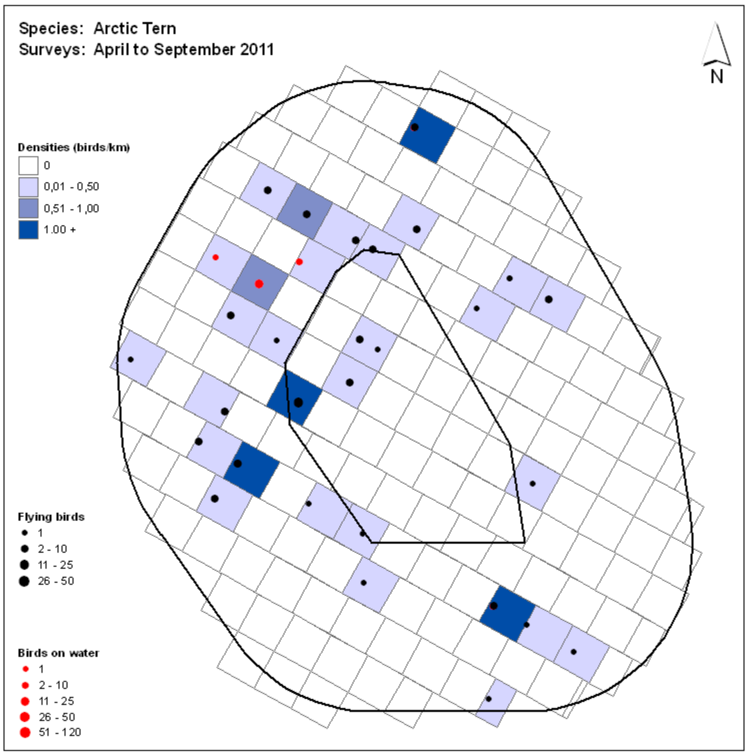


Figure 4.77 Arctic tern abundance between April and September, Year 2

A total of 964 Arctic terns were recorded in flight in Years 1 and 2, with almost all birds (99.6%) flying below 22.5 m in height (Table 4.3). A total of four birds (0.4%) were recorded flying above 22.5 m, at estimated heights of 25 and 30 m. In addition, a further 210 unidentified common/Arctic terns and 34 unidentified tern species were recorded in flight in Years 1 and 2. All birds were recorded flying below 22.5 m.

Foraging behaviour was recorded for 624 Arctic terns in the Neart na Gaoithe study area in Year 1, with five types of foraging behaviour recorded (Table 4.71). The majority of all foraging birds were recorded actively searching (49.0%) and dipping (51.0%). Foraging behaviour was recorded for a further 149 unidentified common/arctic terns, with 47.0% of birds actively searching and 53.0% of birds dipping.

Table 4.71 Arctic tern foraging behaviour in the Neart na Gaoithe Study Area in Years 1 and 2

Behaviour	Number of birds
Actively searching	306
Deep plunging	7
Dipping	309
Shallow plunging	1
Surface pecking	1
Total	**624**

4.4.26.3 Species sensitivity

Arctic tern is listed on Annex I of the EU Birds Directive (2009/147/EEC), and the species is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

A recent review assessed Arctic tern as being at moderate risk of collision and habitat loss or changes in prey distribution resulting from offshore wind farms. Displacement and barrier effects were rated as low risk. Overall, Arctic tern was assessed as being at moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

Arctic tern is listed as a qualifying interest species in the breeding season for three SPAs on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development (Table 4.72). These SPAs held 9.3% of the UK breeding population and > 0.4% of the biogeographic population at the time of designation (JNCC, 2012). The distance between the offshore site and one SPA (Forth Islands SPA) is within the mean maximum foraging range of Arctic tern (24.2 km) (Thaxter *et al.*, 2012). Arctic tern mean maximum foraging range from breeding SPAs in relation to the offshore site are shown in Figure 4.78.

Table 4.72 SPAs for Arctic tern in the breeding season between Peterhead and Blyth

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count [2]	Year
Coquet Island	106	700	0.1	1.6	1,259	2009
Farne Islands	72	2,840	0.3	6.5	2,198	2009
Firth of Forth Islands	**16**	**540**	**<0.1**	**1.2**	**365**	**2009**
Total	**-**	**4,080**	**>0.4**	**9.3**	**3,822**	**-**

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. Sites in italics lie within the maximum known foraging range of 30 km. Sites in bold lie within the mean maximum foraging range of 24.2 km (Thaxter *et al.*, 2012).

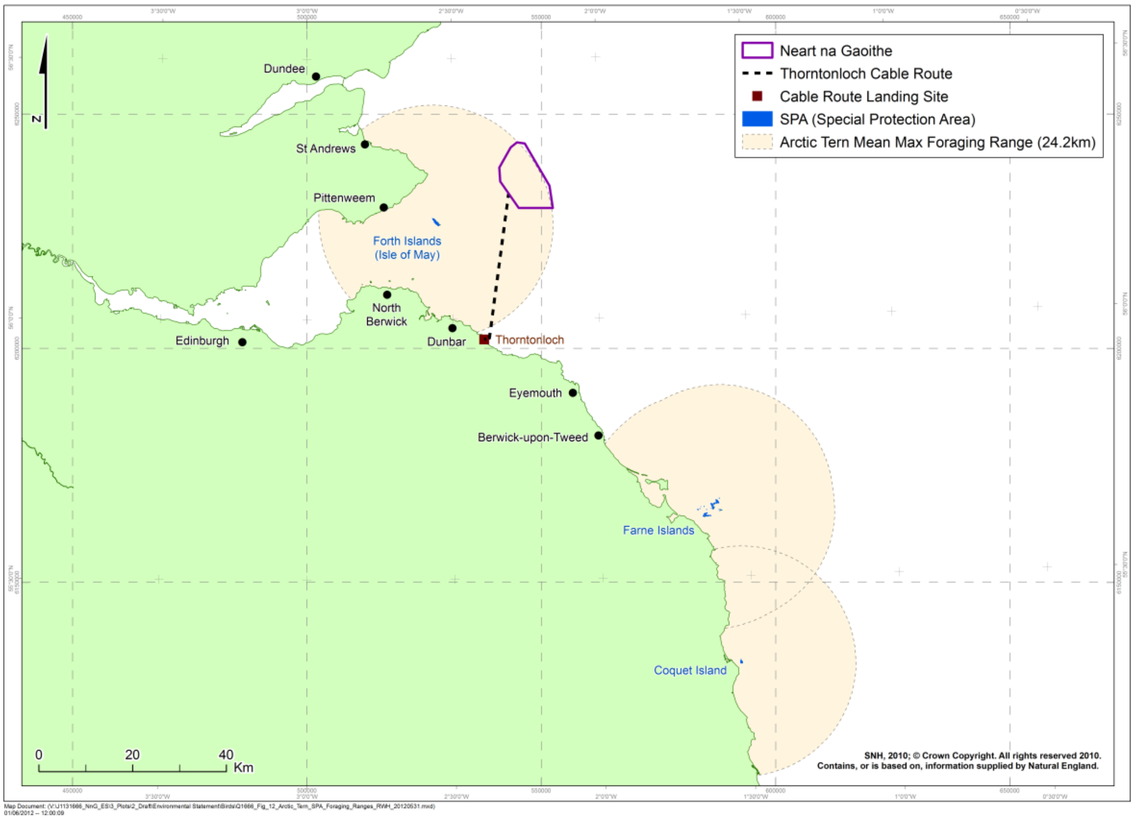


Figure 4.78 Arctic tern mean maximum foraging range from breeding SPAs in relation to the Development

4.4.26.4 Assessment

Definition of seasons

Arctic terns were mainly recorded in the Survey Area in August and September, which is the autumn passage period of the year for this species. Small numbers were also seen in May, these are likely to have been spring passage birds although it is also possible that some or all these birds were breeding locally.

Baseline conditions

In the autumn passage period the estimated number of Arctic terns present in the offshore site was 337 birds, and the estimated number in the offshore site buffered to 1 km was 388 birds. In addition, in this period there were on average a further 26 and 37 unidentified common/Arctic terns present in these areas, respectively. It is likely that the vast majority of these unidentified birds were Arctic terns, and for the purposes of assessment it is assumed that they were this species.

In the spring passage period (May) the estimated number of Arctic terns present in the offshore site was 20 birds, and the estimated number in the offshore site buffered to 1 km was 29 birds. No unidentified common/Arctic terns were recorded in the spring passage period.

Nature conservation importance

The Nature Conservation Importance of Arctic tern using the offshore site is rated as high. The species merits high NCI because it is on Annex 1 of the EU Birds Directive. It is also likely that a high proportion of individuals present in the autumn passage period are from SPA designated breeding sites where this species is a qualifying interest, including the Forth Islands SPA. The mean number present in the autumn passage period in the offshore site buffered to 1 km exceeds 1% of the regional breeding population.

Populations

There is no published estimate of the size of the regional autumn passage population of Arctic tern. The regional (Peterhead to Blyth) breeding population size is assumed to be 10,056 birds (5,028 pairs) (SMP, 2012). However, the birds present in the autumn passage period are likely to originate from a much wider area, including Orkney and Shetland, and possibly Scandinavian colonies. Therefore, the size of the regional population in the autumn passage period may be much greater than during the breeding season.

A more appropriate population to base the EIA assessment on is the number breeding in eastern Scotland and north-east England; approximately 98,052 adults based on the Seabird 2000 census (Mitchell *et al*., 2004).

Offshore wind farm studies of Arctic tern

Arctic and common terns can be difficult to distinguish during fieldwork and studies commonly pool sightings of both species and report results for common/Arctic tern.

Results from bird monitoring at operational wind farms indicate that common/Arctic terns are not likely to be displaced from offshore wind farms. At Horns Rev, Denmark, studies concluded that common/Arctic terns showed "no general avoidance reaction to offshore wind farms" (Diersche and Garthe, 2006). At Egmond aan Zee, the Netherlands (Leopold *et al*., 2011), the results of modelling the distribution of common/Arctic terns for three post construction surveys, showed there was no significant avoidance of or attraction to the wind farm. Summarising the survey results for this study, Lindeboom *et al*., (2011) states "…terns (unspecified) did not avoid the wind farm and used it for foraging". At Arklow Bank, Ireland post-construction monitoring found no evidence that common/Arctic terns were significantly displaced from the wind farm (Barton *et al*., 2009, Barton *et al*., 2010). In German marine areas wind farms were assessed to have little or no effect in displacing Arctic or common terns (Diersche and Garthe, 2006).

Most studies of common/Arctic tern flying behaviour suggest that wind turbines are unlikely to present a barrier, however evidence from Den Oever, the Netherlands illustrates the potential for terns to be deflected around a turbine. Here, visual and radar monitoring of flight paths of approximately 6,500 common terns passing a single turbine showed that the terns deviated to both sides but only by a distance of 50-100 m from the turbine (Diersche and Garthe, 2006). Of *ca.* 500 flying common terns recorded at Zeebrugge Harbour, Belgium, 94% of birds crossing the wind farm did not react to the turbines; the remaining 6% showed a reaction to the turbines but nearly all these birds flew through the wind farm once they had changed direction (Everaert, 2003). In a later five-year study of *ca.* 2,500 common terns breeding at this site the authors concluded that "during the breeding season the line of wind turbines at the eastern port breakwater didn't act as a barrier for the foraging flights of the terns" (Everaert and Stienen, 2007). Similarly only 5% of common terns monitored at Maaskvlakte, the Netherlands showed avoidance behaviour when approaching turbines and the authors concluded that the wind farm did not act as a barrier to the daily foraging movements of the terns (van den Bergh *et al*., 2002). At Yttre Stengrund, Sweden, radar combined with visual monitoring of *ca.* 1,000 Arctic and common tern flights indicated that during the autumn passage period, flights were not deflected in response to the wind farm (Pettersson, 2005). The author stated that these species "passed in small flocks of 6-27 birds without making any great deviation manoeuvre and they also flew between or alongside the turbines" (Pettersson, 2005). The review by Hötker *et al*., (2006) of studies of wind farms as barriers to birds, in identified three studies where wind farms were, and one where they were not concluded to act as barriers to common terns; no other details were given.

The number of Arctic terns colliding with wind turbines is likely to be low based on collision risk studies and reported flying heights from operational wind farms. At Zeebrugge Harbour, Belgium, a five-year study of collision risk to the adjacent common tern breeding colony found that the collision probability for common terns crossing the line of wind turbines was 0.110 - 0.118% for flights at rotor height and 0.007 - 0.030% for all flights (Everaert and Stienen, 2007). At Yttre Stengrund, Sweden, *ca* .900 flying Arctic and common terns observed during the 2002 autumn migration period typically flew at *ca.* 10 m above sea level (Pettersson, 2005). Similarly, low flight heights were recorded for common/Arctic terns monitored in the vicinity of Arklow Bank, Ireland where less than 2% of 565 birds monitored over two years were recorded flying at a height over 20 m above the sea surface (Barton *et al*., 2009, Barton *et al*., 2010). At Zeebrugge Harbour, 82% of common terns were recorded flying at below 16 m. At North Hoyle, Wales 47 of 79 (59%) of common terns were estimated to be flying below 20 m (PMSS, 2007). At Zeebrugge Harbour 89 common tern turbine collision fatalities were found during the five-year study. The corrected yearly estimate of common tern fatalities expressed as a proportion of the number of breeding individuals present in any one year ranged from 0.4% to 3.7% (Everaert and Stienen, 2007).

EIA Construction Phase assessment for Arctic tern

The construction phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

Arctic terns are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during construction operations on the regional autumn passage populations of Arctic tern is ***not significant*** under the EIA Regulations.

EIA Operational Phase assessment for Arctic tern

Potential for Arctic tern to be affected by displacement

For the purpose of assessing displacement it is assumed that 25% of Arctic terns will be displaced from the proposed wind farm footprint and a surrounding buffer of 1 km. This is likely to be a cautious assumption as existing evidence suggests that it is likely that the species will not be displaced during the operational phase.

Table 4.73 The mean estimated number and percentage of the regional population of Arctic terns potentially at risk of displacement from offshore site plus 1 km buffer during the autumn passage period. Numbers include unidentified common/Arctic terns; these comprised approximately 9% of the total

Receptor population (Source)	Population size (adults)	No. assumed at sea	Dev.Area		Dev.Area + 1km	
			Mean no. at risk	%	Mean no. at risk	%
National, breeding (Seabird 2000)	105,159	105,159	363	0.3%	425	0.4%
UK east coast, breed (Shetland to Blyth) (Seabird 2000)	98,052	98,052	363	0.4%	425	0.4%
Regional, breeding (Seabird 2000)	10,056	10,056	363	3.6%	425	4.2%

The value of the offshore site buffered to 1 km as a foraging site for Arctic terns in the autumn passage period was estimated from the mean proportion of regional breeding population present in autumn passage period and from the mean proportion of eastern Scotland and north-east England breeding population present (Table 4.73).

On this basis, if the origin of the birds in the autumn is limited to breeding sites within the defined breeding region (Peterhead to Blyth) it is inferred that the offshore site buffered to 1 km might provide up to approximately 4.2% of the autumn passage period foraging resources (Table 4.73). However, for the reasons discussed earlier, it is considered more likely that birds in the autumn passage period originate from colonies across eastern Scotland and north-east England (and possibly further afield) in which case it would then be inferred that the offshore site buffered to 1 km might provide up to approximately 0.4% of the autumn passage period foraging resources (Table 4.73).

Likely impacts of displacement on Arctic tern population

Were 25% of Arctic terns to be displaced from the offshore site buffered to 1 km, this would represent a loss of up to approximately 1.1% of the regional autumn passage period foraging resources (25% of the regional breeding population value in Table 4.73). This is likely to be an overestimate of the importance of the area for foraging because as discussed earlier, the size of the autumn passage population is likely to be larger than the regional breeding population. If the autumn passage population comprised birds from breeding grounds across eastern Scotland and north-east England, which is considered more likely, the equivalent loss of foraging resources figure reduces to 0.1% (25% of UK east coast breeding population value in Table 4.73).

The loss of 0.1% of the atumn passage population's foraging resources would be an effect of negligible magnitude (<1%) and temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effect of displacement on the regional Arctic tern population in the autumn passage period is ***not significant*** under the EIA Regulations.

The numbers of Arctic tern present in the spring passage period in the offshore site buffered to 1 km were much lower than in the autumn passage period. Therefore, it follows that any loss of foraging habitat caused by displacement in this period is also ***not significant*** under the EIA Regulations.

Barrier effect

Based on evidence from other wind farm studies summarised above, and the absence of Arctic terns in the offshore site in June and July on baseline surveys, it is concluded that the offshore site will not present a barrier to foraging Arctic terns during the breeding season.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

Arctic terns are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional autumn passage populations of Arctic terns is ***not significant*** under the EIA Regulations.

Collision mortality

Collision risk modelling was undertaken for Arctic tern based on an assumed population of 1,000 birds passing through the offshore site twice per year, once in spring and autumn, with an estimated 3% of birds at flight height (Cook *et al.*, 2012). This resulted in a total of 0.1 collisions predicted per year, based on an avoidance rate of 98.0%, or 0.03 collisions predicted per year, based on an avoidance rate of 99.5%. Further details are presented in Appendix 12.2.

Based on a baseline annual adult mortality rate of 10.0% (BTO, 2012), the additional collision mortality would lead to an increase of 0.1% in the annual adult mortality rate (i.e., to 10.1% p.a.) for collisions predicted using a 98.0% avoidance rate, and to an increase of 0.03% for collisions predicted for a 99.5% avoidance rate. In both case the effect is of negligible magnitude.

Scaling this up to the size of the Arctic tern population of 98,052 adults that might pass through would give a worst case scenario of 98 and 30 collisions per annum for 98.0% and 99.5% avoidance rates respectively. Nevertheless the effect of this on the population's adult mortality rate remains the same as shown above.

This worst case scenario assumes that all birds in the eastern Scotland and north-east England breeding population pass through the wind farm at rotor height twice a year, which is extremely unrealistic for two reasons. First, in actuality a smaller proportion of Arctic tern flight activity would be at rotor height, indeed, 0.4% of 964 flying Arctic terns recorded in baseline surveys were above the proposed rotor height, compared to 3% assumed here. Second, Arctic terns migrate on a relatively broad front that is wider than the offshore site. Therefore, only a relatively small proportion (say, <25%) of the population would be expected to pass through the offshore site. It is likely therefore that the actual effects of collision mortality on migrating Arctic terns is considerably lower than the worst case scenario figures presented above.

The potential effect of the collision mortality of Arctic terns on the baseline mortality rate is categorised as negligible in magnitude (<1%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of collision mortality on Arctic terns is ***not significant*** under the terms of the EIA Regulations.

EIA Decommissioning Phase assessment for Arctic tern

The decommissioning phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

Arctic terns are not considered susceptible to disturbance impacts and were assessed as being at low risk of displacement resulting from offshore wind farms (Langston 2010). Any such impact is therefore categorised as negligible (<1%) magnitude, and temporally short-term and reversible (Table 3.1 and Table 3.2). Given the expected low sensitivity of the population to displacement it is concluded that the impacts of displacement during decommissioning operations on the regional autumn passage populations of Arctic terns is ***not significant*** under the EIA Regulations.

EIA summary of effects combined for Arctic tern

The adverse impacts of the effects assessed will act in a broadly additive manner. In combination it is judged that the magnitude of the three effects on the population is negligible. It is concluded that the overall impact on the regional population of Arctic tern in the autumn passage period is ***not significant*** under the EIA regulations (Table 4.74).

Table 4.74 Summary of effects on the regional population of Arctic terns in the autumn passage period

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	Not significant
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Low	Not significant
Vessel disturbance	Negligible	Long term	Low	Not significant
Collision mortality	Negligible	Long term	Low	Not significant
All effects combined	Negligible	Long term	Low	Not significant

4.4.26.5 Cumulative Impact Assessment for Arctic tern

Combining the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts on the regional population of Arctic terns in the autumn passage period will be as shown in Table 4.75. The individual impacts for displacement and collision are combined by addition to give the cumulative impact.

Predicted displacement causing the potential effective loss of up to 0.8% of the foraging habitat of the autumn passage population of Arctic tern is rated as negligible magnitude (<1%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional population of Arctic terns in the autumn passage period is ***not significant*** under the terms of the EIA Regulations.

Table 4.75 Summary of CIA for the three proposed offshore wind farms in south-east Scotland on the regional autumn passage population of Arctic tern

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF	Predicted impact from Inch Cape pOWF	Predicted impact from Outer Forth pOWF	Cumulative Impact
Displacement: effective loss of foraging habitat during the autumn passage period (%)	25%	0.1%, negligible	<0.1%, negligible	0.6%, negligible	0.8%, Negligible
Collision: % increase in assumed annual adult mortality rate. All designs.	0%	No change, Negligible	No change, Negligible	No change, Negligible	No change, Negligible

The potential cumulative impact of collision mortality on the baseline mortality rate of the regional Arctic tern population in the autumn passage period is rated as negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional population of Arctic tern in the autumn passage period is ***not significant*** under the terms of the EIA Regulations.

4.4.26.6 Mitigation measures for Arctic tern

The assessment does not identify any significant adverse effects on the regional population of Arctic tern in the autumn passage period. Therefore no mitigation measures are required for this species.

4.4.27 Common Guillemot *Uria aalge*

4.4.27.1 Status

Guillemots are one of the commonest seabird species in Britain, breeding in large colonies on suitable coastal cliff habitat. There are several large colonies on the east coast, and Seabird 2000 recorded 1,322,830 individuals at breeding colonies in Britain (Mitchell *et al.,* 2004). The closest large colonies to the Neart na Gaoithe study area are the Isle of May, St Abb's Head and Fowlsheugh. Guillemots mostly prey on small fish species such as lesser sandeels, sprat and gadoid fish (Mitchell *et al.,* 2004).

4.4.27.2 Neart na Gaoithe Study Area

Guillemot was the third most frequently recorded seabird recorded on surveys in Year 1, with a total of 7,898 birds. Overall, the majority of guillemots (84.1%) were recorded in the buffer area (Table 4.2). Peak numbers were recorded in the offshore site (705 birds) and the buffer area (1,833 birds) in October. A further 3,323 unidentified guillemot/razorbills, and 1,348 unidentified auks were also seen on surveys in Year 1.

In Year 2, guillemot was the second most frequently recorded species on surveys in the Neart na Gaoithe study area (11,730 birds). The majority of birds (86.8 %) were again recorded in the buffer area (Table 4.2). Numbers of guillemots in the offshore site were slightly higher than in Year 1, with a peak of 427 birds in April. Numbers recorded in the buffer area peaked in September (2,242 birds). A further 1,532 unidentified guillemots/razorbills and 827 unidentified auks were also seen on the Year 2 surveys.

During the breeding period (April to June), the peak estimated number of guillemots in Year 1 occurred in June, with 354 birds in the offshore site and 4,474 birds in the buffer area (Table 4.76) (Figure 4.79). In Year 2, estimated numbers of guillemots in the offshore site peaked in April (3,272 birds), with 6,650 birds in the buffer area in May.

During the chick period (July and August), the peak estimated number of guillemots in the offshore site in Year 1 occurred in July (130 birds), with 1,551 birds in the buffer area in August (Table 4.76) (Figure 4.79). In Year 2, estimated numbers of guillemots peaked in July, with 1,186 birds in the offshore site, and 6,724 birds in the buffer area.

In the post-breeding season (September and October), peak estimated numbers of guillemots occurred in October of Year 1, with 8,315 birds in the offshore site, and 20,136 birds in the buffer area (Table 4.76) (Figure 4.79). In Year 2, peak estimated numbers of guillemots occurred in September, with an estimated 2,439 birds in the offshore site, and 21,760 birds in the buffer area.

In the non-breeding season (November to March), peak estimated numbers of guillemots in the offshore site occurred in November of Year 1 (1,039 birds), with 3,785 birds in the buffer area in March (Table 4.76) (Figure 4.79). In Year 2, peak estimated numbers of guillemots in the offshore site occurred in December and March, with 726 and 725 birds respectively. Estimated numbers in the buffer area were highest in March (8,410 birds) of Year 2.

Estimated numbers in the buffer area exceeded 1% of the national breeding population (13,228 birds) (Mitchell *et al.,* 2004) in October of Year 1 and September and October of Year 2. Estimated numbers in October of Year 1 and September of Year 2 exceeded the nominal internationally important threshold of 20,000 birds (Holt *et al.,* 2011) (Figure 4.79).

Table 4.76 Estimated numbers of guillemots in the offshore site and 8 km buffer in Years 1 and 2

Month	Estimated nos on water Offshore Site	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying Offshore Site	Estimated total Offshore Site	Estimated total 8 *km* buffer [1]	Estimated total
Yr1 Nov	1,025	762	1,378	14	1,039	3,416	4,455
Yr1 Dec	258	160	416	7	265	1,435	1,700
Yr1 Jan	26	13	52	0	26	1,072	1,098
Yr1 Feb	65	39	107	34	99	1,475	1,574
Yr1 Mar	340	201	574	13	353	3,785	4,138
Yr1 Apr	162	89	296	143	305	3,874	4,179
Yr1 May	12	6	24	109	121	3,384	3,505
Yr1 Jun	347	241	499	7	354	4,474	4,828
Yr1 Jul	130	91	185	0	130	757	887
Yr1 Aug	28	15	53	0	28	1,551	1,579
Yr1 Sep	1,604	1,139	2,259	0	1,604	11,489	13,093
Yr1 Oct	8,281	6,998	9,799	34	8,315	20,136	28,451
Yr2 Nov	-	-	-	-	-	-	-
Yr2 Dec	705	486	1,021	21	726	3,420	4,146
Yr2 Jan	128	83	198	48	176	3,573	3,749
Yr2 Feb	482	261	889	20	502	2,867	3,369
Yr2 Mar	692	444	1,077	33	725	8,410	9,135
Yr2 Apr	3,013	1,899	4,778	259	3,272	3,773	7,045
Yr2 May	241	154	377	94	335	6,650	6,985
Yr2 Jun	33	20	54	74	107	4,195	4,302
Yr2 Jul	1,152	794	1,672	34	1,186	6,724	7,910
Yr2 Aug	332	225	490	20	352	5,570	5,922
Yr2 Sep	2,439	1,627	3,657	0	2,439	21,760	24,199
Yr2 Oct	1,209	856	1,709	7	1,216	13,776	14,992

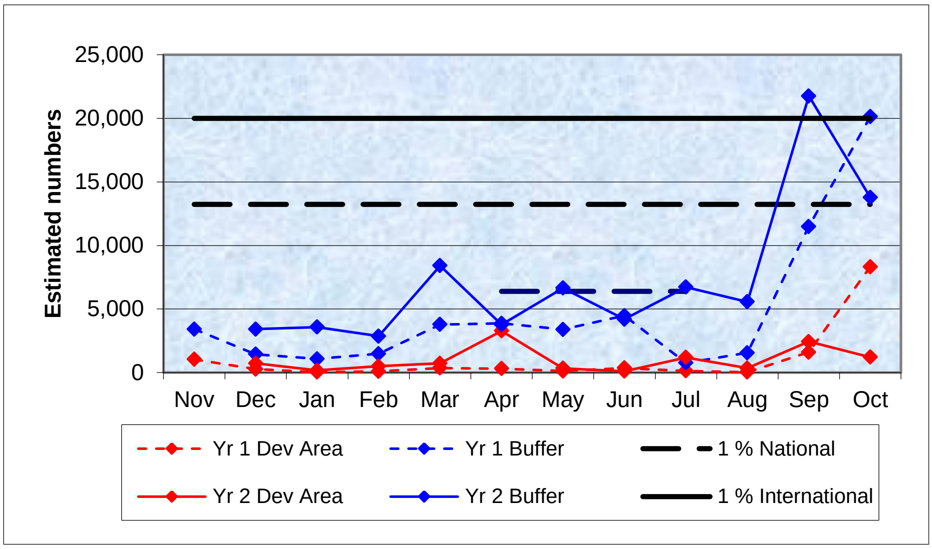


Figure 4.79 Monthly estimated numbers of guillemots in the Neart na Gaoithe Development & buffer areas in Years 1 and 2

Mean monthly guillemot density was generally similar between the offshore site and the buffer area, in Years 1 and 2 (Figure 4.80). In the offshore site, mean density peaked in October of Year 1 (77.6 birds/km^2), and in April of Year 2 (34.0 birds/km^2), with a slightly lower peak in September (31.8 birds/km^2). In the buffer area, mean density peaked in October of Year 1 (47.1 birds/km2), and September of Year 2 (47.9 birds/km^2). ESAS mean density data peaked in August, with 26.4 birds/km^2 in the surrounding ICES rectangles, and 13.6 birds/km^2 in Regional Sea 1 (Figure 4.80).

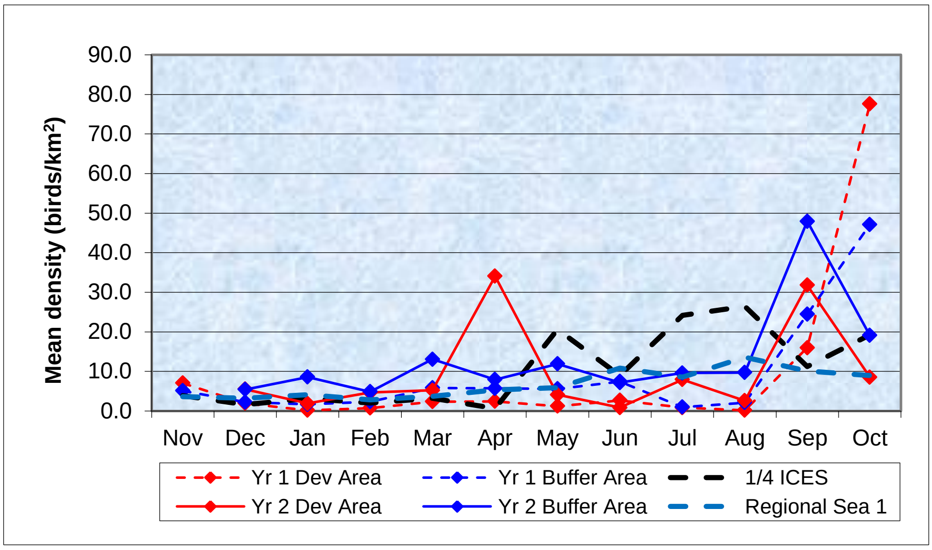


Figure 4.80 Comparison of monthly mean densities for guillemot in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

Between November and March of Year 1, guillemots were widespread throughout the study area at mostly low to moderate densities, with high densities in the offshore site and in the east and west of the buffer area (Figure 4.81).

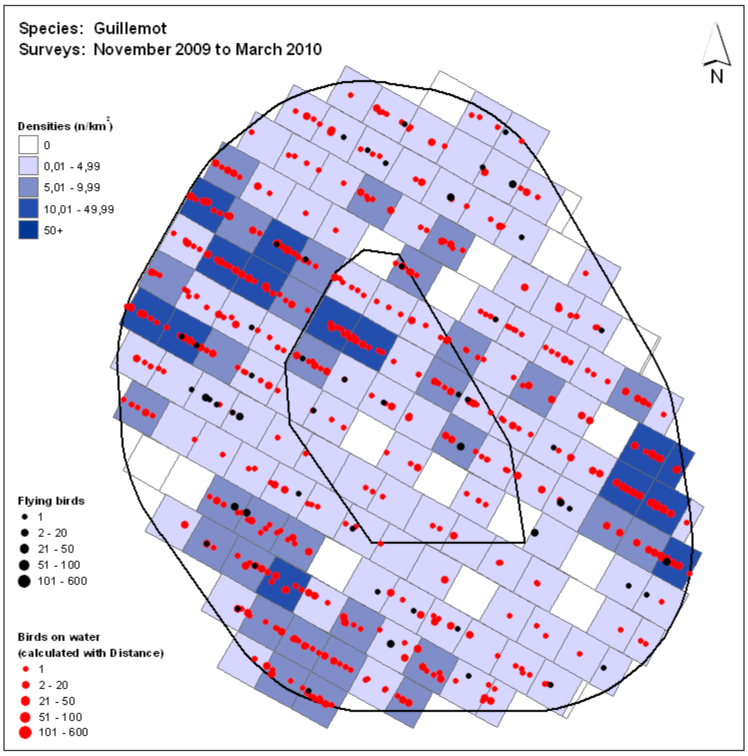


Figure 4.81 Guillemot density between November and March, Year 1

Guillemot distribution over the same period in Year 2 was broadly similar, with moderate to high densities of guillemots recorded in the offshore site, and in the north-east and south of the buffer area with low densities elsewhere (Figure 4.82).

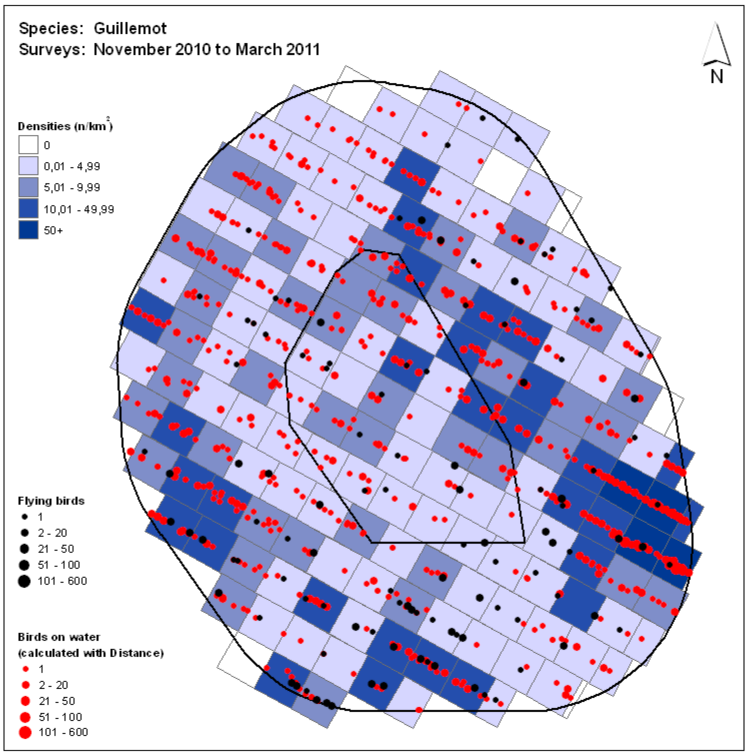


Figure 4.82 Guillemot density between November and March, Year 2

Between April and June of Year 1, guillemots were less widespread across the study area. Densities in the offshore site were mostly low, with higher densities in the south (Figure 4.83). In the buffer area, highest densities were recorded in the south-east at this time.

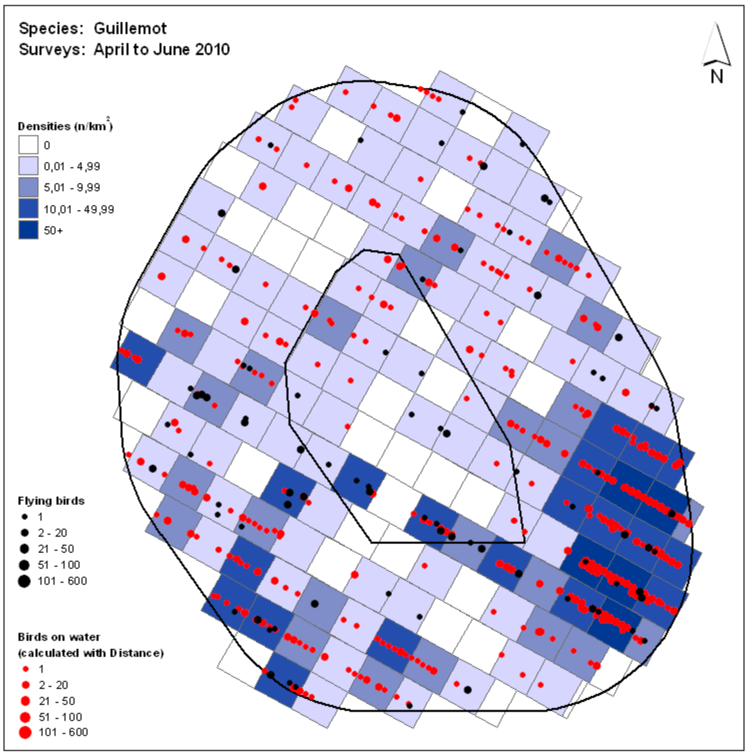


Figure 4.83 Guillemot density between April and June, Year 1

A similar distribution pattern was recorded between April and June in Year 2, with highest densities of guillemots in the south-east of the buffer area (Figure 4.84). Densities in the offshore site were mostly low to moderate at this time.

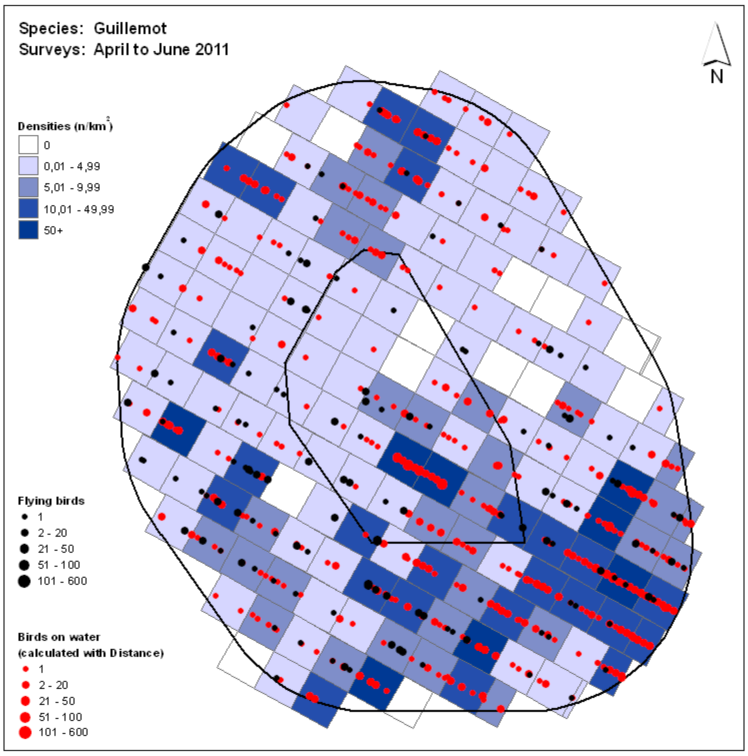


Figure 4.84 Guillemot density between April and June, Year 2

Guillemot were least widespread with lowest density in July and August of Year 1 (Figure 4.85). Birds were largely absent from the offshore site at this time, apart from low densities in the north. In contrast to previous months, the south-east of the buffer area held very few birds, with greatest concentrations recorded in the south-west of the buffer area.

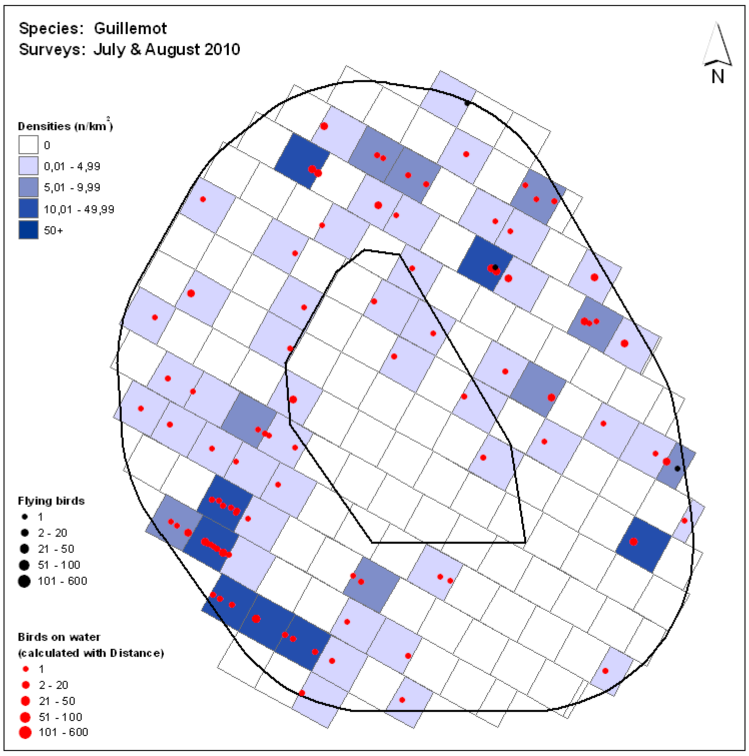


Figure 4.85 Guillemot density in July and August, Year 1

Guillemots were more widespread at high densities in July and August of Year 2 compared to Year 1 (Figure 4.86). Densities in the offshore site at this time were mainly moderate to high. Highest densities of guillemots in the buffer area were recorded in the north and south, with lower densities in the south-east and north-west of the buffer area.

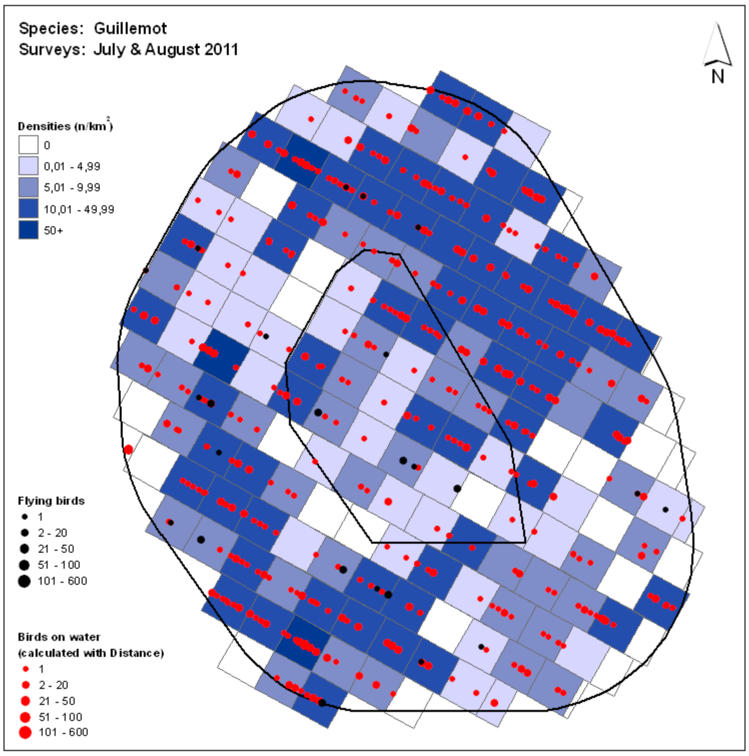


Figure 4.86 Guillemot density in July and August, Year 2

In Year 1, peak densities of guillemots were recorded in September and October (Figure 4.87). Birds were widespread at mostly high densities across the offshore site and most of the buffer area at this time, apart from in the south-east of the buffer area, where densities were low or zero.

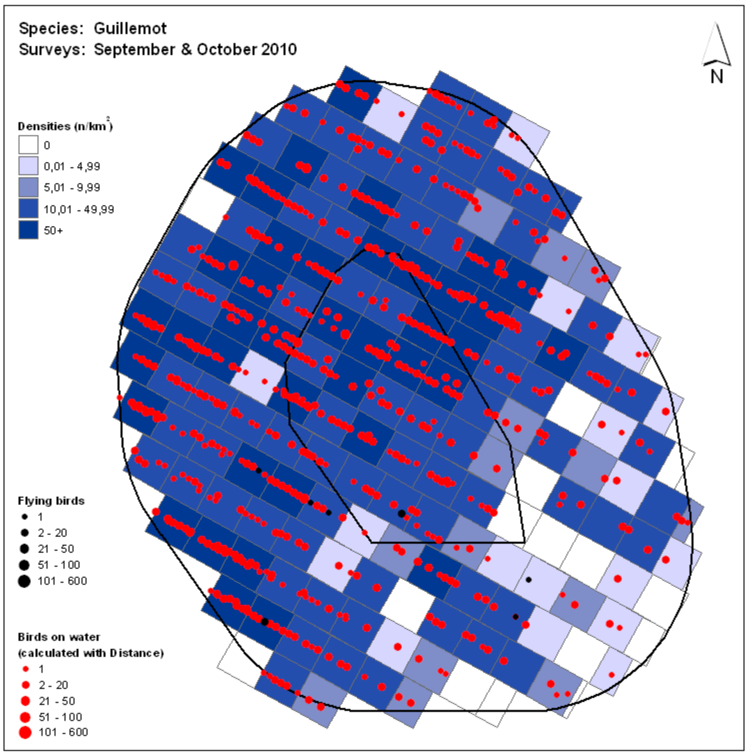


Figure 4.87 Guillemot density in September and October, Year 1

A similar pattern was recorded in September and October of Year 2, with high densities of guillemots in the west and south of the offshore site and buffer areas (Figure 4.88). In the north-east of the offshore site and east of the buffer area, densities were lower or zero at this time.

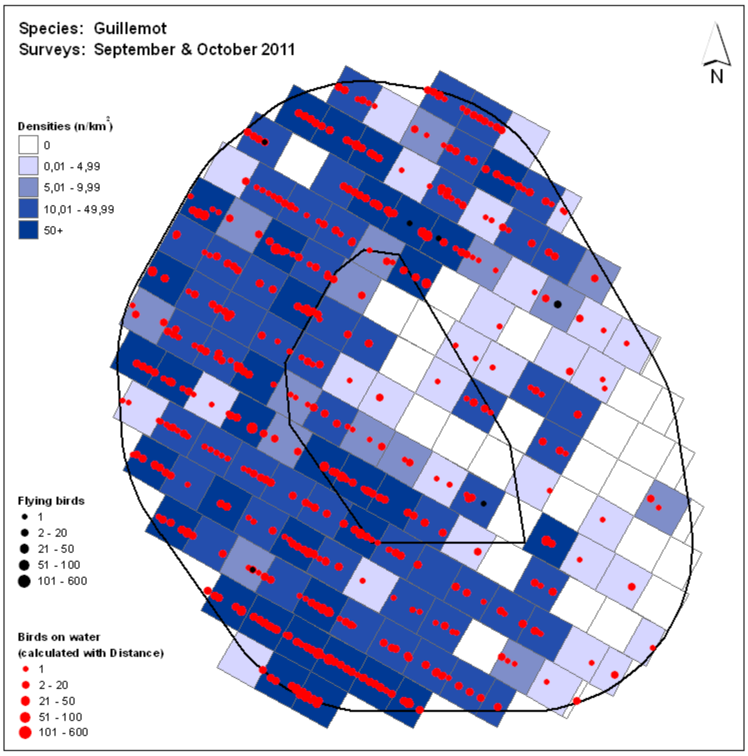


Figure 4.88 Guillemot density in September and October, Year 2

Foraging behaviour was recorded for 197 guillemots in the Neart na Gaoithe study area in Years 1 and 2, with five types of foraging behaviour recorded (Table 4.77). The majority of all foraging birds were recorded holding fish (51.8%) and pursuit diving (38.6%). Prey was identified for 21 prey items, with 20 sandeels and one herring/sprat recorded.

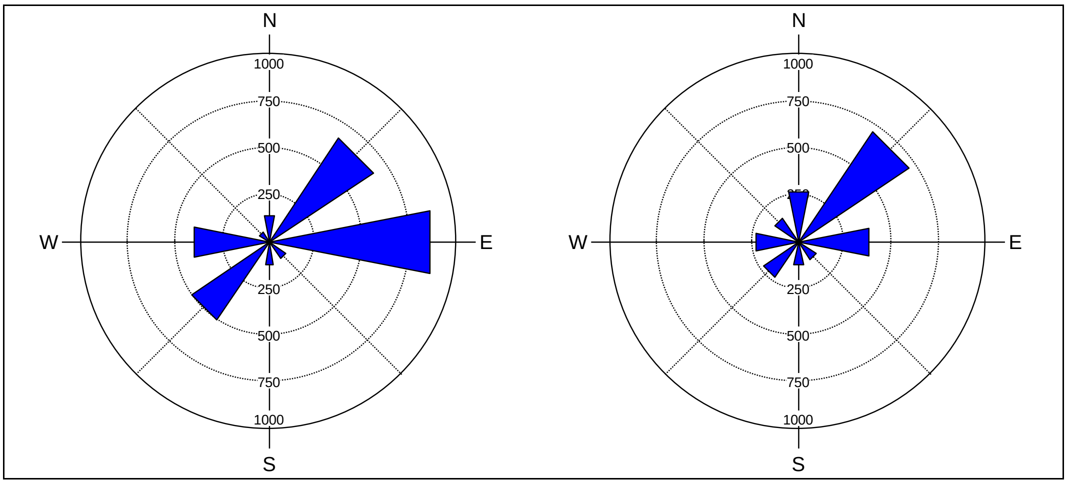
Table 4.77 Guillemot foraging behaviour in the Neart na Gaoithe Study Area in Years 1 and 2

Behaviour	Number of birds
Actively searching	7
Pursuit diving	76
Holding fish	102
Surface pecking	12
Total	**197**

In Years 1 and 2, 5,061 guillemots were recorded in flight, with almost all birds recorded flying below 22.5 m in height (Table 4.3). The majority of birds (98.4%) were recorded flying below 7.5 m in height. Two birds (0.04%) were recorded flying above 22.5 m i.e. within the rotor-swept zone, at estimated heights of 25 m to 30 m.

Flight direction was recorded for 2,875 guillemots in the breeding season (April to June), with direction recorded for 2,193 guillemots in the post-breeding and non-breeding seasons combined (July to March) (Figure 4.89).

In the breeding season, just under a third of all birds recorded were flying east (30.2%), with 23.0% of birds flying north-east. In the post-breeding and non-breeding seasons, approximately one third of birds were recorded flying north east (32.1%), with 17.2% flying east.



April to June (n=2,875 birds) July to March (n=2,193 birds)
Numbers shown on figures are number of birds recorded

Figure 4.89 Flight direction of guillemots in the Neart na Gaoithe Study Area in Years 1 and 2

Recent tracking studies on guillemots breeding on the Isle of May, Fowlsheugh and St Abb's Head undertaken by CEH at the request of FTOWDG indicate that guillemots from the Isle of May use both coastal and offshore areas, with a mean maximum range of 18 km and a maximum of 61 km (Daunt *et al.*, 2011a). Guillemots breeding at Fowlsheugh had a mean maximum range of 12 km, while guillemots at St Abb's Head had a mean maximum range of 16 km. The maximum range for the latter two colonies for guillemot was 55 km (Daunt *et al.*, 2011b).

4.4.27.3 Species sensitivity

Guillemot is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009).

A recent review assessed guillemot as being at moderate risk of displacement, barrier effects, habitat loss or changes in prey distribution resulting from offshore wind farms, while collision risk was rated as low risk. Overall, guillemot was assessed as being at moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

Guillemot is listed as a qualifying interest species in the breeding season for five SPAs on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development (Table 4.78). These SPAs held 65.2% of the UK breeding population and 20.4% of the biogeographic population at the time of designation (JNCC, 2012). The distance between the offshore site and four SPAs (Farne Islands SPA, Forth Islands SPA, Fowlsheugh SPA and St Abb's Head to Fast Castle SPA) is within the mean maximum foraging range of guillemot (84.2 km). The distance to the remaining SPA (Buchan Ness to Collieston Coast SPA) is within the maximum known foraging range (135 km) (Thaxter *et al.*, 2012). Guillemot mean maximum foraging range from breeding SPAs in relation to the offshore site are shown in Figure 4.90.

Table 4.78 SPAs for breeding guillemot between Peterhead and Blyth

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count (birds at colony) [2]	Year
Buchan Ness to Collieston Coast	*113*	*8,640*	*0.4*	*1.2*	*20,858*	*2007*
Farne Islands	**72**	**23,499**	**1.0**	**3.3**	**48,126**	**2009**
Forth Islands	**16**	**22,452**	**1.0**	**3.2**	**23,967**	**2009**
Fowlsheugh	**62**	**40,140**	**1.8**	**5.7**	**50,556**	**2009**
St Abb's Head to Fast Castle	**31**	**20,971**	**0.9**	**3.0**	**33,181**	**2008**
Total	**-**	**115,702**	**5.1**	**16.4**	**176,688**	**-**

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. Sites in italics lie within the maximum known foraging range of 135 km. Sites in bold lie within the mean maximum foraging range of 84.2 km (Thaxter *et al.*, 2012).

A recent JNCC statistical analysis of ESAS data investigating possible marine SPAs, identified waters to the east of the Neart na Gaoithe study area as an important area for guillemots during the breeding season (Kober *et al.,* 2010). The waters around the Neart na Gaoithe study area were also identified as an important area for guillemots during the non-breeding season (October to April) (Kober *et al.,* 2010).

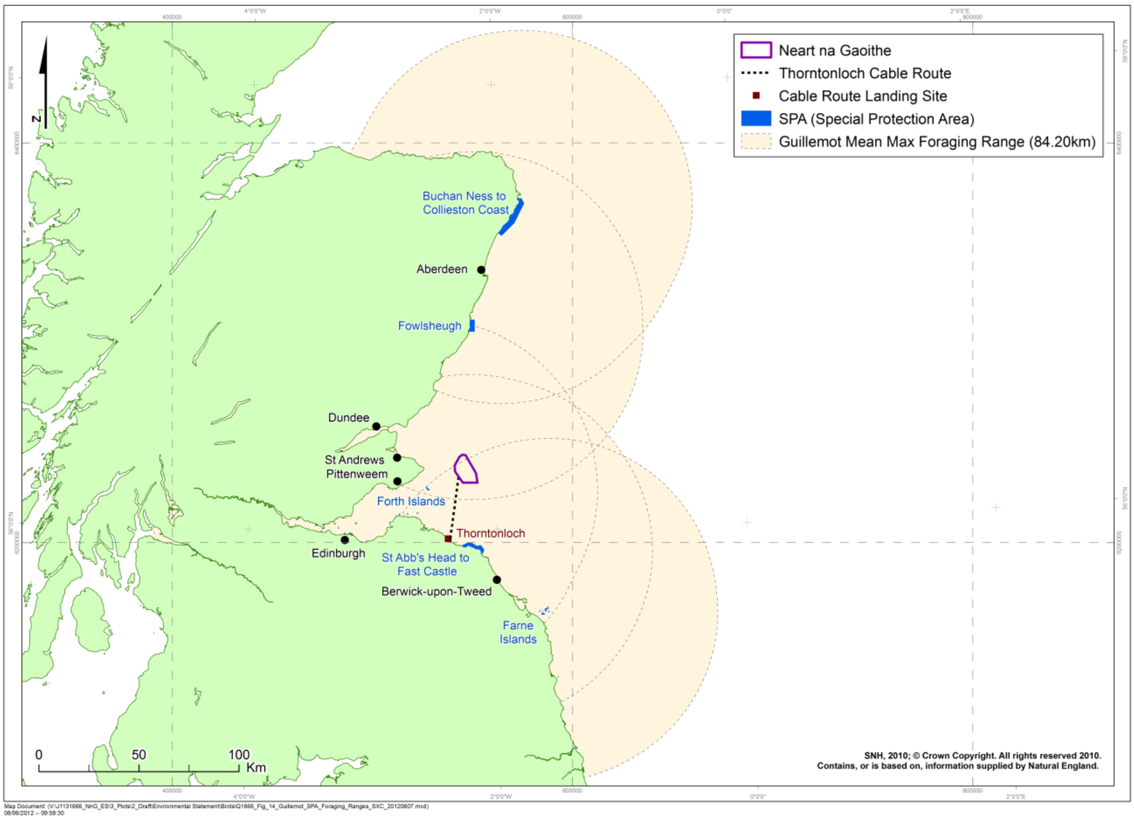


Figure 4.90 Guillemot mean maximum foraging range from breeding SPAs in relation to the Development

4.4.27.4 Assessment

Treatment of unidentified guillemots

Guillemots and razorbill are similar in their appearance and some individuals could not be identified to species level during surveys, for example birds seen in poor light in the outer parts of the survey strip. In all survey visits the vast majority of individuals of these two species were positively identified. For the purposes of assessment the unidentified birds in a survey visit were included in population estimates by apportioning them in accordance to the ratio of positively identified individuals on that survey visit. This was considered to be the best way of dealing with this issue as it makes best use of the available data without introducing obvious biases. It is also the method recommended by Maclean *et al*., (2009) for dealing with the issue of unidentified individuals.

Breeding population size correction

The published estimates of breeding population size for guillemot and razorbill are based on the number of individuals present at breeding colonies during census counts. These are inevitably an underestimate of the number of breeding adults in the population as a substantial proportion of adults will be absent from the colony, away foraging, at the time of the census count. For the purposes of assessment the likely total number of adults in the breeding population was calculated by dividing the colony count estimate by the adult colony attendance rate. For example, if the mean colony attendance rate is 60%, a census count of 10,000 adults would translate to an estimated breeding population size of 16,666 adults. This method of calculating breeding population size is necessarily approximate, however, it is important to apply a correction of some form because basing assessments on 'raw' colony count data would introduce a large bias into the assessment process. Mean colony attendance rates were taken from published studies based on detailed information from tagged individuals. The mean colony attendance rate used for guillemot was 60% (Cairns *et al.,* 1987).

Definition of seasons

The annual cycle for guillemots was divided into four parts to reflect the biology of the species and the broad pattern of use of the offshore site.

The main breeding season, the period when breeding adults are attending colonies, is defined as April to June. At this time the vast majority of birds present in the offshore site will be from relatively local breeding colonies (those within foraging range).

The chick period is defined as July and August and corresponds to the time when successful male adults have accompanying dependent young with them on the sea. This period is clearly part of the breeding season, however adults are no longer constrained by having to visit the breeding colony. Indeed it is known that adults and their dependent young can quickly move away from colonies to feeding areas some distance away (Camphuysen, 2002). Nevertheless, it is likely that almost all individuals present in the offshore site during the chick period are from breeding sites within the region, particularly the relatively close large colonies. Some adult guillemot undergo wing moult in the chick period (Harris & Wanless 1988).

The post-breeding period was defined as September and October. Although birds at this time of year are no longer breeding they were examined separately to the later part of the non-breeding period (the 'winter') because the numbers present were much greater and it is likely that the vast majority of individuals present at this time are still from breeding sites within the region, particularly the relatively close large colonies. Also, a proportion of adult guillemot undergo wing moult in this period (Birkhead & Taylor 1977) which increase their sensitivity to disturbance and displacement effects.

The main non-breeding period is defined as November to March and broadly corresponds to the period when guillemots are in their over-wintering area. In this period it is likely that a high proportion of individuals present in the offshore site are from breeding colonies outwith the region, including birds from other countries (Wernham *et al.,* 2002).

Baseline conditions

In the colony attendance part of the breeding season (April to June) the mean estimated number of guillemots present in the offshore site was 737 birds, and the mean estimated number in the offshore site buffered to 1 km was 930 birds. On average 88.5% of birds present were on the sea and the remainder were in flight.

For the purposes of assessment it is assumed that all birds in summer plumage were breeding birds. Studies have estimated that mean colony attendance rate of guillemot during this part of the breeding season is 60% (Cairns *et al.,* 1987) and this figure is used in estimates of the importance of the offshore site for foraging.

In the chick-on-sea part of the breeding season period (July and August) the mean estimated number of adult guillemots present in the offshore site was 311 birds, and the mean estimated number in the offshore site buffered to 1 km was 499 birds. In this period 98% of the 1730 birds recorded during baseline surveys were sitting on the sea, and 11.6% were juveniles.

In the post-breeding period (September and October) the mean estimated number of adult guillemots present in the offshore site was 3,588 birds, and the mean estimated number in the offshore site buffered to 1 km was 5,111. In this period 99.7% of birds present were on the sea.

In the non-breeding period (November to March) the mean estimated number of guillemots present in the offshore site was 665 birds, and the mean number in the offshore site buffered to 1 km was 879 birds. In this period 92% of birds present were on the sea.

Populations

Guillemot is the commonest seabird species breeding in the region. The breeding population of guillemots in Scotland has undergone a prolonged period of decline and recent colony counts indicate that the decline is on-going (SMP, 2012). Assuming that the sub-set of colonies that have been recently counted is representative, the regional decline since the Seabird 2000 counts amounts to -13%. The decline in breeding numbers means that published figures on population size for this species (e.g., Mitchell *et al*., 2004) no longer accurately reflect the current population size in eastern Scotland. To prevent this causing assessment of effects to be biased low, the regional total derived from Seabird 2000 results has been adjusted downwards by 13%. On this basis, the regional breeding population is assumed to be 218,510 birds in terms of the sum of adjusted colony counts which translates to a total of approximately 312,151 breeding adults after taking colony attendance rate into account.

The size of the post-breeding-period regional guillemot population is assumed to be 274,050 adults. This is derived by summing the September to October period estimates for localities 7, 8, 9, 10, 11 and 12 given in Skov *et al*. (1995) and adjusting this figure downwards by 13% to reflect recent declines in breeding numbers (making this correction leads to more cautious assessment conclusions). This figure is remarkably similar to the estimated size of the regional breeding population despite it being derived from an independent data. The similarity of the two figures perhaps suggests that the majority of the regional breeding population remain in the area during this period.

The size of the winter-period regional guillemot population is assumed to be 521,000 birds. This is derived by summing the November to February period estimates for localities 4, 6, and 7 given in Skov *et al*. (1995).

An unknown proportion of the regional breeding population may remain in the region through the non-breeding period but these are joined by many more birds from more distant colonies.

Nature conservation importance

For EIA assessment purposes, the nature conservation importance of guillemot using the offshore site is categorised as high during the breeding season. The species is classed as high NCI because a high proportion of birds using the offshore site are likely to be from the breeding colonies within the Forth Islands SPA and St Abb's Head to Fast Castle SPA, where this species is a qualifying interest. Mean numbers present in the offshore site buffered to 1 km exceeded 1% of the (at-sea) regional population in the post-breeding period. This species is not subject to any special legislative protection, nor is it on any conservation priority lists.

Offshore wind farm studies of guillemot

The extent to which guillemots are displaced from operational wind farms differs between studies. Monitoring studies at offshore wind farms in Denmark and the Netherlands indicate conflicting evidence on the extent that guillemot are displaced, however, low statistical power as a consequence of low bird densities, clumped distributions or between year variation in bird numbers may explain some of the apparent differences in these results.

Studies at Horns Rev, Denmark report that although guillemots were recorded in relatively low numbers in the wind farm and buffer compared to the wider monitoring area during the pre-construction surveys, no guillemots occurred within 4 km of the wind farm during the construction period representing a significant decrease. In the operational period the selectivity index for the wind farm plus a 4 km buffer was significantly lower, compared to the equivalent figure for the pre-construction period suggesting a reduced use of the sea area occupied by, and surrounding the wind farm during the operational period (Diersche and Garthe, 2006). However, these findings were not corroborated by a significant result when a subset of the Horns Rev guillemot data was analysed (Petersen *et al*., 2006) and therefore some caution is implied when interpreting the response of guillemot to the Horns Rev wind farm. Furthermore, the authors stress that displaced birds should not only be attributed to the physical presence of the turbines, but possibly also to service boat traffic, which occurred on *ca.* 150 days of the year.

Compared to Horns Rev, the modelled results from the Egmond aan Zee and the adjacent Princess Amalia wind farm, the Netherlands did not conclusively show that guillemots were displaced from either of these wind farms (Leopold *et al*., 2011). Where guillemots were significantly displaced (2 out of 9 survey visits) this was not total, with birds recorded within both wind farms. However, the authors suggest that higher turbine density probably increased displacement of guillemots. The authors of this study conclude that the magnitude of the displacement effect for guillemots was less than 50% (Leopold *et al*., 2011).

In other post-construction monitoring studies reviewed, there was no clear evidence showing that guillemots were displaced from the wind farm and the surrounding sea. At North Hoyle, Wales, a highly significant increase in guillemot numbers (estimated at 55%) was reported since the wind farm became operational. However, this finding appears to result from comparing monitoring results from the operational period with those from the construction period (RWE Group). Despite this, the results corroborate the findings from the studies at Egmond aan Zee that guillemots are frequently present within wind farms (Leopold *et al.,* 2011). Results from North Hoyle are of particular interest to the proposed development because they are likely to include actively breeding guillemots, unlike the Danish and Netherlands studies where guillemots were only recorded outside the breeding season. Post-construction monitoring undertaken at other offshore wind farms have not recorded any displacement of guillemots from constructed wind farms, e.g. at Arklow Bank where there was no statistical difference in the number of guillemots recorded between pre and post construction (Barton *et al*., 2009). In summary, it is likely that guillemots will be partly displaced from the wind farm footprint and the proportion of birds displaced may be sensitive to spacing distance between turbines.

There is limited evidence of guillemot flights deflecting around or away from wind farms. Visual monitoring during boat surveys at Egmond aan Zee reported that guillemots showed a "strong avoidance behaviour in their flight pattern" in the vicinity of the farm, deflecting typically at between 2 km and 4 km from the wind farm perimeter (Lindeboom *et al*., 2011). At Horns Rev, Denmark, visual monitoring from an observation platform positioned at the edge of the wind farm found that 3.8% (sample size not given) of flying guillemots/razorbills were either within or flying into the wind farm (Diersche and Garthe, 2006). Summarising the barrier effect of wind farms on seabirds in German marine areas, guillemots were categorised as having a strong deflection/avoidance response (Diersche and Garthe, 2006).

The risk of guillemots colliding with wind turbine rotors is likely to be very low based on reported flying heights at operational wind farms. Of approximately 1,000 flying guillemots recorded during two years of monitoring in the vicinity of Arklow Bank, Ireland, no birds were recorded flying at a height over 20 m above the sea surface (Barton *et al*., 2009, Barton *et al*., 2010). At North Hoyle, Wales, only 4% (3 of 85) birds flying in the vicinity of the wind farm were above 20 m. The review of offshore wind farm effects on birds (Diersche and Garthe, 2006) acknowledges the low flying height of guillemots. Although the evidence from these operational wind farms strongly suggests a very low risk of guillemots colliding with turbines, a single fatality reported in a review of the number of collision victims at wind farms in eight European countries demonstrates that collisions do occur (Hötker *et al*., 2006). It is not known if this fatality occurred as a result of collision with a rotor or a turbine tower.

EIA Construction Phase assessment for guillemot

The construction phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

The extent that birds may be displaced by vessels varies depending on the type and speed of the vessel and possibly the time of year. Guillemots are not thought to be particularly sensitive to vessel movements but birds disturbed by vessels will either swim or fly away to an alternative location until the vessels have passed, after which they may return to the area. Consequently, there may be a localised, short-term effect.

During the construction phase there is the potential for the prey species, e.g. sandeels, of guillemot to be displaced, particularly during piling activities. Should this occur then it is predicted that the guillemots will also relocate as they follow the movements of their prey.

Noise modelling undertaken indicates that behavioural impacts on sandeels from piling noise is predicted to extend less than 1.5 km from the piling activities (See Chapter 15). Therefore, the effect on guillemots foraging on sandeels is likely to be relatively localised.

Based on this, possible displacement impacts were categorised as being of low (1-5%) magnitude, and being temporally short-term and reversible (Table 3.1 and Table 3.2). Overall, it is concluded that the impacts of displacement during construction operations on the regional population of guillemots throughout the year is ***not significant*** under the EIA Regulations.

EIA Operational Phase assessment for guillemot

Potential guillemot to be affected by displacement

For the purposes of assessment it is assumed that 50% of guillemots would be displaced from the proposed development and a surrounding buffer of 1 km.

The value of the offshore site buffered to 1 km as a foraging site for breeding guillemots was estimated for the regional breeding population from the proportion of the adults likely to be at sea (not attending a colony) that were on average present and on the sea (Table 4.79). This is likely to overestimate the importance for feeding as it is likely that some individuals that were present were not feeding.

Table 4.79 The mean number of guillemots present in the offshore site and 1 km buffer in each period of the year and this figure expressed as the percentage of the (at-sea) receptor population

Receptor population	Population size (adults, colony count *1.666)	No. assumed at sea	Dev. Area		Dev. Area + 1km	
			Mean no. at risk	%	Mean no. at risk	%
National, colony-attend period (Seabird 2000)	2,208,812	883,525	737	<0.1	930	0.1
National, chicks-on-sea period (Seabird 2000)	2,208,812	2,208,812	311	<0.1	499	<0.1
North Sea, post-breeding period (Skov *et al.*, 1995)	1,426,100	1,426,100	3,588	0.3	5,111	0.4
North Sea, non-breeding period (Skov *et al.*, 1995)	1,562,400	1,562,400	665	<0.1	879	<0.1
Regional, colony-attend period (Seabird 2000 x 13% decline)	312,151	124,860	737	0.6	930	0.7
Regional, chicks-on-sea period (Seabird 2000 x 13% decline)	312,151	312,151	311	<0.1	499	0.2
Regional, post-breeding period (Skov *et al.*, 1995)	274,050	274,050	3,588	1.3	5,111	1.9
Regional, non-breeding period (Skov *et al.*, 1995)	521,000	521,000	665	0.1	879	0.2

On this basis, it is inferred that proportion of the foraging resources provided by the offshore site buffered to 1 km for the regional population present during the different periods of the year is as follows (Table 4.79):

- 0.7% during the colony-attendance part of the breeding season;
- 0.2% during the chicks-on-sea part of the breeding season;
- 1.9% during the post-breeding period;
- 0.2% during the winter period.

Likely impacts of displacement on guillemot population

A recent study modelling predicted displacement for guillemots breeding on the Isle of May resulting from the development of Neart na Gaoithe Wind Farm demonstrated that displacement of foraging seabirds from an offshore wind farm can result in changes to their time/energy budgets that could impact on their breeding performance and/or survival.

This study also concluded that the impact of displacement is driven by two main processes: 1) the increased travelling costs to the subset of the population that is displaced or for which the wind farm forms a barrier to movement, and 2) the reduction in average prey densities in the remaining habitat due to intensified intra-specific competition, affecting not just displaced birds but the population as a whole.

The model compared the time/energy budget of 1,000 breeding guillemots over a 24 hour period in the absence or presence of a wind farm. The model was run for scenarios simulating different prey distributions (dispersed or patchy) and different levels of interference competition among guillemots feeding in the same patch.

Under all scenarios, the presence of the Neart na Gaoithe wind farm resulted in an increase in the average costs of foraging. For example, where prey were randomly distributed, mean flight and foraging costs in the absence of the wind farm were 1.18 hours (± 0.60 hours) and 2.19 hours (± 0.96 hours) respectively; equivalent values in the presence of the wind farm were 1.60 (±0.67) hours and 2.58 (± 1.57) hours respectively. Under this scenario, the mean number of birds displaced was 101, and the wind farm was a barrier to movement for 44 birds.

The study concludes that further work would be required to determine the consequences of displacement at population level (McDonald *et al.,* 2012).

For the purposes of this assessment it is cautiously assumed that 50% of guillemots that would otherwise forage in the offshore site buffered to 1 km would be displaced elsewhere.

Colony attendance period

If 50% of guillemots were to be displaced during the breeding season from the offshore site buffered to 1 km the impact of this would be the effective loss of around 0.4% of the foraging habitat of the regional breeding population (50% of value in Table 4.79). This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). The sensitivity of the population to displacement at this time of year is unknown, but is likely to be moderate because birds are attending their breeding colonies and therefore will have high feeding requirements. It is concluded that the effects of displacement on the regional guillemot population during the colony attendance period of the breeding season are ***not significant*** under the EIA Regulations.

Chicks-on-sea period

If 50% of guillemots were to be displaced during the chicks-on-sea period from the offshore site buffered to 1 km the impact of this would be the effective loss of around 0.1% of the foraging habitat of the regional breeding population (50% of value in Table 4.79). This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). The sensitivity of the population to displacement at this time of year is unknown, but is likely to be moderate because some adults are attending dependent flightless young and therefore will have relatively high feeding requirements and limited mobility. It is concluded that the effects of displacement on the regional guillemot population during the chicks-on-sea period of the breeding season are ***not significant*** under the EIA Regulations.

Post-breeding period

If 50% of guillemots were to be displaced during the post-breeding period from the offshore site buffered to 1 km the impact of this would be the effective loss of around 0.9% of the foraging habitat of the post-breeding-period population (50% of value in Table 4.79). This impact is categorised as low magnitude, and temporally long-term and reversible (Table 3.1

and Table 3.2). The sensitivity of the population to displacement at this time of year is unknown, but is likely to be low. Although in this period guillemots are not constrained by attending their colony this is the period when many adults undergo wing moult and will have reduced mobility in the period when they are temporarily flightless. On the other hand, the extent of suitable feeding habitat within the region at this time of year is vast, based on guillemot distribution off the east coast of Britain at this time (Skov *et al.,* 1995). It is concluded that the effects of displacement on the regional guillemot population in the post-breeding period are ***not significant*** under the EIA Regulations.

Non-breeding period

If 50% of guillemots were to be displaced during the non-breeding period from offshore site buffered to 1 km the impact of this would be the effective loss of around 0.1% of the foraging habitat of the non-breeding-period population (50% of value in Table 4.79). This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). The sensitivity of the population at this time of year to displacement is unknown, but is likely to be negligible. It is concluded that the effects of displacement on the regional guillemot population in the non-breeding period are ***not significant*** under the EIA Regulations.

Barrier Effect

Guillemots are considered to have moderate sensitivity to the effects of barriers formed by offshore wind farms (Langston, 2010). The potential effects on guillemots of the proposed wind farm acting as a barrier were assessed only for the part of the breeding season when birds are attending colonies. During this period birds undertake commuting flights to and from feeding grounds and it is the potential for the wind farm to act as a barrier and disrupt these flights that gives cause for concern and the possibility of adverse effects on a population.

The proposed wind farm would potentially form a barrier to commuting birds from all breeding colonies that are closer to the offshore site than typical foraging distance of guillemot during the period of colony attendance. The main guillemot colonies potentially affected are Isle of May, Craigleith, and St Abb's Head. Although in theory birds from more distant colonies could also be affected, the alignment of the proposed wind farm and distance from these colonies make it implausible that barrier effects on birds from these colonies could have more than a negligible effect. Therefore, no attempt is made to quantify it.

For the purposes of assessment the width of the barrier is assumed to extend 1 km either side of the maximum width of the proposed wind farm. The estimated magnitude of the barrier effect to birds from the plausibly affected colonies is summarised in Table 3.6`and Table 3.7. Barrier effects as calculated here concern birds which would otherwise fly over the offshore site to access feeding resources beyond it.

For the Isle of May colony, the offshore site would present a barrier 17.9 km wide, 16.2 km to the north-east. This barrier would potentially affect approximately 33% of the possible flight directions available to guillemots flying out to distances in excess of 16.2 km from the

Isle of May (Table 3.6). Assuming the destinations of affected flights are on average 35 km from the colony (the mean foraging distance is 37.8 km, Thaxter *et al*., 2012), the mean increase in the length of affected flights compared to direct routes to the same destination is estimated at 12.6 % (Table 3.7). The back edge of the barrier formed by the offshore site is approximately 26 km from the Isle of May, well below the mean flight distance of 37.8 km for guillemot. Therefore it is likely that the majority of guillemot flights from this colony in the direction of the offshore site are to intended destinations beyond it.

For Craigleith, the proposed wind farm would present a barrier 17.8 km wide, 31.5 km to the north-east. This barrier would potentially affect approximately 28% of the possible flight directions available to guillemots flying out to distances in excess of 31.5 km from the Craigleith colony (Table 3.6). Assuming the destinations of affected flights are on average 41 km from the colony, the closest possible distance beyond the proposed wind farm, the mean increase in the length of affected flights is estimated at 12.1% (Table 3.7). The back edge of the barrier formed by the offshore site is approximately 41 km from Craigleith, similar to the mean flight distance of 37.8 km for guillemot. Therefore it is likely that only approximately 50% of guillemot flights from this colony in the direction of the offshore site are to intended destinations beyond it.

For the St Abb's Head colony, the proposed wind farm would present a barrier 11.6 km wide 33.4 km to the north. This barrier would potentially affect approximately 9% of the possible flight directions available to guillemots flying out to distances in excess of 33.4 km from the St Abb's Head colony (Table 3.6). Assuming the destinations of affected flights are on average 45 km from the colony, the closest possible distance beyond the proposed wind farm, the mean increase in the length of affected flights is estimated at 5.3% (Table 3.7). The back edge of the barrier formed by the offshore site is approximately 45 km from St Abb's Head, which is slightly further than the mean flight distance of 37.8 km for guillemot. Therefore it is likely that less than half of guillemot flights from this colony in the direction of the offshore site are to intended destinations beyond it.

Of the colonies examined above, the Isle of May and Craigleith colonies could plausibly be adversely affected by more than a negligible amount, and even here the proportion of flights potentially affected and the magnitude of detours is relatively small. Given that these colonies support only approximately 10% of the regional breeding population, the effect of the proposed wind farm acting as a barrier is categorised as negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effect of the proposed wind farm acting as a barrier on the regional guillemot population in the breeding season is ***not significant*** under the terms of the EIA Regulations.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

The presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The extent that birds may be displaced by vessels varies depending on the type and speed of the vessel and possibly the time of year. Guillemots are not thought to be particularly sensitive to vessel movements but birds disturbed by vessels will either swim or fly away to an alternative location until the vessels have passed, after which they may return to the area. Consequently, there may be a localised, short-term effect.

Based on this, possible displacement impacts were categorised as being of low (1-5%) magnitude, and being temporally short-term and reversible (Table 3.1 and Table 3.2). Overall, it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional populations of guillemots in the breeding, post-breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

Collision mortality

Collision risk modelling using a 98.0% avoidance rate predicts that no guillemots would be killed by the proposed wind farm. Therefore, it is not plausible that this species will experience mortality from collision with turbine rotors. The reason why no collisions are predicted is because almost all birds (99.96%) seen in flight during the baseline surveys were below the lowest height of the proposed rotor sweep of turbines.

The likely effect of collision mortality on the baseline mortality rate of the regional guillemot population throughout the year is rated as negligible in magnitude (<1%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of collision mortality on guillemots is ***not significant*** under the terms of the EIA Regulations.

EIA Decommissioning Phase assessment for guillemot

The decommissioning phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

The extent that birds may be displaced by vessels varies depending on the type and speed of the vessel and possibly the time of year. Guillemots are not thought to be particularly sensitive to vessel movements but birds disturbed by vessels will either swim or fly away to an alternative location until the vessels have passed, after which they may return to the area. Consequently, there may be a localised, short-term effect.

During the decommissioning phase there is the potential for the prey species, e.g. sandeels, of guillemot to be displaced. Should this occur then it is predicted that the guillemots will also relocate as they follow the movements of their prey.

Based on this, possible displacement impacts were categorised as being of low (1-5%) magnitude, and being temporally short-term and reversible (Table 3.1 and Table 3.2). Overall, it is concluded that the impacts of displacement during decommissioning operations on the regional populations of guillemots throughout the year is ***not significant*** under the EIA Regulations.

EIA summary of effects combined for guillemot

The impacts of the effects assessed will act in a broadly additive manner on the receptor population. In combination it is judged that the magnitude of the effects on the regional population of guillemots in the breeding season is low (Table 4.80). It is concluded that the overall impact on the regional population of guillemots in the breeding season is **not significant** under the EIA regulations.

The impacts of the effects assessed will act in a broadly additive manner on the receptor population. In combination it is judged that the magnitude of the effects on the regional population of guillemots in the non-breeding-period is negligible (Table 4.81). It is concluded that the overall impact on the regional population of guillemots in the non-breeding-period is ***not significant*** under the EIA regulations.

Table 4.80 Summary of effects on the regional population of guillemot in the breeding period

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Low	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Moderate	***Not significant***
Barrier Effect	Negligible	Long term	Low	***Not significant***
Vessel disturbance	Low	Long term	Moderate	***Not significant***
Collision mortality	Negligible	Long term	Low	***Not significant***
All effects combined	Low	Long term	Low - Moderate	***Not significant***

Table 4.81 Summary of effects on the regional population of guillemot in the non-breeding period

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Low	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat post-breeding period	Low	Long term	Low	***Not significant***
Displacement from foraging habitat non-breeding period	Negligible	Long term	Negligible	***Not significant***
Vessel disturbance	Low	Long term	Moderate	***Not significant***
Collision mortality	Negligible	Long term	Low	***Not significant***
All effects combined	Low	Long term	Low - Moderate	***Not significant***

4.4.27.5 Cumulative Impact Assessment for guillemot

Combining the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts to the regional populations of guillemots in the breeding and post-breeding periods will be as shown in Table 4.82. The individual impacts for displacement and collision are combined by addition to give the cumulative impact. The flight directions from the colonies that are potentially affected by each wind farm acting as a barrier substantially overlap and therefore the cumulative barrier impact from the three developments is not the sum of the individual predicted impacts. The cumulative barrier impact is derived from the overall spread of flight directions at each colony potentially affected by the three developments forming barriers.

The predicted effective loss of up to 5.4% of the foraging habitat of the regional guillemot population during the breeding season (Table 4.82) is rated as an effect of moderate magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional guillemot population during the breeding season is of ***moderate significance*** under the terms of the EIA Regulations.

Table 4.82 Summary of CIA for the three proposed offshore wind farms in south-east Scotland on the regional population of breeding guillemot

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF	Predicted impact from Inch Cape pOWF	Predicted impact from Firth of Forth Round 3 Zone pOWF	Cumulative Impact
Displacement: effective loss of foraging habitat during breeding period (%)	50%	0.4% Negligible	1.4% Low	3.6% Low	5.4% Moderate
Displacement: effective loss of foraging habitat during post-breeding period (%)	50%	0.9% Negligible	0.2% Negligible	0.2% Negligible	≤1.3% Minor
Barrier Effect	100%	Foraging trips of <10% of Forth Islands and St Abb's Head birds increase by 6%. Negligible	Foraging trips of <10% of Fowlsheugh and Forth Islands birds increase by ca. 5%. Negligible	No flights likely to go beyond the offshore site, Negligible	No flights likely to go beyond the offshore site. Negligible
Collision: % increase in assumed annual adult mortality rate. All designs.	0%	No change, Negligible	No change, Negligible	No change, Negligible	No change, Negligible

The predicted effective loss of up to 1.3% of the foraging habitat of the regional guillemot population during the post-breeding season (Table 4.82) is rated as an effect of low magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional guillemot population during the post-breeding period is of ***minor significance*** under the terms of the EIA Regulations.

Barrier effects as assessed here concern birds which would otherwise fly over the offshore site to access feeding resources beyond it. The mean foraging distance from a colony for guillemots is 37.8 km and the mean maximum distance is 84 km (Thaxter *et al*., 2012). This means that almost no flights will be to areas beyond the barrier formed by the three proposed wind farms. As a result any barrier effect will be negligible. The cumulative barrier effect is rated as negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of a barrier effect caused by the three proposed offshore wind farms on the regional population of guillemots in the breeding season is ***not significant*** under the terms of the EIA Regulations.

The amount of guillemot flight activity recorded within the height range of the proposed turbines during baseline surveys was extremely low and therefore the risk of collision strikes is also extremely small. The potential impact of collision mortality on the baseline mortality rate of the regional guillemot population in the breeding season is rated as an effect of negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional guillemot population in the breeding season is ***not significant*** under the terms of the EIA Regulations.

A separate CIA has not been undertaken for the regional guillemot population in the non-breeding period (i.e., the winter months). The predicted displacement and collision mortality due to the offshore site for guillemot in the non-breeding period is negligible and the sensitivity of the population to displacement and barrier effects at this time of year is also negligible. Therefore it is not plausible that the offshore site could contribute to a significant cumulative impact for this receptor population.

4.4.27.6 Mitigation measures for guillemot

The following mitigation measures are suggested for guillemot:

1 - Minimise vessel disturbance by reduced vessel usage during peak periods of guillemot density;

2 - Ensure vessel movement are, as far as practicable, along recognised shipping corridors to and from the wind farm, and that vessels operate at a suitable speed;

3 - Minimise as far as practicable the overall footprint of the wind farm

4.4.28 Razorbill *Alca torda*

4.4.28.1 Status

Razorbills are one of the commonest seabird species in Britain, breeding in large colonies of other seabirds on suitable coastal cliff habitat. There are several large colonies on the east coast, and Seabird 2000 recorded 164,557 individuals breeding in Britain (Mitchell *et al.,* 2004). The closest large colonies to the Neart na Gaoithe study area are at the Isle of May, St Abb’s Head and Fowlsheugh.

Razorbills prey on sandeels and other small fish species (Snow & Perrins 1998). A study in the Netherlands concluded that razorbills are probably more dependent on a specialised diet of small schooling fish such as herring, sprat or sandeels than guillemots, which have a much broader diet (Ouwehand *et al*., 2004).

4.4.28.2 Neart na Gaoithe Study Area

Razorbill was the fifth most frequently recorded seabird recorded on surveys in the Neart na Gaoithe study area in Year 1, with a total of 3,980 birds. Overall, the majority of razorbills (85.0%) were in the buffer area (Table 4.2). Peak numbers were recorded in the offshore site (283 birds) and the buffer area (2,043 birds) in October. A further 3,323 unidentified guillemot/razorbills, and 1,348 unidentified auks were also seen in Year 1.

In Year 2, razorbill was the fifth most frequently recorded species on surveys in the Neart na Gaoithe study area (3,131 birds). The majority of birds (87.4%) were again recorded in the buffer area (Table 4.2). Numbers of razorbills in the offshore site peaked at 135 birds in September. Numbers recorded in the buffer area peaked at 1,681 birds in September. A further 1,532 unidentified guillemots/razorbills and 827 unidentified auks were also seen on the Year 2 surveys.

During the breeding period (April to June), the peak estimated number of razorbills in the offshore site in Year 1 occurred in June (48 birds), with 465 birds in the buffer area in April (Table 4.83) (Figure 4.91). In Year 2, estimated numbers of razorbills occurred in May, with 346 birds in the offshore site, and 1,048 birds in the buffer area.

During the chick period (July and August), the peak estimated number of razorbills occurred in August of Year 1, with 1,500 birds in the offshore site, and 4,359 birds in the buffer area (Table 4.83) (Figure 4.91). In Year 2, estimated numbers of razorbills peaked in July, with 376 birds in the offshore site, and 1,687 birds in the buffer area.

In the post-breeding season (September and October), peak estimated numbers of razorbills occurred in October, with 3,054 birds in the offshore site, and 19,892 birds in the buffer area in Year 1 (Table 4.83) (Figure 4.91). In Year 2, peak estimated numbers of razorbills also occurred in October, with an estimated 877 birds in the offshore site, and 14,165 birds in the buffer area.

In the non-breeding season (November to March), peak estimated numbers of razorbills in the offshore site occurred in November of Year 1 (264 birds), with 986 birds in the buffer area in March (Table 4.83) (Figure 4.91). In Year 2, peak estimated numbers of razorbills in the offshore site occurred in December (303 birds), with 997 birds in the buffer area in February.

Estimated numbers of razorbills in the offshore site exceeded 1% of the national breeding population (1,646 birds) (Mitchell *et al.,* 2004) in October of Year 1. Estimated numbers in the buffer area exceeded 1% of the bio-geographic breeding population (5,300 birds) (Mitchell *et al.,* 2004) in October of Years 1 and 2 (Figure 4.91).

Table 4.83 Estimated numbers of razorbills in the offshore site and 8 km buffer in Years 1 and 2

Month	Estimated nos on water Offshore Site	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying Offshore Site	Estimated total Offshore Site	Estimated total 8 km buffer [1]	Estimated total
Yr1 Nov	257	159	414	7	264	670	934
Yr1 Dec	170	88	330	0	170	539	709
Yr1 Jan	0	0	0	7	7	22	29
Yr1 Feb	14	6	31	0	14	328	342
Yr1 Mar	43	26	72	7	50	986	1,036
Yr1 Apr	0	0	0	20	20	465	485
Yr1 May	0	0	0	27	27	278	305
Yr1 Jun	48	23	101	0	48	259	307
Yr1 Jul	0	0	0	7	7	200	207
Yr1 Aug	1,500	1,130	1,992	0	1,500	4,359	5,859
Yr1 Sep	748	548	1,021	0	748	2,777	3,525
Yr1 Oct	3,054	2,232	4,178	0	3,054	19,892	22,946
Yr2 Nov	-	-	-	-	-	-	-
Yr2 Dec	296	184	478	7	303	537	840
Yr2 Jan	0	0	0	0	0	293	293
Yr2 Feb	51	29	89	0	51	997	1,048
Yr2 Mar	50	25	103	0	50	496	546
Yr2 Apr	109	49	243	14	123	543	666
Yr2 May	306	158	593	40	346	1,048	1,394
Yr2 Jun	16	7	37	20	36	440	476
Yr2 Jul	376	205	691	0	376	1,687	2,063
Yr2 Aug	109	57	208	0	109	742	851
Yr2 Sep	191	115	318	0	191	3,085	3,276
Yr2 Oct	796	601	1,054	81	877	14,165	15,042

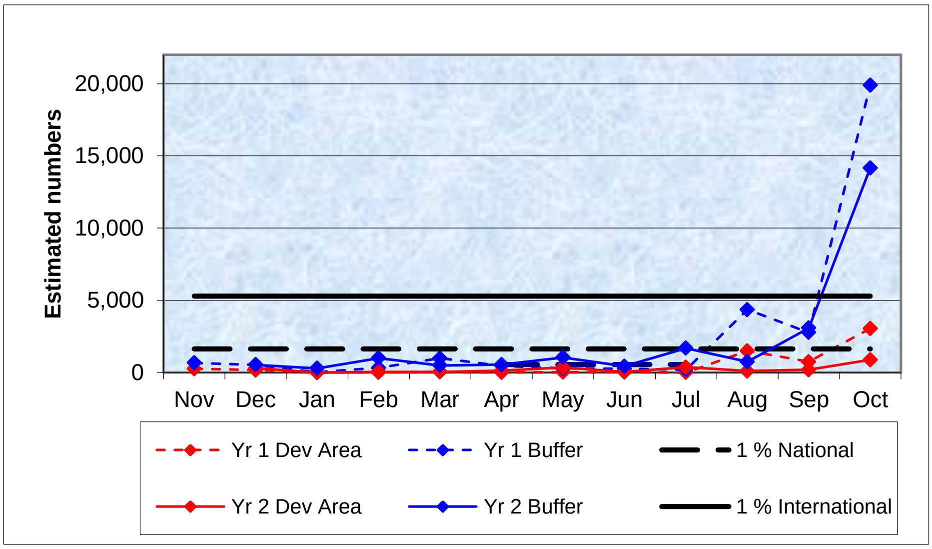


Figure 4.91 Monthly estimated numbers of razorbills in the Neart na Gaoithe Development & buffer areas in Years 1 and 2

Mean monthly razorbill density was generally similar between the offshore site and the buffer area, in Years 1 and 2 (Figure 4.92). In both years, mean density in the offshore site peaked in October, with 36.9 birds/km^2 in Year 1, and a lower peak of 7.3 birds/km^2 in Year 2. A similar pattern was recorded in the buffer area, with 51.5 birds/km^2 recorded in October of Year 1, compared to 23.6 birds/km^2 in October of Year 2. In comparison, ESAS mean density data for the surrounding ICES rectangles peaked in July (6.4 birds/km^2), with a lower peak of 1.9 birds/km^2 in Regional Sea 1 in September and October.

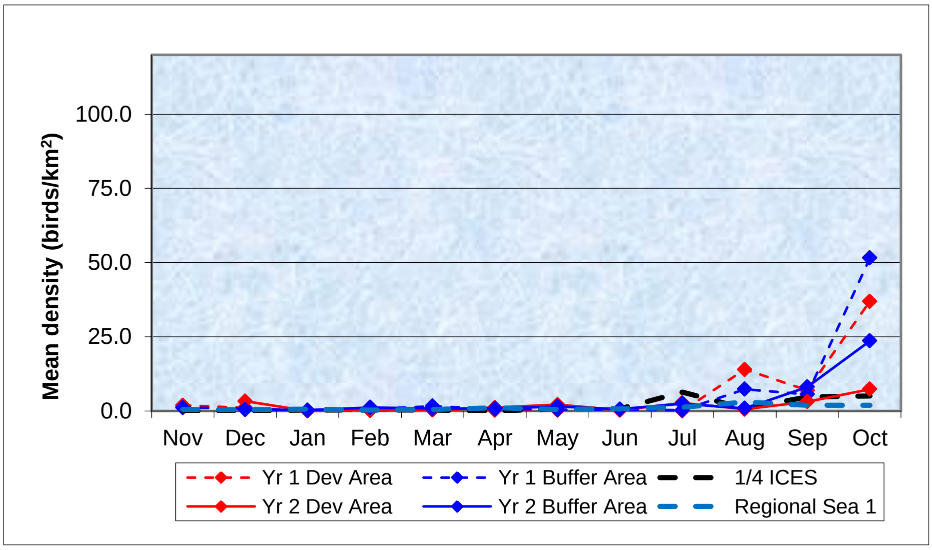


Figure 4.92 Comparison of monthly mean densities for razorbill in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

Between November and March of Year 1, razorbills were widespread throughout the study area at mostly low densities, with higher densities in the north of the buffer area (Figure 4.93). Generally fewer razorbills were recorded in the south-west of the buffer area at this time.

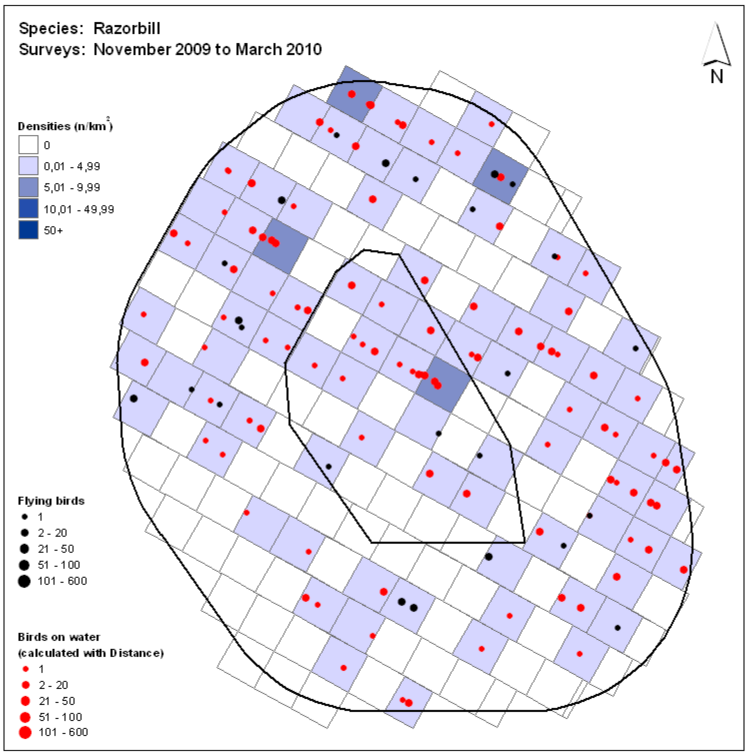


Figure 4.93 Razorbill density between November and March, Year 1

Razorbill distribution over the same period in Year 2 was broadly similar, with mostly low densities of razorbills recorded in the offshore site, and in the south and east of the buffer area. Higher densities were recorded sporadically in the north of the study area at this time (Figure 4.94).

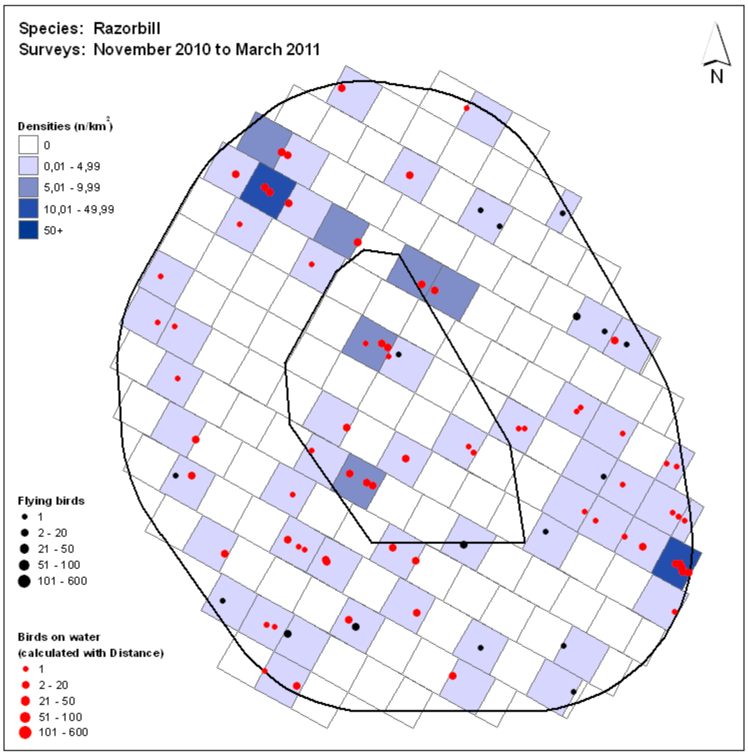


Figure 4.94 Razorbill density between November and March, Year 2

Between April and June of Year 1, razorbills were scattered at mostly low densities across the study area, with low densities in the offshore site (Figure 4.95). In the buffer area, highest densities were recorded in the south-east at this time.

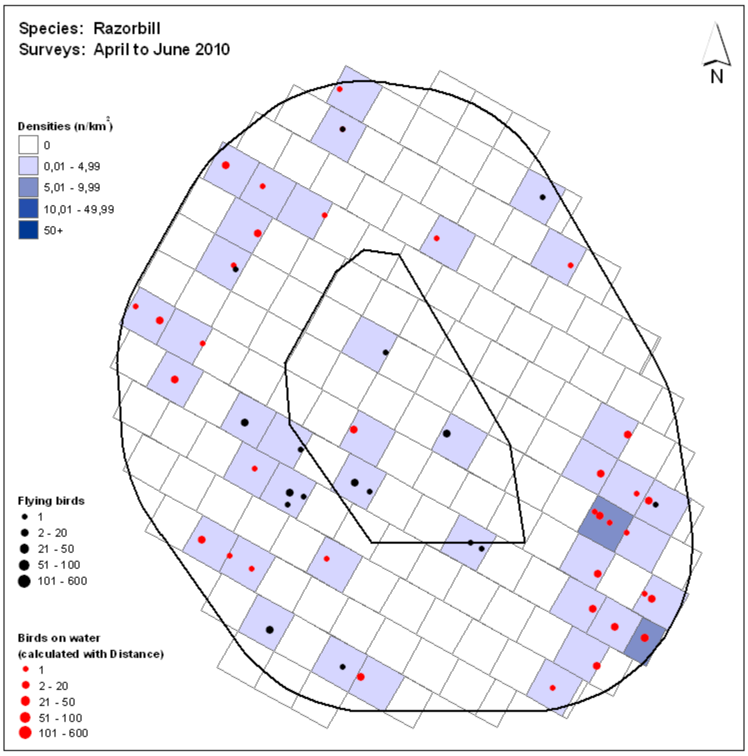


Figure 4.95 Razorbill density between April and June, Year 1

Razorbills were more widespread between April and June of Year 2. In the offshore site, moderate densities were recorded in the south-east (Figure 4.96). Densities in the buffer area were mostly low at this time, with highest densities in the south-east.

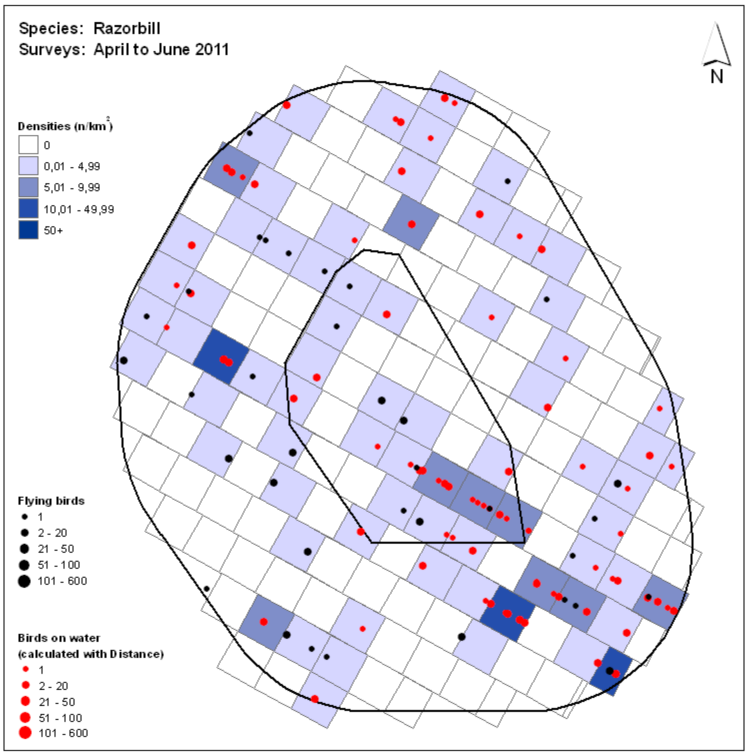


Figure 4.96 Razorbill density between April and June, Year 2

Razorbills were more widespread in the study area in July and August of Year 1, compared to between April and June. Birds occurred throughout the offshore site at mostly moderate to high densities at this time (Figure 4.97). In the buffer area, razorbills were similarly distributed, although birds were absent from parts of the south-west and north at this time.

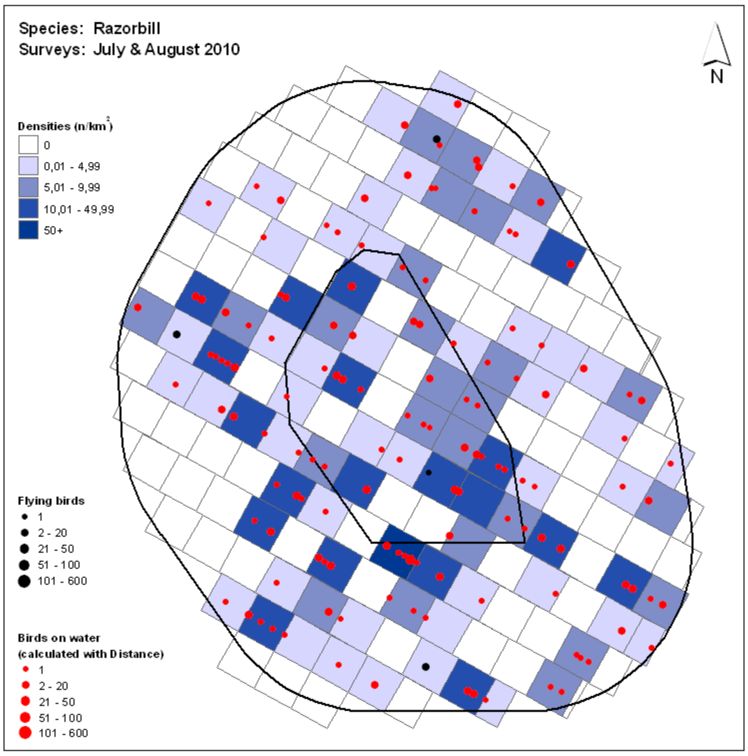


Figure 4.97 Razorbill density in July and August, Year 1

In July and August of Year 2, highest densities of razorbills were recorded in the north of the buffer area, with fewer birds in the south (Figure 4.98). In the offshore site, most razorbills were recorded in the east, at moderate to high densities at this time.

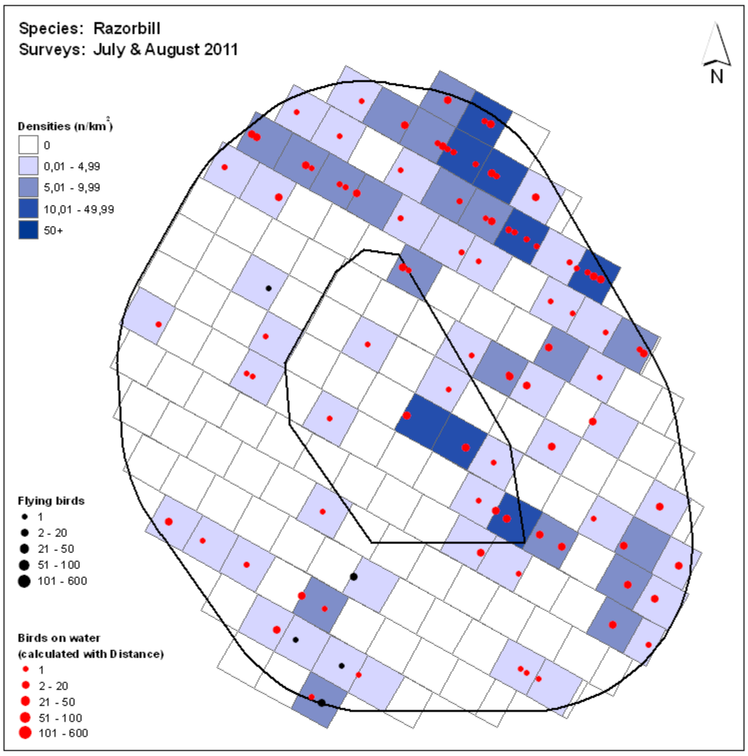


Figure 4.98 Razorbill density in July and August, Year 2

In Year 1, peak densities of razorbills were recorded in September and October (Figure 4.99). Birds were widespread at mostly high densities across the offshore site and most of the buffer area at this time, apart from in the south-east of the buffer area, where densities were lower or zero.

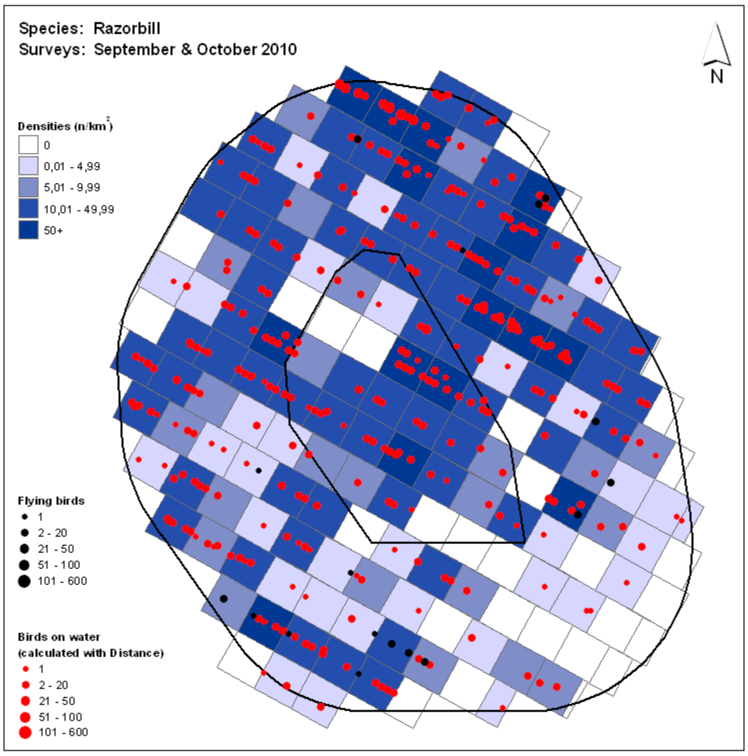


Figure 4.99 Razorbill density in September and October, Year 1

A similar pattern was recorded in September and October of Year 2, with highest densities of razorbills in the north and west of the buffer area, with patches of higher densities in the south and east (Figure 4.100). Razorbill densities in the offshore site at this time were lower than in the same period of Year 1.

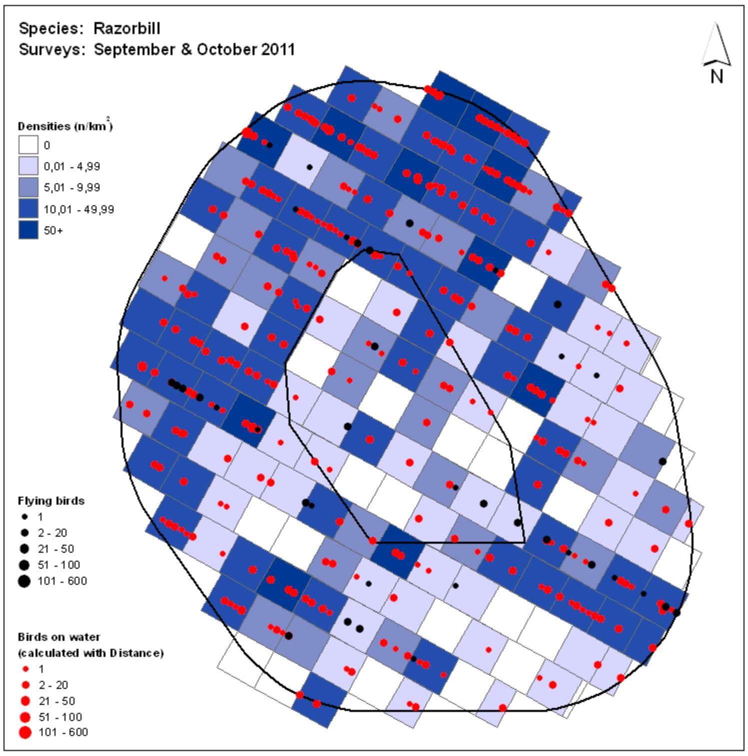
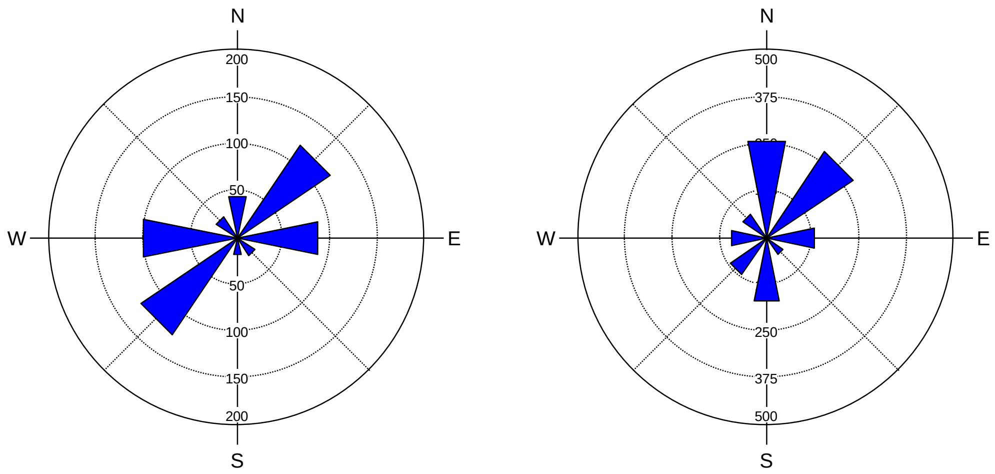


Figure 4.100 Razorbill density in September and October, Year 2

In Years 1 and 2, 1,726 razorbills were recorded in flight, with all birds recorded flying below 22.5 m in height. The majority (98.4%) were recorded flying below 7.5 m in height (Table 4.3).

Flight direction was recorded for 544 razorbills in the breeding season (April to July), with direction recorded for 1,181 razorbills in the post-breeding and non-breeding seasons (August to March) (Figure 4.101).

In the breeding season, just over a fifth of all birds recorded were flying south-west (22.8%), or north-east (21.9%), with slightly lower numbers flying east/west. In the non-breeding season, just over a fifth of all birds recorded were flying north-east (21.7%) with lower numbers flying north (15.8%).



April to July (n=544 birds) August to March (n=1,181 birds)
Numbers shown on figures are number of birds recorded

Figure 4.101 Flight direction of razorbills in the Neart na Gaoithe Study Area in Years 1 and 2

Recent tracking studies on 18 razorbills breeding on the Isle of May undertaken by CEH at the request of FTOWDG indicate that razorbills from the Isle of May use both coastal and offshore areas, with a mean maximum range of 14 km and a maximum of 69 km, although they avoided the deeper water between the Isle of May and the Wee Bankie. In addition, the ranges recorded during this study for razorbill were intermediate when compared with historical data. The study also indicated that razorbills did not use the Neart na Gaoithe site for non-flight activities such as foraging or resting (Daunt *et al.,* 2011a).

4.4.28.3 Species sensitivity

Razorbill is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009). A recent review assessed razorbill as being at moderate risk of displacement, barrier effects, habitat loss or changes in prey distribution resulting from offshore wind farms, while collision risk was rated as low risk. Overall, razorbill was assessed as being at moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

Razorbill is listed as a qualifying interest species in the breeding season for three SPAs on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development (Table 4.84). These SPAs held 8.7% of the UK breeding population and 1.5% of the biogeographic population at the time of designation (JNCC, 2012). The distance between the offshore site and two of these SPAs (Forth Islands SPA and St Abb's Head to Fast Castle SPA) is within the mean maximum foraging range of razorbill (48.5 km). The distance to Fowlsheugh SPA is within the maximum known foraging range (95 km) (Thaxter *et al.*, 2012). Razorbill mean maximum foraging range from breeding SPAs in relation to the offshore site are shown in Figure 4.102.

Table 4.84 SPAs for breeding razorbill between Peterhead and Blyth

SPA site	Distance to site (km)	Site total (pairs) [1]	% of biogeo-graphic popn [1]	% of national popn [1]	Recent count [2]	Year
Forth Islands	**16**	**2,693**	**0.5**	**2.7**	**3,580**	**2009**
Fowlsheugh	*62*	*4,576*	*0.8*	*4.6*	*4,632*	*2009*
St Abb's Head to Fast Castle	**31**	**1,407**	**0.2**	**1.4**	**1,687**	**2008**
Total	**-**	**8,976**	**1.5**	**8.7**	**9,899**	**-**

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. Sites in italics lie within the maximum known foraging range of 95 km. Sites in bold lie within the mean maximum foraging range of 48.5 km (Thaxter *et al.*, 2012).

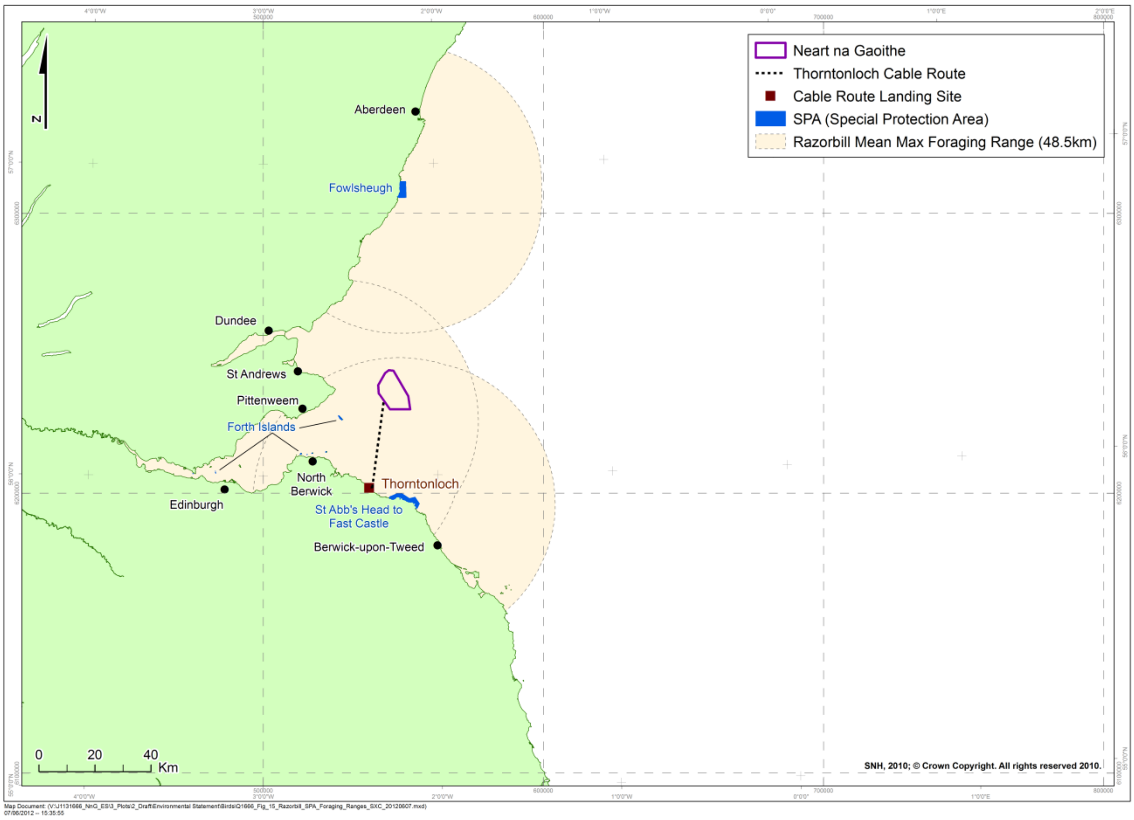


Figure 4.102 Razorbill mean maximum foraging range from breeding SPAs in relation to the Development

Razorbill mean maximum foraging range from breeding SPAs in relation to the offshore site are shown in Figure 4.102.

4.4.28.4 Assessment

Treatment of unidentified razorbills

For the purposes of assessment the unidentified birds seen during surveys that could have been either razorbills or guillemots were included in population estimates by apportioning them in accordance to the ratio of positively identified individuals on survey visits. This was considered to be the best way of dealing with the issue of unidentified individuals as it makes best use of the available data without introducing obvious biases. It is also the method recommended by Maclean *et al*., (2009) for dealing with the issue of unidentified individuals.

Breeding population size correction

The published estimates of breeding population size for razorbill are based on the number of individuals present at breeding colonies during census counts. These are inevitably an underestimate of the number of breeding adults in the population as a substantial proportion of adults will be absent from the colony, away foraging, at the time of the census count. For the purposes of assessment the likely total number of adults in the breeding population was calculated by dividing the colony count estimate by the adult colony attendance rate. For example, if the mean colony attendance rate is 65%, a census count of 10,000 adults would translate to an estimated breeding population size of 15,385 adults. This method of calculating breeding population size is necessarily approximate, however, it is important to apply a correction of some form because basing assessments on 'raw' colony count data would introduce a large bias into the assessment process. Mean colony attendance rates were taken from published studies based on detailed information from tagged individuals. The mean colony attendance rate used for razorbill was 65% (Benvenuti *et al.,* 2001).

Definition of seasons

The annual cycle for razorbills is divided into four parts to reflect the biology of the species and the broad pattern of use of the offshore site.

The main breeding season, the period when breeding adults are attending colonies, is defined as April to June. At this time the vast majority of birds present in the offshore site will be from relatively local breeding colonies (those within foraging range).

The chick period is defined as July and August and corresponds to the time when successful male adults have accompanying dependent young with them on the sea. This period is clearly part of the breeding season, however adults are no longer constrained by having to visit the breeding colony. Indeed it is known that adults and their dependent young can quickly move away from colonies to feeding areas some distance away. Nevertheless, it is likely that almost all individuals present in the offshore site during the chick period are from breeding sites within the region, particularly the relatively close large colonies. Some adult razorbill undergo wing moult in the chick period (Cramp & Simmons 1983).

The post-breeding period was defined as September and October. Although birds at this time of year are no longer breeding they were examined separately to the later part of the non-breeding period (the 'winter') because the numbers present were much greater and it is likely that the vast majority of individuals present at this time are still from breeding sites within the region, particularly the relatively close large colonies. Also, a proportion of adult razorbill undergo wing moult in this period (Cramp & Simmons 1983) which increase their sensitivity to disturbance and displacement effects.

The main non-breeding period is defined as November to March and broadly corresponds to the period when razorbills are in their over-wintering area. In this period it is likely that a high proportion of individuals present in the offshore site are from breeding colonies outwith the region, including birds from other countries (Wernham *et al.,* 2002).

Baseline conditions

In the colony attendance period (April to June) the mean estimated number of razorbills present in the offshore site was 101 birds, and the mean estimated number in the offshore site buffered to 1 km was 124 birds. On average 84.5% of the birds present were on the sea and the remainder were in flight.

For the purposes of assessment it is assumed that all birds in summer plumage were breeding birds. Studies have estimated that mean colony attendance rate of razorbill during this part of the breeding season is 65% (Benvenuti *et al.,* 2001) and this figure is used in estimates of the importance of the offshore site for foraging.

In the chicks-on-sea part of the breeding season period (July and August) the mean estimated number of adult razorbills present in the offshore site was 215 birds, and the mean estimated number in the offshore site buffered to 1 km was 301 birds. In this period 95.6% of the 900 birds recorded during baseline surveys were sitting on the sea, and 15.0% were juveniles.

In the post-breeding period (September and October) the mean estimated number of adult razorbills present in the offshore site was 1,429 birds, and the mean estimated number in the offshore site buffered to 1 km was 2,055 birds. In this period 98.1% of the birds present were on the sea.

In the non-breeding period (November to March) the mean estimated number of razorbills present in the offshore site was 116 birds, and the mean number in the offshore site buffered to 1 km was 190 birds. In this period 90.1% of the birds present were on the sea.

Populations

Colony-attendance period

The breeding population of razorbills in Scotland has undergone a prolonged period of decline and recent colony counts indicate that the decline is on-going (SMP, 2012). Assuming that the subset of colonies that have been recently counted are representative, the regional decline since the Seabird 2000 counts amounts to -14%. The decline in breeding numbers means that published figures on population size for this species (e.g., Mitchell *et al*., 2004) no longer accurately reflect the current population size in eastern Scotland. To prevent this causing assessment of effects to be biased low, the regional total derived from Seabird 2000 results has been adjusted downwards by 14 %. On this basis, the regional breeding population is assumed to be 17,385 adults in terms of colony counts, which translates to a population size of 26,737 breeding adults.

This figure is similar to the estimate that can be derived from the results given by Skov *et al*. (2005) for the March to June period. Their results suggest an at-sea regional (locality 7 plus half of locality 5) population at this time of year of around 11,900 birds. Assuming that 35% of the breeding population are at sea at any one time and that there has been a 14% decline since the Skov *et al*. (2005) survey work, this would suggest a current regional breeding population of around 29,240 birds, only 11% higher than the figure based on colony counts.

Chick-rearing period

The regional estimate derived from the results in Skov *et al.* (1995) for the July to September period (Localities 7, 8 and 9 and half of 6) is 94,200 birds, and this reduces to 81,012 birds if a 14% decline is assumed. This is at least six times greater than the estimate for March to June, when birds are attending breeding colonies, as derived above from the Skov *et al*. (2005) results. The difference between the Skov *et al.* (2005) derived estimates for March to June and July to September periods is strong evidence that there is a large influx of razorbills into the region from elsewhere soon after the birds stop attending colonies (typically this is around early-mid July). It is likely that birds moving into the region at this time are failed breeders of both sexes and successful breeding females. Successful males from other breeding regions such as the Moray Firth and Norway are unlikely to be present in this period, at least initially, as these will be accompanying their dependent (and flightless) chick. Therefore, they are restricted to swimming to relocate, although over a period of weeks they can potentially swim large distances).

The estimate derived from Skov *et al.* (2005) for July to September covers a longer period than the chick-rearing period defined for the assessment (July and August). To reduce the possibility of this causing a bias (e.g., this could be caused if Skov *et al*. (1995) surveys had particularly high numbers present in September) and build caution into assessments, the population size present in the region during the chick-rearing period is assumed to be the mean of the regional breeding population size (26,737 adults) and the Skov *et al.* (1995) derived estimate for July to September (81,012 birds); this gives an assumed population size of 52,429 birds.

Post-breeding period

The size of the post-breeding-period regional razorbill population is assumed to be 64,715 birds, this figure is derived from the results given in Skov *et al*. (1995) on the following basis. The seasonal periods used by Skov *et al*. (1995) do not correspond to the defined post-breeding-period (September and October) used for the EIA. Therefore, the assumed population present at this time is the mean of the Skov *et al*. (1995) derived estimates for the July to September period (94,200 birds, using localities described earlier) and the October to November period (57,000, based on localities 6 and 7. This gives a mean of 75,600 birds.

Non-breeding period

The size of the non-breeding (winter) period regional razorbill population is assumed to be 14,400 birds. This is derived by summing the December to February period estimates for localities 3, 4, and 5 given in Skov *et al*. (1995). No reduction has been made for recent declines in the regional breeding population as at this time of year it is likely that most birds present are from breeding grounds outside the region.

Nature conservation importance

For EIA assessment purposes, the nature conservation importance of razorbill using the offshore site is categorised as high during the breeding season. The species is classed as high NCI because the offshore site is at times likely to have >1% of the population from at least one SPA breeding colony where razorbill is a qualifying interest. The mean numbers present in the offshore site buffered to 1 km exceed 1% of the (at-sea) regional population during the colony attendance part of the breeding season and the winter period, and exceed 5% of the regional population during the post-breeding season. This species is not subject to any special legislative protection, nor is it on any conservation priority lists.

Offshore wind farm studies of razorbill

In general, the evidence of the displacement, barrier and collision effects of existing wind farms on razorbills appears to be similar as those for guillemot, a closely related species. This is partly because the difficulty in identifying between the two species has resulted in undifferentiated records with findings and conclusions grouped as guillemot/razorbill. This is justified because it is assumed that these species respond similarly to wind farm developments (Christensen *et al*., 2003). At Horns Rev, Denmark razorbills/guillemots were totally displaced from the wind farm during the construction phase and showed a reduced selectivity of the wind farm and its buffer during the operational phase (Diersche and Garthe, 2006). By contrast, the modelled results for razorbill from Egmond aan Zee, the Netherlands, identified only one of five surveys where the probability of finding birds within the perimeter of the wind farm was significantly lower than expected on the basis of the general distribution pattern in the larger study area. Some razorbills, like some guillemots, were found amongst the Egmond aan Zee turbines, but unlike guillemots they were never recorded within the adjacent Princess Amalia wind farm where turbine density was higher, suggesting that razorbills may be totally displaced only when turbine density exceeds a particular point (Leopold *et al*., 2011). The authors of this study concluded that the magnitude of the displacement effect for razorbills was less than 50%.

At North Hoyle, Wales, razorbills were recorded within the wind farm perimeter (PMSS, 2006), and at Arklow Bank, Ireland, the numbers of razorbills in the vicinity of the single row of turbines were reported to have increased generally, however there was “no evidence of any relationship between the increase in numbers and the distance to the nearest turbine” (Barton *et al*., 2009).

There is limited evidence in post-construction monitoring reports of razorbill flights deflecting around or away from wind farms. Studies at Egmond aan Zee reported that razorbills showed “strong avoidance behaviour in their flight pattern” in the vicinity of the farm deflecting typically at between 2 km and 4 km from the wind farm perimeter (Lindeboom *et al*., 2011). At Horns Rev, visual monitoring from an observation platform positioned at the edge of the wind farm found that 3.8% (sample size not given) of flying guillemots/razorbills were either within or flying into the wind farm (Diersche and Garthe, 2006). Summarising the barrier effect of wind farms on seabirds in German marine areas, razorbills were categorised as having a strong deflection/avoidance response (Diersche and Garthe, 2006).

The risk of razorbills colliding with wind turbines is likely to be very low based on reported flying heights from existing wind farm studies. Of approximately 1,100 flying razorbills monitored over two years in the vicinity of the Arklow Bank wind farm, Ireland, no birds were recorded flying at a height over 20 m above the sea surface (Barton *et al*., 2009, Barton *et al*., 2010). At North Hoyle, Wales, of 85 birds flying in the vicinity of the wind farm three were flying higher than 20 m above the sea surface. The review of offshore wind farm effects on birds acknowledges the general low flying height of razorbills (Diersche and Garthe, 2006). Evidence from other operational wind farms to some extent corroborates the very low risk of razorbills colliding with turbines with no fatalities recorded in a review of the number of collision victims at wind farms in eight European countries (Hötker *et al*., 2006) although the low probability of detecting such fatalities should be recognised.

EIA Construction Phase assessment for razorbill

The construction phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

The extent that birds may be displaced by vessels varies depending on the type and speed of the vessel and possibly the time of year. Razorbills are not thought to be particularly sensitive to vessel movements but birds disturbed by vessels will either swim or fly away to an alternative location until the vessels have passed, after which they may return to the area. Consequently, there may be a localised, short-term effect.

During the construction phase there is the potential for the prey species, e.g. sandeels, of razorbill to be displaced, particularly during piling activities. Should this occur then it is predicted that the razorbills will also relocate as they follow the movements of their prey.

Noise modelling undertaken indicates that behavioural impacts on sandeels from piling noise is predicted to extend less than 1.5 km from the piling activities (See Chapter 15). Therefore, the effect on razorbills foraging on sandeels is likely to be relatively localised.

Based on this, possible displacement impacts were categorised as being of low (1-5%) magnitude, and being temporally short-term and reversible (Table 3.1 and Table 3.2). Overall, it is concluded that the impacts of displacement during construction operations on the regional population of razorbills throughout the year is ***not significant*** under the EIA Regulations.

EIA Operational Phase assessment for razorbill

Potential for razorbill to be affected by displacement

For the purposes of assessment it is assumed that 50% of razorbills would be displaced from the offshore site and a surrounding buffer of 1 km.

The value of the offshore site buffered to 1 km as a foraging site for razorbills was estimated for the regional breeding population from the proportion of the adults likely to be at sea (not attending a colony) that were on average present and on the sea(Table 4.85).

Table 4.85 The mean number of razorbill present in the offshore site and 1 km buffer in each period of the year expressed as the percentage of the (at-sea) receptor population

Receptor population	Population size (adults, colony count *1.538)	No. assumed at sea	Dev. Area		Dev. Area + 1 km	
			Mean no. at risk	%	Mean no. at risk	%
National, colony-attend period (Seabird 2000)	253,418	88,696	101	0.1	124	0.1
National, chicks-on-sea period (Seabird 2000)	253,418	253,418	215	0.1	301	0.1
North Sea, post-breeding period (Skov *et al*., 1995)	218,620	218,620	1,429	0.7	2,055	0.9
North Sea, non-breeding period (Skov *et al*., 1995)	324,000	324,000	116	<0.1	190	<0.1
Regional, colony-attend period (Seabird 2000 x 14% decline)	26,737	9,358	101	1.1	124	1.3
Regional, chicks-on-sea (with 14% decline). Hybrid estimate, see text.	52,429	52,429	215	0.4	301	0.6
Regional, post-breeding period (Skov *et al*., 1995)	75,600	75,600	1,429	1.9	2,055	2.7
Regional, non-breeding period (Skov *et al*., 1995)	14,400	14,400	116	0.8	190	1.3

On this basis, it is inferred that proportion of the foraging resources provided by the offshore site buffered to 1 km for the regional razorbill population during the different periods of the year was as follows (Table 4.85):

- 1.3% during the colony-attendance part of the breeding season;
- 0.6% during the chicks-on-sea part of the breeding season;
- 2.7% during the post-breeding period;

- 1.3% during the non-breeding (winter) period.

Likely impacts of displacement on razorbill populations

A recent study modelling predicted displacement for guillemots breeding on the Isle of May resulting from the development of Neart na Gaoithe Wind Farm demonstrated that displacement of foraging guillemots from an offshore wind farm can result in changes to their time/energy budgets that could impact on their breeding performance and/or survival.

Under all scenarios, the presence of the Neart na Gaoithe wind farm resulted in an increase in the average costs of foraging. The study concludes that further work would be required to determine the consequences of displacement at population level (McDonald *et al.,* 2012).

Although this modelling work was conducted for guillemot, it is likely that results for razorbill would be similar, as the two species are closely related. For the purposes of this assessment it is cautiously assumed that 50% of razorbills that would otherwise forage in the offshore site buffered to 1 km would be displaced elsewhere.

Colony-attendance period

If 50% of razorbills were to be displaced from the offshore site buffered to 1 km the impact of this would be the effective loss of around 0.7% of the foraging habitat of the regional breeding population during the colony-attendance period (50% of the value in Table 4.85). This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). The sensitivity of the population to displacement at this time of year is unknown, but is likely to be moderate because birds are attending breeding colonies, which constrains their foraging, and they have additional energy demands associated with breeding. It is concluded that the impact of displacement on the regional razorbill population during the colony-attendance period of the breeding season is ***not significant*** under the EIA Regulations.

Chicks-on-sea period

If 50% of razorbills were to be displaced from the offshore site buffered to 1 km the impact of this would be the effective loss of around 0.3% of the foraging habitat of the regional breeding population during the chicks-on-sea period (50% of the value in Table 4.85). This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). The sensitivity of the population to displacement at this time of year is unknown, but is likely to be low or moderate. In this period birds are no longer constrained in where they go by their attendance at a breeding colony. However, successful males will be rearing chicks and this will constrain their movements and impose additional feeding demands. It is concluded that the impact of displacement on the regional razorbill population during the chicks-on-sea period of the breeding season is ***not significant*** under the EIA Regulations.

Post-breeding period

If 50% of razorbills were to be displaced during the post-breeding period from the offshore site buffered to 1 km the impact of this would be the effective loss of around 1.4% of the foraging habitat of the post-breeding-period population (50% of the value in Table 4.85). This impact is categorised as low magnitude, and temporally long-term and reversible (Table

3.1 and Table 3.2). The sensitivity of the population to displacement at this time of year is unknown, but is likely to be moderate. Although in this period razorbills are not constrained by attending the breeding colony this is the period when many adults undergo wing moult and so they have reduced mobility when they are flightless. Furthermore, the extent of favoured feeding habitat at this time of year is apparently relatively restricted with birds congregating in favoured areas such as the Firth of Forth and Moray Firth (Skov *et al.,* 1995). It is concluded that the impact of displacement on the regional razorbill population in the post-breeding period is an effect of ***minor significance*** under the EIA Regulations.

Non-breeding period

If 50% of razorbills were to be displaced during the winter period from the offshore site buffered to 1 km the impact of this would be the effective loss of around 0.7% of the foraging habitat of the non-breeding-period population (50% of the value in Table 4.85). This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). The sensitivity of the population to displacement at this time of year is unknown, but is likely to be negligible. It is concluded that the effects of displacement on the regional razorbill population in the non-breeding period are ***not significant*** under the EIA Regulations.

Barrier Effect

Razorbills are considered to have moderate sensitivity to the effects of barriers formed by offshore wind farms (Langston, 2010). The potential effects on razorbill of the proposed wind farm acting as a barrier were assessed only for the part of the breeding season when birds are attending colonies. During this period birds undertake commuting flights to and from feeding grounds and it is the potential for the wind farm to act as a barrier and disrupt these flights that gives cause for concern and the possibility of adverse effects on a population.

The proposed wind farm would potentially form a barrier to commuting birds from all breeding colonies that are closer to the offshore site than the typical foraging distance of razorbills during the period of colony attendance. The main razorbill colonies potentially affected are Isle of May, Craigleith and St Abb's Head. Although in theory birds from more distant colonies could also be affected, the alignment of the proposed wind farm and distance from these colonies make it implausible that barrier effects on birds from these colonies could have more than a negligible effect. Therefore, no attempt is made to quantify it.

For the purposes of assessment the width of the barrier is assumed to extend 1 km either side of the maximum width of the proposed wind farm. The estimated magnitude of the barrier effect to birds from the plausibly affected colonies is summarised in Table 3.6 and Table 3.7. Barrier effects as calculated here concern birds which would otherwise fly through the offshore site to access feeding resources beyond it.

For the Isle of May colony, the proposed wind farm would present a barrier 17.9 km wide, 16.2 km to the north-east. This barrier would potentially affect approximately 33% of the possible flight directions available to razorbills flying out to distances in excess of 16.2 km from the Isle of May (Table 3.6). Assuming the destinations of affected flights are on average 25.6 km from the colony, i.e. immediately beyond the wind farm, the mean increase in the length of affected flights is estimated at 28.4%, equating to about 7 km (Table 3.7). The back edge of the barrier formed by the offshore site is approximately 26 km from the Isle of May, similar to the mean foraging distance of 23.7 km for razorbill (Thaxter *et al.,* 2012). Therefore it is likely that only approximately 50% of razorbill flights from this colony in the direction of the offshore site are to intended destinations beyond it. This would mean that approximately 17% of foraging flights (half the flights in the 33% of possible directions possibly affected) could be affected by a barrier effect.

For Craigleith, the proposed wind farm would present a barrier 17.8 km wide, 31.5 km to the north-east. This barrier would potentially affect approximately 28% of the possible flight directions available to razorbills flying out to distances in excess of 31.5 km from the Craigleith colony (Table 3.6). Assuming the destinations of affected flights are on average 41 km from the colony, the closest possible distance beyond the proposed wind farm, the mean increase in the length of affected flights is estimated at 12.1% (Table 3.7). The back edge of the barrier formed by the offshore site is approximately 41 km from Craigleith, which is almost twice the mean foraging distance of 23.7 km for razorbill and approaching the mean maximum foraging range (48.5 km). Therefore it is likely that only a small minority of razorbill flights from this colony in the direction of the offshore site are to intended destinations beyond it.

For the St Abb's Head colony, the proposed wind farm would present a barrier 11.6 km wide 33.4 km to the north. This barrier would potentially affect approximately 9% of the possible flight directions available to razorbills flying out to distances in excess of 33.4 km from the St Abb's Head colony (Table 3.6). Assuming the destinations of affected flights are on average 45 km from the colony, the closest possible distance beyond the proposed wind farm, the mean increase in the length of affected flights is estimated at 5.3% (Table 3.7). The back edge of the barrier formed by the offshore site is approximately 45 km from St Abb's Head, almost twice the mean foraging distance of 23.7 km for razorbill and similar to the mean maximum foraging range (48.5 km) (Thaxter *et al.,* 2012). Therefore it is likely that only a small minority of razorbill flights from this colony in the direction of the offshore site are to intended destinations beyond it.

Of the three colonies examined, only the Isle of May colony could plausibly be adversely affected by more than a negligible amount, and even here the proportion of flights potentially affected and the magnitude of detours is relatively small. The Isle of May supports approximately 22% of the regional breeding population of razorbills, and approximately 17% of flights may be affected by barrier effects, i.e. approximately 4% of the flights in the region. The effect of the proposed wind farm acting as a barrier to the flights of razorbills of the regional breeding population is categorised as an effect of low magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effect of the proposed wind farm acting as a barrier on the regional breeding razorbill population is of ***minor significance*** under the terms of the EIA Regulations.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

The presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The extent that birds may be displaced by vessels varies depending on the type and speed of the vessel and possibly the time of year. Razorbills are not thought to be particularly sensitive to vessel movements but birds disturbed by vessels will either swim or fly away to an alternative location until the vessels have passed, after which they may return to the area. Consequently, there may be a localised, short-term effect.

Based on this, possible displacement impacts were categorised as being of low (1-5%) magnitude, and being temporally short-term and reversible (Table 3.1 and Table 3.2). Overall, it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional populations of razorbills in the breeding, post-breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

Collision mortality

Collision risk modelling using a 98.0% avoidance rate predicts that no razorbills will collide with the proposed turbines. Therefore, it is not plausible that this species will experience mortality from collision with turbine rotors. The reason why no collisions are predicted is because all razorbill seen in flight during the baseline surveys were below the lowest height of the proposed rotor sweep of turbines.

The likely effect of collision mortality on the baseline mortality rate of the regional razorbill population throughout the year is rated as negligible in magnitude (<1%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of collision mortality on razorbills is ***not significant*** under the terms of the EIA Regulations.

4.4.28.5 EIA Decommissioning Phase assessment for razorbill

The decommissioning phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

The extent that birds may be displaced by vessels varies depending on the type and speed of the vessel and possibly the time of year. Razorbills are not thought to be particularly sensitive to vessel movements but birds disturbed by vessels will either swim or fly away to an alternative location until the vessels have passed, after which they may return to the area. Consequently, there may be a localised, short-term effect.

During the decommissioning phase there is the potential for the prey species, e.g. sandeels, of razorbill to be displaced. Should this occur then it is predicted that the razorbills will also relocate as they follow the movements of their prey.

Based on this, possible displacement impacts were categorised as being of low (1-5%) magnitude, and being temporally short-term and reversible (Table 3.1 and Table 3.2). Overall, it is concluded that the impacts of displacement during decommissioning operations on the regional populations of razorbills in the breeding, post-breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

4.4.28.6 EIA summary of effects combined for razorbill

The impacts of the effects assessed will act in a broadly additive manner on the receptor population. In combination it is judged that the magnitude of the effects on the regional razorbill population in the breeding season is low (Table 4.86). It is concluded that the overall impact on the regional population of razorbills in the breeding season is of ***minor significance*** under the EIA regulations.

Table 4.86 Summary of effects on the regional population of razorbills in the breeding season

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Low	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat, colony-attendance period	Negligible	Long term	Moderate	***Not Significant***
Displacement from foraging habitat, chick-on-sea period	Negligible	Long term	Moderate	***Not Significant***
Barrier Effect, colony-attendance period	Low	Long term	Moderate	***Minor significance***
Vessel disturbance	Negligible	Long term	Minor	***Not significant***
Collision mortality	Negligible	Long term	Low	***Not significant***
All effects combined	Low	Long term	Moderate	***Minor significance***

In combination it is judged that the magnitude of the effects on the regional razorbill population in the non-breeding-period is low (Table 4.87). It is concluded that the overall impact on the regional population of razorbills in the post-breeding and non-breeding-period is of ***minor significance*** under the EIA regulations.

Table 4.87 Summary of effects on the regional population of razorbill in the post-breeding and non-breeding-periods

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Low	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat, post-breeding period	Low	Long term	Moderate	***Minor significance***
Displacement from foraging habitat, non-breeding period	Negligible	Long term	Negligible	***Not significant***
Vessel disturbance	Negligible	Long term	Minor	***Not significant***
Collision mortality	Negligible	Long term	Low	***Not significant***
All effects combined	Low	Long term	Low	***Minor significance***

4.4.28.7 Cumulative Impact Assessment for razorbill

Combining the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts to the regional populations of razorbill in the breeding and post-breeding periods will be as shown in Table 4.88. The individual impacts for displacement and collision are combined by addition to give the cumulative impact. The flight directions from the colonies that are potentially affected by each wind farm acting as a barrier substantially overlap and therefore the cumulative barrier impact from the three developments is not the sum of the individual predicted impacts. The cumulative barrier impact is derived from the overall spread of flight directions at each colony potentially affected by the three developments forming barriers.

The predicted effective loss of up to 6.0% of the foraging habitat of the regional razorbill population during the breeding season (Table 4.88) is rated as an effect of moderate magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional razorbill population during the breeding season is of ***moderate significance*** under the terms of the EIA Regulations.

Table 4.88 Summary of CIA for the three proposed offshore wind farms in south-east Scotland on the regional population of breeding razorbill

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF	Predicted impact from Inch Cape pOWF	Predicted impact from Outer Forth pOWF	Cumulative Impact
Displacement: effective loss of foraging habitat during the breeding period (%)	50%	0.7%, Negligible	1.7%, Low	3.6%, Low	6.0%, Moderate
Displacement: effective loss of foraging habitat during the post-breeding period (%)	50%	1.4%, Low	0.9%, Negligible	2.0%, Low	4.3%, Low
Barrier Effect:	100%	Foraging flights of ca. 17% of Isle of May birds potentially increase in length on average by 28%. Low	No flights likely to go beyond the offshore site. Negligible	No flights likely to go beyond the offshore site. Negligible	No flights likely to go beyond the offshore site. Negligible
Collision: % increase in assumed annual adult mortality rate. All designs.	0%	No change, Negligible	No change, Negligible	No change, Negligible	No change, Negligible

The predicted effective loss of up to 4.3% of the foraging habitat of the regional razorbill population during the post-breeding season (Table 4.88) is rated as an effect of low magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional razorbill population during the post-breeding period is of ***minor significance*** under the terms of the EIA Regulations.

Barrier effects as assessed here concern birds which would otherwise fly through the offshore site to access feeding resources beyond it. The mean foraging distance for razorbill is 23.7 km and the mean maximum foraging distance is 48.5 km (Thaxter *et al*. 2012). This means that almost no flights by razorbill will be to areas beyond the barrier formed by the three proposed wind farms. As a result any barrier effect will be negligible. The cumulative barrier effect is rated as negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of a barrier effect caused by the three proposed offshore wind farms on the regional population of razorbills in the colony attendance part of the breeding season is ***not significant*** under the terms of the EIA Regulations.

The amount of razorbill flight activity recorded within the height range of the proposed turbines during baseline surveys was extremely low and therefore the risk of collision strikes is also extremely small. The potential impact of collision mortality on the baseline mortality rate of the regional breeding razorbill population is rated as an effect of negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional breeding razorbill population is ***not significant*** under the terms of the EIA Regulations.

A separate CIA has not been undertaken for the regional razorbill population in the non-breeding-period (i.e., the winter months). The predicted displacement and collision mortality due to the offshore site for razorbill in the non-breeding period is negligible and the sensitivity of the population to displacement and barrier effects at this time of year is also negligible. Therefore it is not plausible that the offshore site could contribute to a significant cumulative impact for this receptor population.

4.4.28.8 Mitigation measures for razorbill

The following mitigation measures are suggested for razorbill:

1 - Minimise vessel disturbance by reduced vessel usage during peak periods of guillemot density;

2 - Ensure vessel movement are, as far as practicable, along recognised shipping corridors to and from the wind farm, and that vessels operate at a suitable speed;

3 - Minimise as far as practicable the overall footprint of the wind farm

4.4.29 Atlantic puffin *Fratercula arctica*

4.4.29.1 Status

Puffins are one of the commonest seabird species in Britain, breeding in coastal colonies. There are several large colonies on the east coast of Scotland, and Seabird 2000 recorded 579,500 breeding pairs in Britain (Mitchell *et al.,* 2004). The closest large colony to the Neart na Gaoithe study area is the Isle of May, with a population of 56,867 pairs in 2009

(SMP, 2012). Lesser sandeel is the commonest prey item for puffins, but they also eat sprat, herring and a wide range of young gadoid fish (Harris 1984).

4.4.29.2 Neart na Gaoithe Study Area

Puffin was the second commonest seabird recorded on surveys in Year 1, with a total of 11,199 birds seen on surveys. The majority of birds (88.3%) were in the buffer area (Table 4.2). Peak numbers of puffins were recorded in the offshore site (574 birds) and the buffer area (5,862 birds) in August. Three unidentified puffin/little auks were also seen in Year 1.

In Year 2, puffin was the third most frequently recorded species on surveys in the Neart na Gaoithe study area, with 6,622 birds seen, just over half of the Year 1 total. The majority of birds (83.2%) were again recorded in the buffer area (Table 4.2). Numbers of puffins in the offshore site peaked at 298 birds in July. Numbers recorded in the buffer area peaked at 1,198 birds in September. Both peaks were lower than the peaks recorded in Year 1.

During the breeding period (April to August), the peak estimated number of puffins occurred in August, with 2,461 birds in the offshore site, and 23,728 birds in the buffer area (Table 4.89) (Figure 4.103). In Year 2, estimated numbers of puffins in the offshore site were highest in July (2,480 birds), with 7,127 birds in the buffer area in May.

In the post-breeding season (September and October), peak estimated numbers of puffins occurred in October, with 2,174 birds in the offshore site, and 11,893 birds in the buffer area in Year 1 (Table 4.89) (Figure 4.103). In Year 2, an estimated 1,776 birds were in the offshore site, with 14,892 birds in the buffer area.

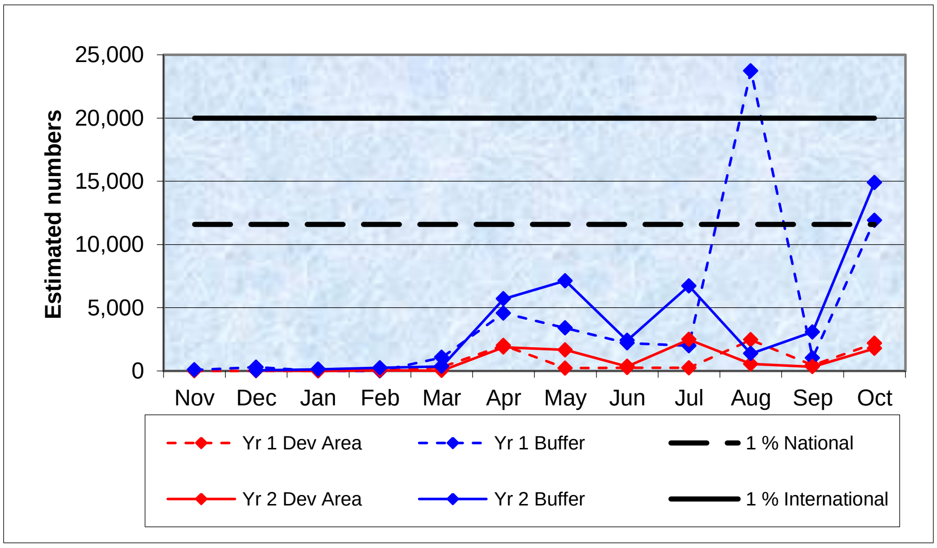


Figure 4.103 Monthly estimated numbers of puffins in the Neart na Gaoithe Development & buffer areas in Years 1 and 2

Estimated numbers of puffins during the non-breeding season (November to March) were lower, with a peak in March in Year 1, when 288 birds were in the offshore site, with 1,062 birds in the buffer area (Table 4.89) (Figure 4.103). In Year 2, peak estimated numbers of puffins in the offshore site occurred in February (54 birds), with 349 birds in the buffer area in March.

Table 4.89 Estimated numbers of puffins in the offshore site and 8 km buffer in Years 1 and 2

Month	Estimated nos on water Offshore Site	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying Offshore Site	Estimated total Offshore Site	Estimated total 8 km buffer [1]	Estimated total
Yr1 Nov	0	0	0	0	0	78	78
Yr1 Dec	0	0	0	0	0	296	296
Yr1 Jan	0	0	0	0	0	0	0
Yr1 Feb	0	0	0	0	0	0	0
Yr1 Mar	288	205	404	0	288	1,062	1,350
Yr1 Apr	1,771	1,110	2,826	245	2,016	4,574	6,590
Yr1 May	200	108	371	20	220	3,392	3,612
Yr1 Jun	215	130	354	41	256	2,208	2,464
Yr1 Jul	204	129	322	41	245	1,989	2,234
Yr1 Aug	2,345	1,427	3,853	116	2,461	23,728	26,189
Yr1 Sep	453	307	668	0	453	1,023	1,476
Yr1 Oct	2,174	1,628	2,904	0	2,174	11,893	14,067
Yr2 Nov	-	-	-	-	-	-	-
Yr2 Dec	32	10	102	0	32	32	64
Yr2 Jan	0	0	0	0	0	128	128
Yr2 Feb	47	36	61	7	54	239	293
Yr2 Mar	31	15	61	0	31	349	380
Yr2 Apr	1,826	1,275	2,616	48	1,874	5,701	7,575
Yr2 May	1,547	1,227	1,950	108	1,655	7,127	8,782
Yr2 Jun	231	128	418	129	360	2,400	2,760
Yr2 Jul	2,278	1,670	3,108	202	2,480	6,717	9,197
Yr2 Aug	494	331	737	55	549	1,368	1,917
Yr2 Sep	323	209	500	0	323	3,082	3,405
Yr2 Oct	1,776	1,445	2,183	0	1,776	14,892	16,668

Estimated mean monthly puffin density in the offshore site and buffer area was very low between November and February (Figure 4.104). Mean density increased from March in both Years 1 and 2, reaching a peak in August of Year 1 of 55.2 birds/km^2 in the offshore site and 90.1 birds/km^2 in the buffer area. In Year 2, peak mean density was lower, with 19.0 birds/km^2 recorded in the offshore site in July, and 21.4 birds/km^2 in the buffer area in October. In comparison, ESAS mean density data for the surrounding ICES rectangles peaked in July (18.6 birds/km^2), with a lower peak of 2.5 birds/km^2 in Regional Sea 1 in June.

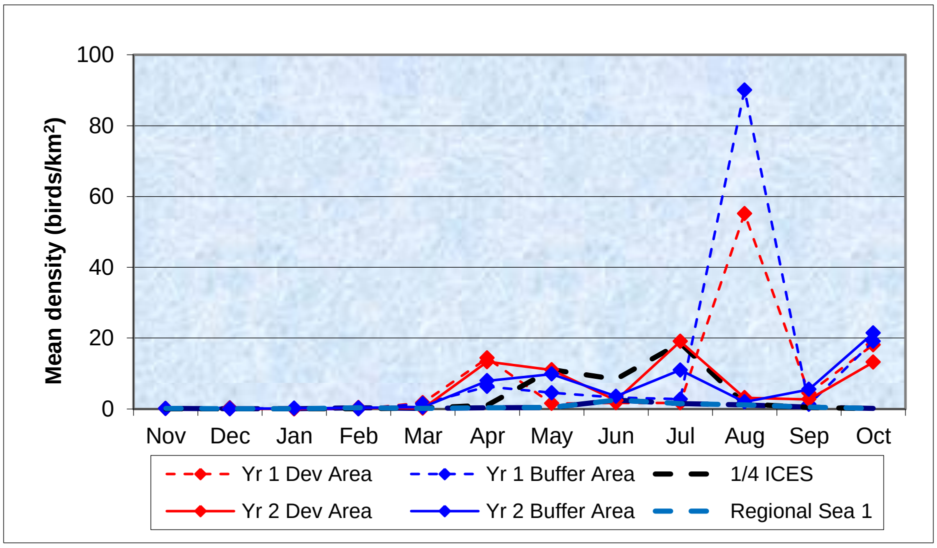


Figure 4.104 Comparison of monthly mean densities for puffin in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

Between November and March of Year 1, puffins were widespread in the eastern side of the study area at mostly low densities, with fewer birds in the south west of the buffer area (Figure 4.105).

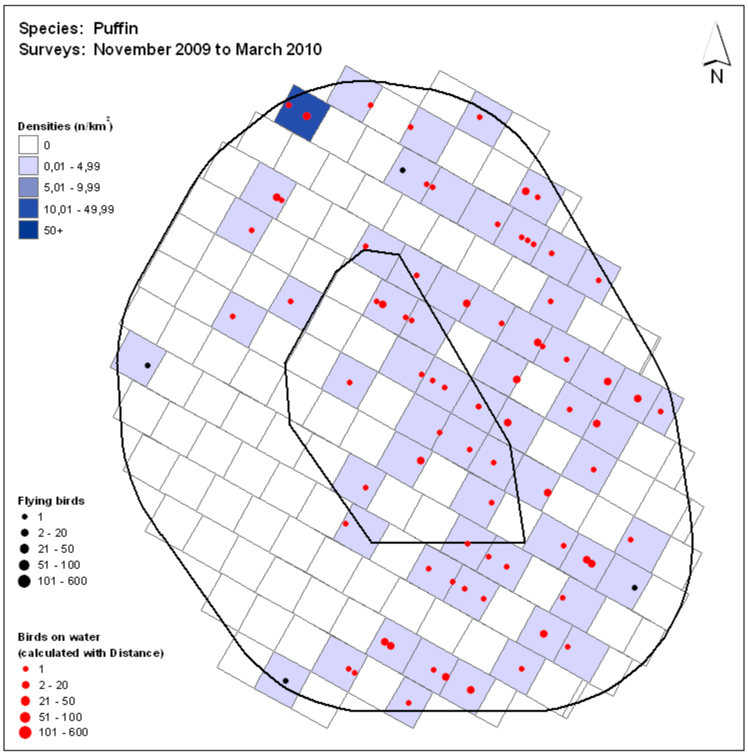


Figure 4.105 Puffin density between November and March, Year 1

Puffin distribution over the same period in Year 2 was very similar, with low densities of birds recorded in the eastern side of the study area, and few birds in the west of the buffer area (Figure 4.106). Puffins were slightly less widespread in the offshore site at this time in Year 2, compared to the same period of Year 1.

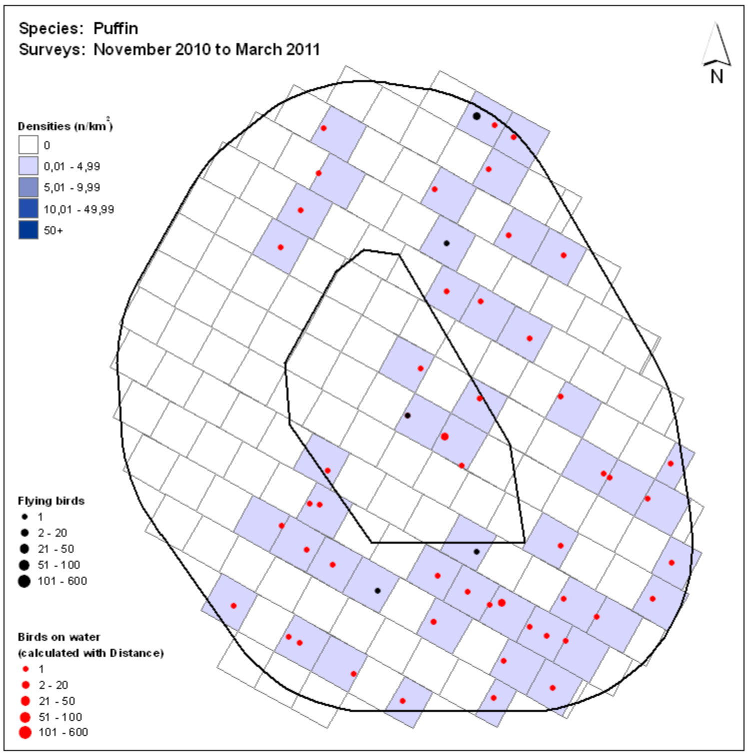


Figure 4.106Puffin density between November and March, Year 2

During the breeding season (April to August) of Year 1, highest densities of puffins were recorded in the southern half of the study area, with lower densities recorded in the north (Figure 4.107).

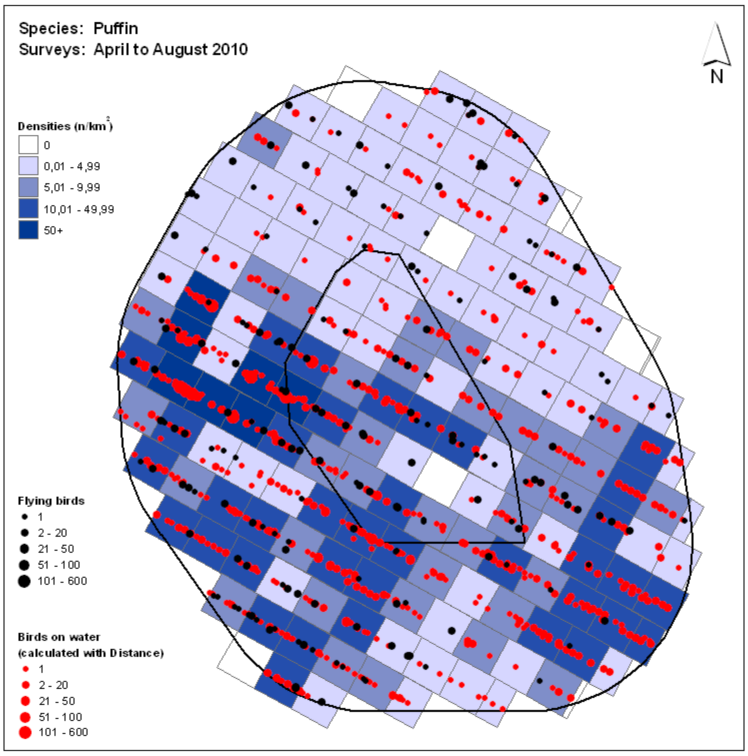


Figure 4.107Puffin density between April and August, Year 1

A broadly similar distribution pattern was recorded in the Year 2 breeding season, with lowest densities of puffins in the north-east of the buffer area, and predominantly high densities elsewhere throughout the study area at this time (Figure 4.108).

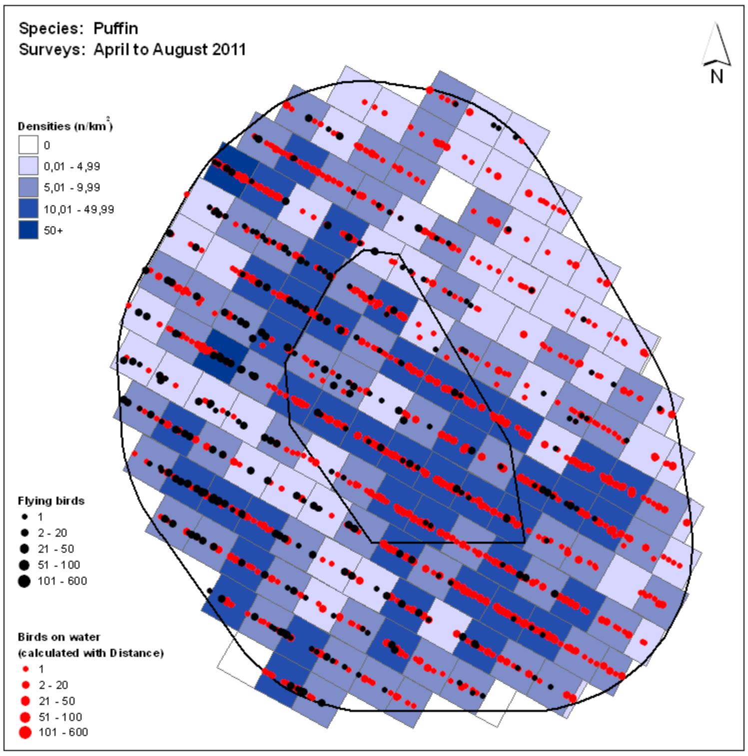


Figure 4.108 Puffin density between April and August, Year 2

In September and October of Year 1, highest densities of puffins were recorded in the eastern half of the offshore site and buffer areas, with lower densities in the western half (Figure 4.109).

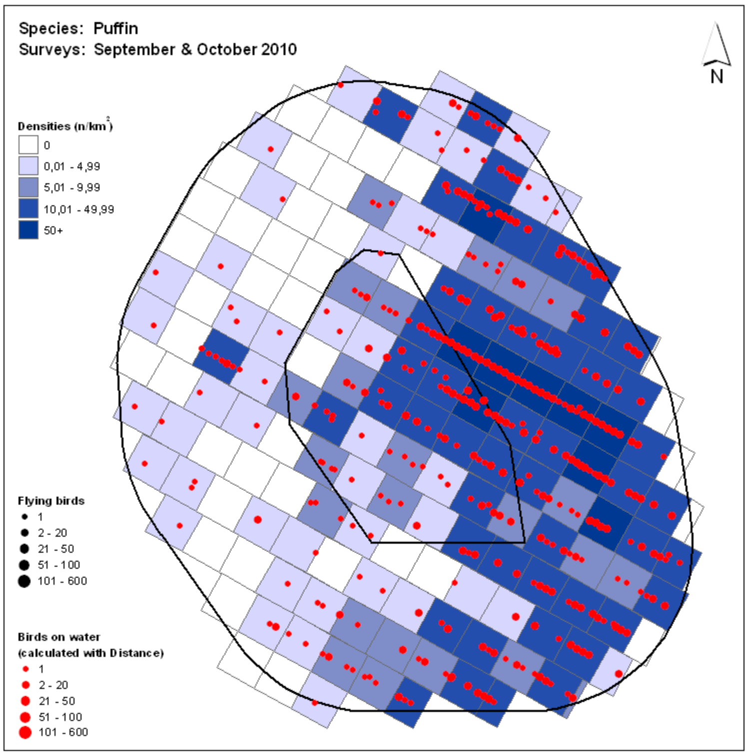


Figure 4.109 Puffin density in September and October, Year 1

In contrast, puffins remained widespread at mostly high densities in September and October of Year 2, with lower densities in the east of the buffer area at this time (Figure 4.110).

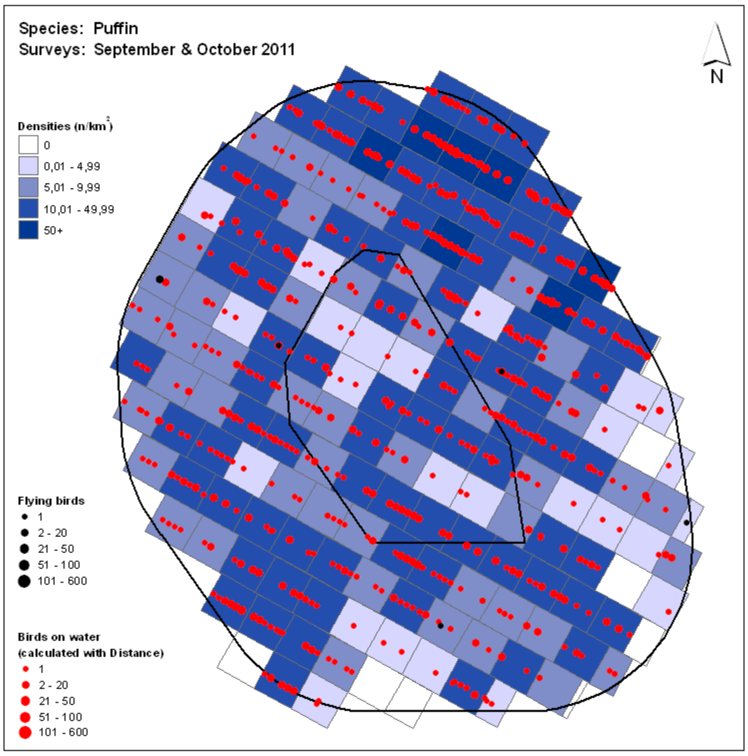


Figure 4.110Puffin density in September and October, Year 2

Foraging behaviour was recorded for 115 puffins in the Neart na Gaoithe study area in Years 1 and 2, with four types of foraging behaviour recorded (Table 4.90). The majority of all foraging birds were recorded holding fish (67.8%). Prey identification was only recorded in four instances (all sandeels), with the remaining sightings being "unidentified fish".

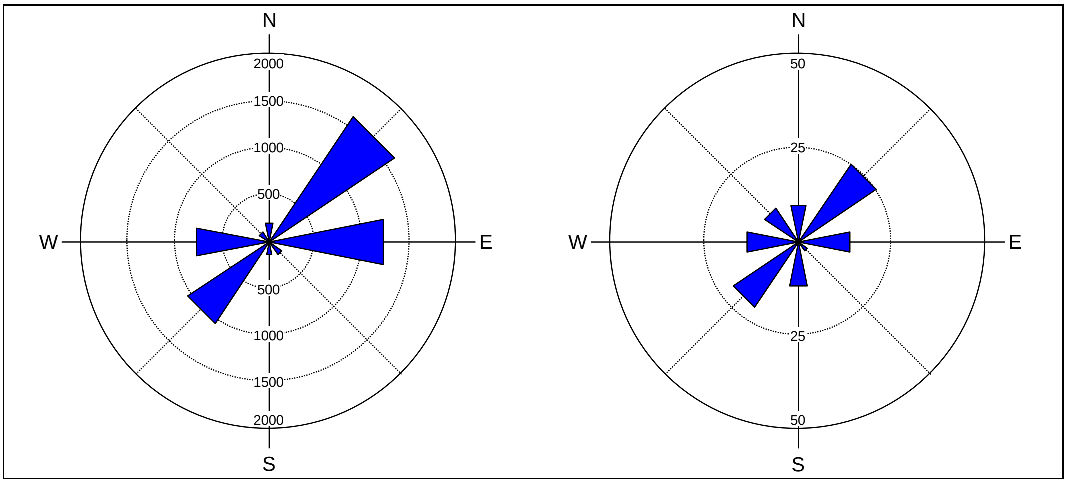
Table 4.90 Puffin foraging behaviour in the Neart na Gaoithe Study Area in Years 1 and 2

Behaviour	Number of birds
Actively searching	13
Pursuit diving	18
Holding fish	73
Pursuit plunging	11
Total	**115**

In Years 1 and 2, 5,779 puffins were recorded in flight, with almost all birds recorded flying below 22.5 m in height, and 98.5% of birds recorded flying below 7.5 m in height (Table 4.3). Two birds (0.03%) were recorded flying above 22.5 m, i.e. within the rotor swept zone, at an estimated height of 25 m.

Flight direction was recorded for 5,305 puffins in the breeding season (April to August), with direction recorded for 110 puffins in the non-breeding season (September to March) (Figure 4.111).

In the breeding season, just over half of all birds recorded were flying north-east (30.3%) or east (23.4%), with just over one third of birds flying south-west (19.5%), and west (14.8%). In the non-breeding season, one quarter of all birds recorded were flying north-east (25.5%), with just less than a fifth of birds flying south-west (19.1%).



April to August (n=5,305 birds) September to March (n=110 birds)
Numbers shown on figures are number of birds recorded

Figure 4.111 Flight direction of puffins in the Neart na Gaoithe Study Area in Years 1 and 2

4.4.29.3 Species sensitivity

Puffin is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009).

A recent review assessed puffin as being at moderate risk of displacement, barrier effects, habitat loss or changes in prey distribution resulting from offshore wind farms, while collision risk was rated as low risk. Overall, puffin was assessed as being at moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

Puffin is listed as a qualifying interest species in the breeding season for three SPAs on the UK east coast between Peterhead and Blyth that could potentially be affected by the Neart na Gaoithe development (Table 4.91). These SPAs held 14.9% of the UK breeding population and 7.5% of the biogeographic population at the time of designation (JNCC, 2012). The distance between the offshore site and two SPAs (Farne Islands SPA and Forth Islands SPA) is within the mean maximum foraging range of puffin (105.4 km). The distance to Coquet Island SPA is within the maximum known foraging range of 200 km (Thaxter *et al.*, 2012). Puffin mean maximum foraging range from breeding SPAs in relation to the offshore site are shown in Figure 4.112.

Table 4.91 SPAs for puffin in the breeding season between Peterhead and Blyth

SPA site	Distance to site (km)	Site total (pairs)[1]	% of biogeo-graphic popn[1]	% of national popn[1]	Recent count[2]	Year
Coquet Island	*106*	*11,400*	*1.3*	*2.5*	*15,812*	*2009*
Farne Islands	**72**	**34,710**	**3.9**	**7.7**	**36,835**	**2008**
Forth Islands	**16**	**21,000**	**2.3**	**4.7**	**62,167**	**2009**
Total	**-**	**67,110**	**7.5**	**14.9**	**114,814**	**-**

Sources: 1 JNCC (2012) – SPA online species accounts. 2 SMP (2012) – Seabird Monitoring Programme Online Database. Sites in italics lie within the maximum known foraging range of 200 km. Sites in bold lie within the mean maximum foraging range of 105.4 km (Thaxter *et al.*, 2012).

A recent JNCC statistical analysis of ESAS data investigating possible marine SPAs, identified waters around the Neart na Gaoithe study area as an important area for puffins during the breeding season (Kober *et al.,* 2010).

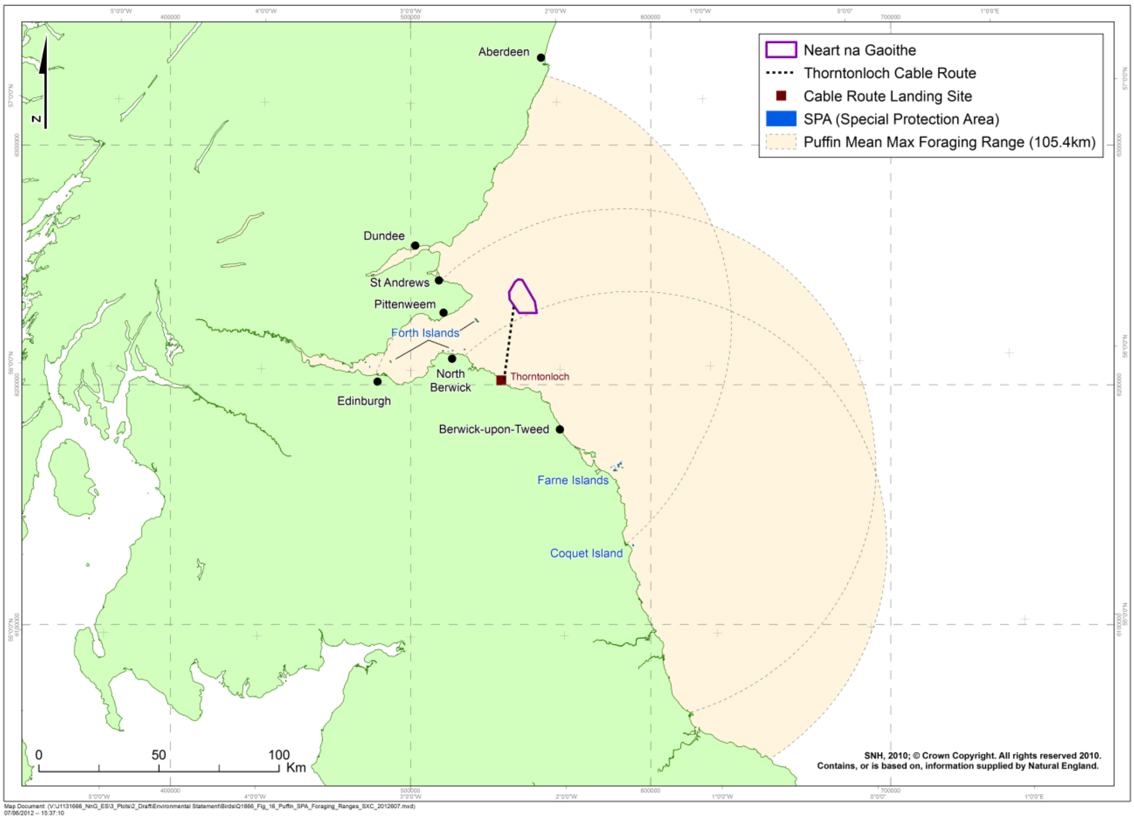


Figure 4.112 Puffin mean maximum foraging range from breeding SPAs in relation to the Development

4.4.29.4 Assessment

Definition of seasons

The annual cycle for puffin was divided into three periods to reflect the biology of the species and the broad pattern of use of the offshore site.

The breeding season, when breeding adults are attending colonies, is defined as April to August. At this time the vast majority of birds present in the offshore site will be from relatively local breeding colonies (i.e. those for which the offshore site is within foraging range).

The post-breeding period defined as September and October. Although birds at this time of year are no longer breeding they were examined separately to the later part of the non-breeding period because the numbers present were much greater and it is likely that the vast majority of individuals present at this time are from breeding sites within the region.

The non-breeding period (the 'winter' period) is defined as November to March and broadly corresponds to the period when puffins are in their over-wintering area. In this period it is likely that a high proportion of individuals present in the offshore site are from breeding colonies outwith the region, including birds from other countries (Wernham *et al.,* 2002).

Baseline conditions

In the breeding season (April to August) the mean estimated number of puffins present in the offshore site was 1,305 birds, and the mean estimated number in the offshore site buffered to 1 km was 1,877 birds. During this period 90.5% of the birds present were on the sea and the rest were in flight. For the purposes of assessment it is assumed that all birds in summer plumage were breeding birds. Studies have estimated that mean colony attendance rate of puffins during the breeding season is 35% (Creelman & Storey 1991) and this figure is used to calculate the importance of the offshore site for foraging birds in the breeding season.

In the post-breeding period (September and October) the mean estimated number of adult puffins present in the offshore site was 857 birds, and the mean estimated number in the offshore site buffered to 1 km was 1,348 birds. During this period 99.8% of the birds present were on the sea and the remainder were in flight.

In the non-breeding period (November to March) the mean estimated number of puffins present in the offshore site was 25 birds, and the mean estimated number in the offshore site buffered to 1 km was 33 birds. During this period 95.2% of the birds present were on the sea and the remainder were in flight.

Populations

Puffin is the second commonest seabird species breeding in the region. The breeding population of puffins in south-east Scotland and north-east England has undergone recent declines (SMP, 2012). Assuming that the sub-set of colonies that have been recently counted is representative, the regional decline since the Seabird 2000 counts amounts to -12%. The decline in breeding numbers means that published figures on population size for this species (e.g., Mitchell *et al*., 2004) no longer accurately reflect the current population size in eastern Scotland. To prevent this causing assessment of effects to be biased low, the regional total derived from Seabird 2000 results has been adjusted downwards by 12%. On this basis, the regional breeding population is assumed to be 258,543 breeding adults.

There is uncertainty over the size of the regional puffin population in the post-breeding-period but for the purposes of assessment it is assumed to be 167,790 birds. The reasoning behind this estimate is explained as follows. The estimate given by Skov *et al.* (1995) for birds at sea in the whole of eastern Scotland (including Orkney and Shetland) and north-east England during August to September is 179,100 birds, approximately one third of the number of breeding birds in this area (636,864 adult birds - Mitchel *et al*., 2004). This suggests that there are considerably fewer birds present in east Scotland and north-east England during the post-breeding period than in the breeding season. However, the estimates by Skov *et al.* (1995) for April to July (i.e. the colony attendance period) for eastern Scotland and north-east England are only 155,000 birds, which, assuming a 35% colony attendance rate in this period, translates into a breeding population size of approximately 238,460 adults, far lower than indicated by colony counts (1985-88 SCR Census estimated approximately 433,000 birds and Seabird 2000 estimates approximately 637,000 birds, Mitchel *et al*., 2004). This suggests that the Skov *et al*. (1995) estimates are significantly biased low despite correction factors being used to account for under-recording. Nevertheless, the difference (-35%) between the Skov *et al.* derived estimates for April to July (238,460 birds) and for August to September (179,100 birds) is likely to broadly reflect the actual change in the numbers present between the two periods. On this basis, the regional post breeding population is assumed to be 35% less than the regional breeding population, which gives an estimate of 167,790 birds.

The size of the regional puffin population in the non-breeding-period is assumed to be 37,500 birds. This is the mean of the October to January (25,000) and February to March (50,000) estimates given in Skov *et al*. (1995).

Nature conservation importance

For EIA assessment purposes, the nature conservation importance of puffin using the offshore site is rated as High during the breeding season and post-breeding period. The species is merits High NCI because a high proportion of birds using the offshore site are likely to be from the breeding colonies within the Forth Islands SPA, where this species is a qualifying interest. The mean number present in the offshore site buffered to 1 km in the breeding season is approximately 1% of the (at-sea) regional population. Puffin is not subject to any special legislative protection, nor is it on any conservation priority list.

Offshore wind farm studies of puffin

There is little field-based evidence on the effects on puffins from operational wind farms. This is because existing offshore wind farms for which published results are available are located in areas where puffins are naturally scarce. Occasionally puffins were recorded during Horns Rev, Egmond aan Zee and Arklow Bank post-construction monitoring but not in sufficient numbers to undertake any statistical analysis of effects (Petersen, 2005, Leopold *et al*., 2011 Barton *et al,* 2010).

The extent to which wind farms are likely to act as a barrier to puffins is unknown. However, a recent study looking at the theoretical energy costs of a barrier effect concludes, “If an Atlantic puffin were to travel an additional 10,000 m due to the presence of wind farms then it would expend 103% of its daily energy expenditure on the extended flight activity alone" (Masden *et al*., 2010).

The review of offshore wind farm effects on birds categorises displacement, barrier and collision risk effects all as unknown for puffin (Diersche and Garthe, 2006). No puffin fatalities are reported in a review of the number of collision victims at wind farms in eight European countries (Hötker *et al*., 2006) although the very low probability of detecting seabird fatalities should be recognised together with the natural scarcity of this species in the areas studied.

EIA Construction Phase assessment for puffin

The construction phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

The extent that birds may be displaced by vessels varies depending on the type and speed of the vessel and possibly the time of year. Puffins are not thought to be particularly sensitive to vessel movements but birds disturbed by vessels will either swim or fly away to an alternative location until the vessels have passed, after which they may return to the area. Consequently, there may be a localised, short-term effect.

During the construction phase there is the potential for the prey species, e.g. sandeels, of puffin to be displaced, particularly during piling activities. Should this occur then it is predicted that the puffins will also relocate as they follow the movements of their prey.

Noise modelling undertaken indicates that behavioural impacts on sandeels from piling noise is predicted to extend less than 1.5 km from the piling activities (See Chapter 15). Therefore, the effect on puffins foraging on sandeels is likely to be relatively localised.

Based on this, possible displacement impacts were categorised as being of low (1-5%) magnitude, and being temporally short-term and reversible (Table 3.1 and Table 3.2). Overall, it is concluded that the impacts of displacement during construction operations on the regional populations of puffins in the breeding, post-breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

EIA Operational Phase assessment for puffin

Potential for puffin to be affected by displacement

For the purposes of assessment it is assumed that 50% of puffins would be displaced from the proposed development and a surrounding buffer of 1 km.

The value of the offshore site buffered to 1 km as a foraging site for puffins was estimated for the regional population for the three periods of the year from the proportion of the adults likely to be at sea (not attending a colony) that were on average present (Table 4.92). This is likely to overestimate the importance for feeding as it is likely that some individuals present were not feeding.

Table 4.92 The mean number of puffin present in the offshore site and 1 km buffer in each period of the year expressed as the percentage of the (at-sea) receptor population

Receptor population	Population size (adults)	No. assumed at sea	Dev. area		Dev. area + 1km	
			Mean no. at risk	%	Mean no. at risk	%
National, colony-attend period (Seabird 2000)	1,159,447	753,640	1,305	0.2	1,877	0.2
North Sea, post-breeding period (Skov *et al*., 1995)	1,159,447	1,159,447	857	<0.1	1,348	0.1
North Sea, non-breeding period (Skov *et al*., 1995)	50,000	50,000	25	<0.1	33	<0.1
Regional, colony-attend period (Seabird 2000 x 12% decline)	258,543	168,053	1,305	0.8	1,877	1.1
Regional, post-breeding period (Skov *et al*., 1995)	167,790	167,790	857	0.5	1,348	0.8
Regional, non-breeding period (Skov *et al*., 1995)	37,500	37,500	25	<0.1	33	<0.1

On this basis, it is inferred that the proportion of the foraging resources provided by the offshore site buffered to 1 km during the different periods of the year was as follows (Table 4.92):

- 1.1% during the colony-attendance part of the breeding season;
- 0.8% during the post-breeding period;
- <0.1% during the non-breeding (winter) period.

Likely impacts of displacement on puffin populations

For the purposes of assessment it is cautiously assumed that 50% of puffins that would otherwise forage in the offshore site buffered to 1 km would be displaced elsewhere.

Colony-attendance period

If 50% of puffins were to be displaced from the offshore site buffered to 1 km the impact of this would be the effective loss of around 0.6% of the foraging habitat of the regional population during the colony-attendance period (50% of value in Table 4.92). This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and

Table 3.2). The sensitivity of the population to displacement at this time of year is unknown, but is likely to be moderate because birds are attending breeding colonies, which constrains their foraging, and they have additional energy demands associated with breeding. It is concluded that the impact of displacement on the regional puffin population in the breeding season is ***not significant*** under the EIA Regulations.

Post-breeding period

If 50% of puffins were to be displaced during the post-breeding period from the offshore site buffered to 1 km the impact of this would be the effective loss of around 0.40% of the foraging habitat of the post-breeding-period population (50% of value in Table 4.92). This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). The sensitivity of the population to displacement at this time of year is unknown, but is likely to be low, as in this period puffins are not constrained by attending their colony. It is concluded that the effects of displacement on the regional puffin population in the post-breeding period are ***not significant*** under the EIA Regulations.

Non-breeding period

If 50% of puffins were to be displaced during the non-breeding period from the offshore site buffered to 1 km the impact of this would be the effective loss of <0.1% of the foraging habitat of the non-breeding-period population (50% of value in Table 4.92). This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). The sensitivity of the population to displacement at this time of year is unknown, but is likely to be negligible. It is concluded that the effects of displacement on the regional puffin population in the non-breeding period are ***not significant*** under the EIA Regulations.

Barrier Effect

Puffins are considered by Langston (2010) to have moderate sensitivity to the effects of barriers formed by offshore wind farms.

The potential effects on puffins of the proposed wind farm acting as a barrier were assessed only for the part of the breeding season when birds are attending colonies. During this period birds undertake commuting flights to and from feeding grounds and it is the potential for the wind farm to act as a barrier and disrupt these flights that gives cause for concern and the possibility of adverse effects on a population.

The proposed wind farm would potentially form a barrier to commuting birds from all breeding colonies that are closer to the offshore site than the typical foraging distance of puffin during the period of colony attendance. The only large puffin colonies potentially affected are the Isle of May and Craigleith. Although in theory birds from more distant colonies could also be affected, the alignment of the proposed wind farm and the distance from these colonies make it implausible that barrier effects on birds from these colonies could have more than a negligible effect. Therefore, no attempt is made to quantify it.

For the purposes of assessment the width of the barrier is assumed to extend 1 km either side of the maximum width of the proposed wind farm. The estimated magnitude of the barrier effect to birds from the plausibly affected colonies is summarised in Table 3.6 and

Table 3.7. Barrier effects as calculated here concern birds which would otherwise fly through the offshore site to access feeding resources beyond it.

For the Isle of May colony, the proposed wind farm would present a barrier 17.9 km wide, 16.2 km to the north-east. This barrier would potentially affect approximately 33% of the possible flight directions available to puffins flying out to distances in excess of 16.2 km from the Isle of May (Table 3.6). Tagging studies of puffins on the Isle of May in 2010 showed that the maximum distance from the colony exceeded 16.2 km for 93% of the 15 foraging trips logged, and that the mean maximum distance of the trips that exceeded 16.2 km was at least 43.1 km from the colony (CEH, 2012). Assuming that the average destination of barrier-affected flights lies on average 45 km from the colony, the mean increase in the length of affected flights is estimated at 8.1% (approximately 3.7 km) (Table 3.7). The back edge of the barrier formed by the offshore site is approximately 26 km from the Isle of May. Therefore it is likely that the majority of puffin flights from this colony in the direction of the offshore site are to intended destinations beyond it.

For Craigleith, the proposed wind farm would present a barrier 17.8 km wide, 31.5 km to the north-east. This barrier would potentially affect approximately 28% of the possible flight directions available to puffins flying out to distances in excess of 31.5 km from the Craigleith colony (Table 3.6). Tagging studies of puffins on the Isle of May in 2010 showed that the maximum distance from the colony exceeded 31.5 km for 80% of the 15 foraging trips logged, and that the mean maximum distance of the trips that exceeded 31.5 km was at least 47.4 km from the colony (Daunt pers. comm.). Assuming that the average destination of barrier-affected flights from Craigleith lies on average 50 km from the colony, the mean increase in the length of affected flights is estimated at 6.4%, (approximately 3.2 km) (Table 3.7). The back edge of the barrier formed by the offshore site is approximately 41 km from Craigleith. Therefore it is likely that the majority of puffin flights from this colony in the direction of the offshore site are to intended destinations beyond it.

The Isle of May and Craigleith together hold approximately 48% of the regional breeding puffin population. On the basis of the figures presented above it is estimated that the foraging trips of approximately 16% of birds from the regional population would be potentially affected by the wind farm acting as a barrier and on average it would cause affected flights to increase in length and duration by up to ca. 8% (equivalent to <4 km) compared to direct flights to the same destination. Studies of other auk species at offshore wind farms indicate that some individuals are likely to pass through the barrier and therefore the full potential magnitude of a barrier effect on puffins may not be realised. Studies on the theoretical energetic costs to seabirds caused wind farm barriers show that puffins have a relatively high sensitivity to increases in foraging trip length (Masden *et al.* 2010). Nevertheless, puffin are likely to be able to accommodate increases in trip length of <4 km without experiencing an adverse impact.

The likely impact of the proposed wind farm acting as a barrier to breeding puffins is categorised as low in magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the predicted impact of the proposed wind farm acting as a barrier on the regional puffin population in the breeding season is an effect of ***minor significance*** under the terms of the EIA Regulations.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

The presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The extent that birds may be displaced by vessels varies depending on the type and speed of the vessel and possibly the time of year. Puffins are not thought to be particularly sensitive to vessel movements but birds disturbed by vessels will either swim or fly away to an alternative location until the vessels have passed, after which they may return to the area. Consequently, there may be a localised, short-term effect.

Based on this, possible displacement impacts were categorised as being of low (1-5%) magnitude, and being temporally short-term and reversible (Table 3.1 and Table 3.2). Overall, it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional populations of puffins in the breeding, post-breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

Collision Mortality

Collision risk modelling using a 98.0% avoidance rate predicts that no puffins would be killed by the proposed wind farm (Appendix 12.2). Therefore, it is not plausible that this species will experience mortality from collision with turbine rotors. The reason why no collisions are predicted is because all but two puffins out of 5,779 birds recorded in flight during baseline surveys (99.96%) were below the lowest height of the proposed rotor sweep of turbines.

The likely effect of the collision mortality on the baseline mortality rate on the regional puffin population is rated as negligible in magnitude temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of collision mortality on puffins is ***not significant*** under the terms of the EIA Regulations.

EIA Decommissioning Phase assessment for puffin

The decommissioning phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

The extent that birds may be displaced by vessels varies depending on the type and speed of the vessel and possibly the time of year. Puffins are not thought to be particularly sensitive to vessel movements but birds disturbed by vessels will either swim or fly away to an alternative location until the vessels have passed, after which they may return to the area. Consequently, there may be a localised, short-term effect.

During the decommissioning phase there is the potential for the prey species, e.g. sandeels, of puffins to be displaced. Should this occur then it is predicted that the puffins will also relocate as they follow the movements of their prey.

Based on this, possible displacement impacts were categorised as being of low (1-5%) magnitude, and being temporally short-term and reversible (Table 3.1 and Table 3.2). Overall, it is concluded that the impacts of displacement during decommissioning operations on the regional populations of puffins in the breeding, post-breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

EIA summary of effects combined for puffin

Table 4.93 Summary of effects on the regional population of puffins during the breeding and post-breeding periods

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Moderate	***Not significant***
Operational phase				
Displacement from foraging habitat, colony-attendance period	Negligible	Long term	Low	***Not significant***
Displacement from foraging habitat, post-breeding period	Negligible	Long term	Low	***Not significant***
Barrier Effect	Low	Long term	Moderate	***Minor significant***
Vessel disturbance	Negligible	Long term	Moderate	***Not significant***
Collision mortality	Negligible	Long term	Low	***Not significant***
All effects combined	Low	Long term	Low - Moderate	***Minor significant***

The adverse impacts of the effects assessed will act in a broadly additive manner. In combination it is judged that the magnitude of the effects on the regional puffin population during the breeding and post-breeding period is low. It is concluded that the overall impact on the regional population of puffins in the breeding and post-breeding periods is ***minor significant*** under the EIA regulations (Table 4.93).

The adverse impacts of the effects assessed will act in a broadly additive manner. In combination it is judged that the magnitude of the effects on the regional puffin population during the non-breeding period is low. It is concluded that the overall impact on the regional population of puffins in the non-breeding period is ***not significant*** under the EIA regulations (Table 4.94).

Table 4.94 Summary of effects on the regional population of puffins in the non-breeding period

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Moderate	**Not significant**
Operational phase				
Displacement from foraging habitat, post-breeding period	Negligible	Long term	Moderate	**Not Significant**
Displacement from foraging habitat, winter period	Negligible	Long term	Negligible	**Not significant**
Vessel disturbance	Negligible	Long term	Moderate	**Not significant**
Collision mortality	Negligible	Long term	Low	**Not significant**
All effects combined	Negligible	Long term	Low - Moderate	**Not significant**

4.4.29.5 Cumulative Impact Assessment for puffin

Combining the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts to the regional population of puffins in the breeding and post-breeding periods will be as shown in Table 4.95. The individual impacts for displacement and collision are combined by addition to give the cumulative impact. The flight directions from the colonies that are potentially affected by each wind farm acting as a barrier substantially overlap and therefore the cumulative barrier impact from the three developments is not the sum of the individual predicted impacts. The cumulative barrier impact is derived from the overall spread of flight directions at each colony potentially affected by the three developments forming barriers.

The predicted effective loss of up to 2.4% of the foraging habitat of the regional puffin population during the breeding season (Table 4.95) is rated as an effect of low magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional puffin population during the breeding season is of ***minor significance*** under the terms of the EIA Regulations.

Table 4.95 Summary of CIA for the three proposed offshore wind farms in south-east Scotland on the regional population of puffins in the breeding and post-breeding periods

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF	Predicted impact from Inch Cape pOWF	Predicted impact from Outer Forth pOWF	Cumulative Impact
Displacement: effective loss of foraging habitat during breeding period (%)	50%	0.6%, Negligible	0.5%, Negligible	1.3%, Negligible	2.4%, Low
Displacement: effective loss of foraging habitat during post-breeding period (%)	50%	0.4%, Negligible	0.8%, Negligible	1.7% Low	2.9% Low
Barrier Effect:	100%	Up to ca.16% of foraging flights in the region (all by Forth Islands birds) increase in length by up to 8%. Low	Up to ca.10% of foraging flights in the region (all by Forth Islands birds) increase in length by up to 6%. Negligible	Negligible barrier effect as very few flights are likely to go beyond the offshore site. Negligible	Negligible barrier effect as very few flights are likely to go beyond the offshore site. Negligible
Collision: % increase in assumed annual adult mortality rate, all designs	0%	No change, Negligible	No change, Negligible	No change, Negligible	No change, Negligible

The predicted effective loss of up to 2.9% of the foraging habitat of the regional puffin population during the post-breeding season (Table 4.95) is rated as an effect of low magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of displacement caused by the three proposed offshore wind farms on the regional puffin population during the post-breeding period is of ***minor significance*** under the terms of the EIA Regulations.

Barrier effects as assessed here concern birds which would otherwise fly through the offshore site to access feeding resources beyond it. Tagging studies of 15 breeding puffins on the Isle of May in 2010 showed that the maximum foraging distance from the colony recorded was 66 km (CEH, 2012). The back edge of the barrier formed by the three wind farms would be approximately 75 km from the Isle of May and approximately 90 km from Craigleith. This means that almost no foraging flights by puffins would be expected to be to areas beyond the barrier formed by the three proposed wind farms. As a result any barrier effect will be negligible. The cumulative barrier effect is rated as negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of a barrier effect caused by the three proposed offshore wind farms on the regional population of puffins in the breeding season is ***not significant*** under the terms of the EIA Regulations.

The amount of puffin flight activity recorded within the height range of the proposed turbines during baseline surveys was extremely low and therefore the risk of collision is also extremely small. The potential impact of collision mortality on the baseline mortality rate of the regional puffin population in the breeding and post-breeding periods is rated as an effect of negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional puffin population in the breeding and post-breeding periods is ***not significant*** under the terms of the EIA Regulations.

A separate CIA has not been undertaken for the regional puffin population in the non-breeding-period (i.e. the winter months). The displacement and collision mortality predicted for puffin due to the offshore site in the non-breeding period is negligible and the sensitivity of the population to displacement and barrier effects at this time of year is also negligible. Therefore it is not plausible that the offshore site could contribute to a significant cumulative impact for this receptor population.

10.3.1.8 Mitigation measures for puffin

The assessment does not identify any significant adverse effects on the populations of puffins breeding or overwintering in the region. Therefore, no mitigation measures are required for this species.

4.4.30 Little auk *Alle alle*

4.4.30.1 Status

Little auks breed in the high Arctic in large numbers, and occur in UK waters in late autumn and winter months. Large “wrecks” of birds can occur following winter gales, with birds sometimes found inland on lakes or reservoirs. Little auks mainly feed on *Calanus* *finmarchius*, a planktonic copepod, which is also a major food for sandeels and other fish (Forrester *et al*., 2007).

4.4.30.2 Neart na Gaoithe Study Area

Little auks were only recorded in the Neart na Gaoithe study area in the winter months. Between November and February of Year 1, 135 little auks were recorded in the Neart na Gaoithe study area, with the majority of birds (80.7%) in the buffer area (Table 4.2). Numbers recorded on surveys between December and February of Year 2 were similar (113 birds), with 85.8% of birds seen in the buffer area.

In the non-breeding season (November to February), peak estimated numbers of little auks occurred in November of Year 1, with 536 birds in the offshore site, and 1,222 birds in the buffer area (Table 4.96) (Figure 4.113). Estimated numbers in the buffer area in February of Year 1 were similar (1,191 birds). In Year 2, peak estimated numbers of little auks occurred in December, with 95 birds in the offshore site, and 529 birds in the buffer area.

Table 4.96 Estimated numbers of little auks in the offshore site and 8 km buffer between November and February of Years 1 and 2

Month	Estimated nos on water Offshore Site	Lower 95% C.L.	Upper 95% C.L.	Estimated nos flying Offshore Site	Estimated total Offshore Site	Estimated total 8 km buffer [1]	Estimated total
Yr1 Nov	522	323	843	14	536	1,222	1,758
Yr1 Dec	0	0	0	0	0	145	145
Yr1 Jan	0	0	0	0	0	0	0
Yr1 Feb	159	84	301	0	159	1,191	1,350
Yr2 Nov	-	-	-	-	-	-	-
Yr2 Dec	95	47	192	0	95	529	624
Yr2 Jan	47	22	99	0	47	472	519
Yr2 Feb	0	0	0	0	0	0	0

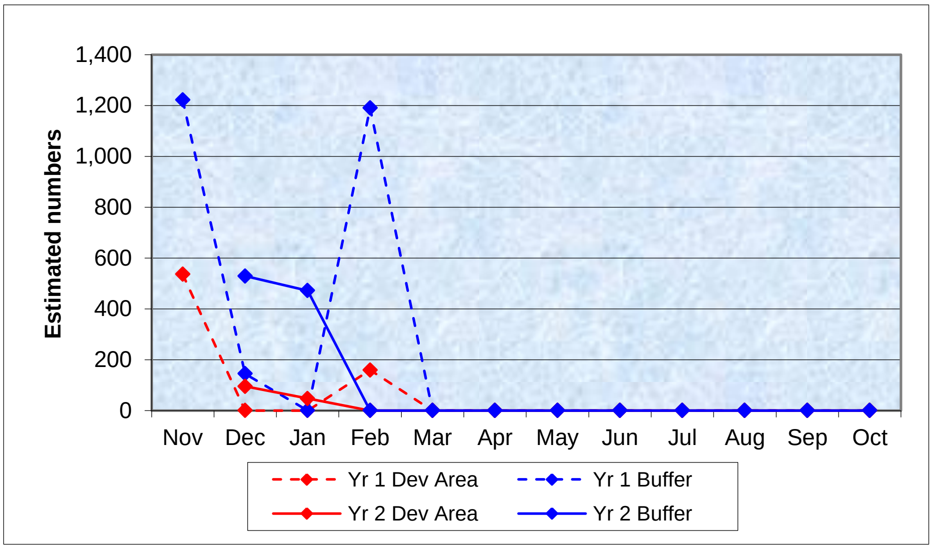


Figure 4.113 Monthly estimated numbers of little auks in the Neart na Gaoithe Development & buffer areas in Years 1 and 2

Mean monthly little auk density was similar between the offshore site and the buffer area in Year 1 (Figure 4.114). Peak mean density was recorded in the offshore site in November of Year 1 (2.2 birds/km^2), however there was no survey in November of Year 2 for comparison. Mean density In the buffer area was similar between Years 1 and 2, with a peak of 1.1 birds/km^2 in November of Year 1 and a peak of 0.7 birds/km^2 in December of Year 2. ESAS mean density data from the surrounding ICES rectangles and Regional Sea 1 was lower between November and March.

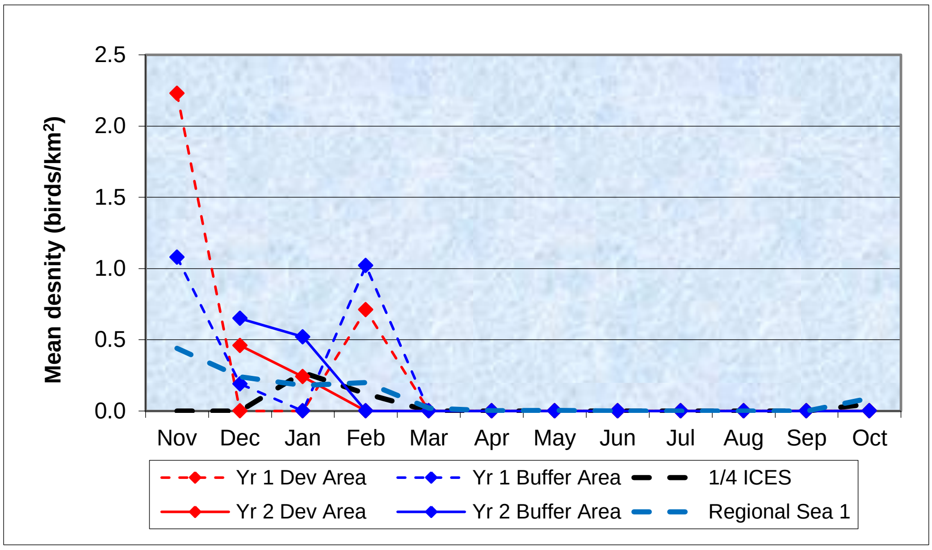


Figure 4.114 Comparison of monthly mean densities for little auk in the Neart na Gaoithe Development & buffer areas in Years 1 and 2, with ESAS data from surrounding ICES rectangles and Regional Sea 1

Between November and February of Year 1, little auks were widespread in the eastern side of the offshore site and buffer areas at low to moderate, occasionally high densities, with fewer birds in the rest of the study area (Figure 4.115).

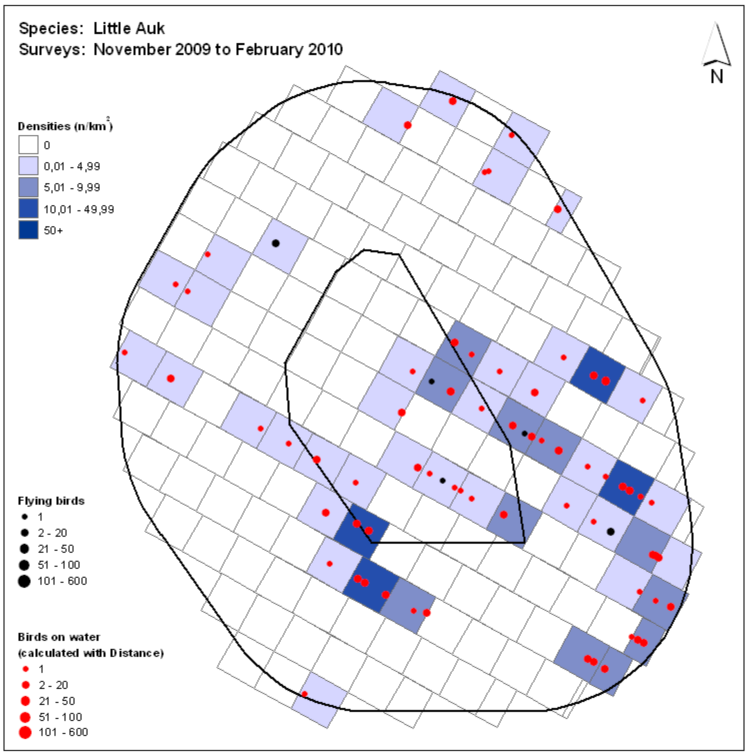


Figure 4.115Little auk density between November and February, Year 1

Little auks were more scattered across the study area over the same period in Year 2, at low to moderate densities (Figure 4.116). Birds were slightly less widespread in the offshore site at this time in Year 2, compared to the same period of Year 1.

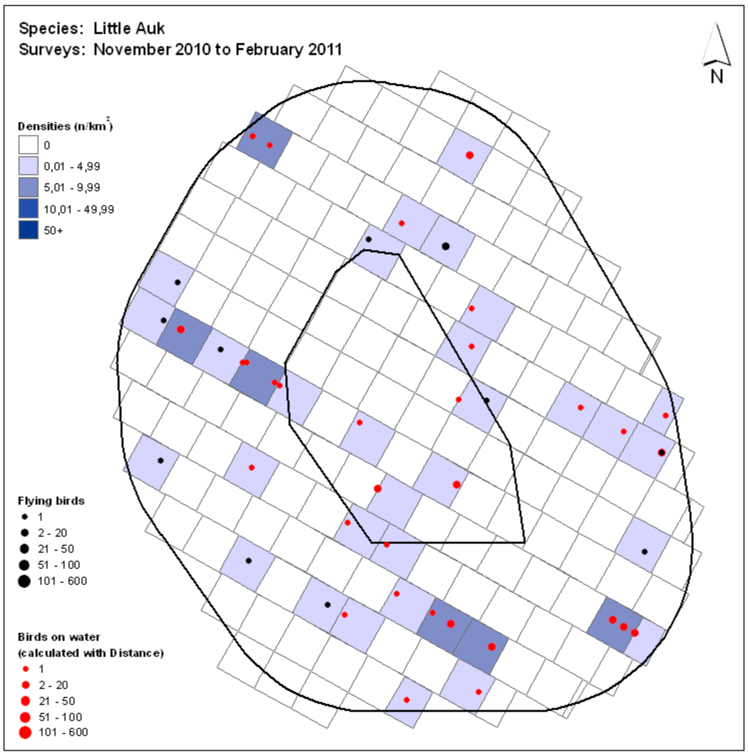


Figure 4.116Little Auk density between November and February, Year 2

In Years 1 and 2, 69 little auks were recorded in flight, with all birds below 22.5 m in height (Table 4.3).

4.4.30.3 Species sensitivity

A recent review assessed little auk as being at moderate risk of displacement, barrier effects, habitat loss or changes in prey distribution resulting from offshore wind farms, while collision risk was rated as low risk. Overall, little auk was assessed as being at possibly moderate risk from offshore wind developments (Langston 2010) (Table 4.4).

4.4.30.4 Assessment

Definition of seasons

Little auks were only recorded in the offshore site between November and February and therefore this was the only period considered.

Baseline conditions

In the non-breeding period (November to February) the mean estimated number of little auks present in the offshore site was 54 birds, and the mean estimated number in the offshore site buffered to 1 km was 88 birds. Of the birds present, 93.6% were on the sea and the remainder were in flight. The numbers recorded were similar in both baseline survey years.

Populations

Little auks are a winter visitor to the seas around the UK from Arctic breeding grounds. The origin of the birds wintering off eastern Scotland is unknown but is likely to be breeding grounds in Iceland, Norway and Russia (Wernham *et al.,* 2002).

Analysis of ESAS data by Skov *et al*. (1995) identifies a relatively discrete wintering concentration in the outer Firth of Forth/Devil's Hole part of the North Sea, estimated at approximately 2,300 birds. However, this figure is not considered to be a reliable estimate of the current regional wintering population because larger numbers have been estimated from recent surveys in the region undertaken to inform the proposed wind farm projects. There are likely to be large year-to-year fluctuations in numbers present.

The peak winter counts from three proposed wind farm survey areas (Neart na Gaoithe, 1,758 birds; Inch Cape, 126 birds in Year 1; and Outer Forth, approximately 2,358 birds in Year 1) suggests a minimum regional population of 4,242 birds. However, these surveys did not cover all areas of the outer Forth area. For the purposes of assessment a regional winter population of 5,000 birds is assumed.

Nature conservation importance

For EIA assessment purposes, the nature conservation importance of little auk using the offshore site is rated as moderate during the non-breeding season. This classification is merited because the Outer Forth area, including the Neart na Gaoithe offshore site, host a wintering population that likely exceeds 1% of the national population, although this is poorly defined and quantified. This species is not subject to any special legislative protection, nor is it on any conservation priority lists.

Offshore wind farm studies of little auk

There are very few records and therefore little field-based evidence of the likely effects of operational wind farms on little auks. This is because all existing offshore wind farms for which published results are available are located in areas where little auks are naturally scarce. Occasional little auks were recorded at Horns Rev but no other details were given (Petersen, 2005). At Arklow Bank, Ireland, two of three little auks recorded during 5-years of post-construction monitoring were within *ca.* 500 m of the turbine row (Barton *et al*., 2010).

The review of offshore wind farm effects on birds categorises displacement, barrier and collision risk effects all as unknown for little auk (Diersche and Garthe, 2006) and no little auk fatalities are reported in a review of the number of collision victims at wind farms in eight European countries (Hötker *et al*., 2006) although the very low probability of detecting seabird fatalities should be recognised together with the natural scarcity of this species in the areas studied.

EIA Construction Phase assessment for little auk

The construction phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Construction activities will involve the use of a number of vessels to install the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The installation of turbines may also cause the temporary displacement of prey species depending on the installation technique, e.g. pile-driving, which may cause seabirds to forage elsewhere until their prey return.

The extent that birds may be displaced by vessels varies depending on the type and speed of the vessel and possibly the time of year. Little auks are not thought to be particularly sensitive to vessel movements but birds disturbed by vessels will either swim or fly away to an alternative location until the vessels have passed, after which they may return to the area. Little auks only occur in the offshore site in the winter months, so will not be present for much of the year. Consequently, there may be a localised, short-term effect.

Based on this, possible displacement impacts were categorised as being of negligible (<1%) magnitude, and being temporally short-term and reversible (Table 3.1 and Table 3.2). Overall, it is concluded that the impacts of displacement during construction operations on the regional population of little auks in the non-breeding period is ***not significant*** under the EIA Regulations.

EIA Operational Phase assessment for little auk

Potential for little auk to be affected by displacement

The value of the offshore site buffered to 1 km as a foraging site to little auks in the non-breeding period was estimated from the mean proportion of the assumed regional population present from November to February (Table 4.97). On this basis, it is inferred that the offshore site buffered to 1 km provides up to 1.8% of the foraging resources of the regional population in the non-breeding period (Table 4.97).

Table 4.97 The mean estimated number and percentage of the regional population of little auks potentially at risk of displacement from the offshore site and 1 km buffer in the non-breeding period

Receptor population	Population size (adults)	No. assumed at sea	Dev.Area		Dev.Area + 1km	
			Mean no. at risk	%	Mean no. at risk	%
North Sea, non-breeding period (Skov *et al.*, 1995)	852,690	852,690	54	<0.1	88	<0.1
Regional, non-breeding period (Recent surveys)	5,000	5,000	54	1.1	88	1.8

Likely impacts of displacement on little auk population

For the purposes of assessment it is assumed that 50% of the little auks that would otherwise forage in the offshore site buffered to 1 km would be displaced elsewhere when the proposed wind farm is operational.

If 50% of little auks were to be displaced from the offshore site buffered to 1 km the impact of this would be the effective loss of up to 0.90% of the foraging habitat of the regional population during the non-breeding-period. This impact is categorised as negligible magnitude, and temporally long-term and reversible (Table 3.1 and Table 3.2). The sensitivity of the population to displacement at this time of year is unknown, but is likely to be negligible to low. It is concluded that the impact of displacement on the regional little auk population in the non-breeding period is ***not significant*** under the EIA Regulations.

Disturbance by vessels

During the construction and decommissioning phases there will be increased vessel traffic in the offshore site associated with the installation or removal of turbines, cables and offshore substation (Section 3.4.7). There will also be vessel traffic associated with routine maintenance during the operational phase of the project.

The presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The extent that birds may be displaced by vessels varies depending on the type and speed of the vessel and possibly the time of year. Little auks are not thought to be particularly sensitive to vessel movements but birds disturbed by vessels will either swim or fly away to an alternative location until the vessels have passed, after which they may return to the area. Little auks only occur in the offshore site in the winter months, so will not be present for much of the year. Consequently, there may be a localised, short-term effect.

Based on this, possible displacement impacts were categorised as being of negligible (<1%) magnitude, and being temporally short-term and reversible (Table 3.1 and Table 3.2). Overall, it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional population of little auks in the non-breeding periods is ***not significant*** under the EIA Regulations.

Collision mortality

Collision Risk Modelling was not undertaken for little auk because all birds seen in flight during the baseline surveys were below the proposed minimum rotor swept height of turbines. Therefore, it is not plausible that this species will experience mortality from collision with turbine rotors.

The potential effect of the predicted collision mortality of little auks on the baseline mortality rate is rated as negligible in magnitude (<1%), temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of collision mortality are ***not significant*** under the terms of the EIA Regulations.

EIA Decommissioning Phase assessment for little auk

The decommissioning phase will be of relatively short duration and consequently potential impacts arising during this period are predicted to also be of short duration. Decommissioning activities will involve the use of a number of vessels to remove the turbines and cables and the presence of these vessels and their activities may cause disturbance and consequently displacement to species that avoid them, e.g. divers and scoter. The removal of turbines may also cause the temporary displacement of prey species depending on the removal technique.

The extent that birds may be displaced by vessels varies depending on the type and speed of the vessel and possibly the time of year. Little auks are not thought to be particularly sensitive to vessel movements but birds disturbed by vessels will either swim or fly away to an alternative location until the vessels have passed, after which they may return to the area. Little auks only occur in the offshore site in the winter months, so will not be present for much of the year. Consequently, there may be a localised, short-term effect.

Based on this, possible displacement impacts were categorised as being of negligible (<1%) magnitude, and being temporally short-term and reversible (Table 3.1 and Table 3.2). Overall, it is concluded that the impacts of displacement during decommissioning operations on the regional population of little auks in the non-breeding periods is ***not significant*** under the EIA Regulations.

EIA summary of effects combined for little auk

The adverse impacts of the effects assessed will act in a broadly additive manner. In combination it is judged that the magnitude of the three effects on the regional population is low. It is concluded that the overall impact on the regional population of little auks in the non-breeding (winter) period is ***not significant*** under the EIA regulations (Table 4.98).

Table 4.98 Summary of effects on the regional population of little auks in the non-breeding (winter) period

Effect	Spatial Magnitude	Temporal Magnitude	Sensitivity	Significance
Construction and decommissioning phases				
Vessel disturbance	Negligible	Short term	Low	***Not significant***
Operational phase				
Displacement from foraging habitat	Negligible	Long term	Low	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality	Negligible	Long term	Low	***Not significant***
All effects combined	Negligible	Long term	Low	***Not significant***

4.4.30.5 Cumulative Impact Assessment for little auk

Combining the predicted individual impacts for the three proposed offshore wind farms suggests that the overall impacts to the regional population of little auks in the non-breeding-period will be as shown in Table 4.99. The individual impacts for displacement and collision are combined by addition to give the cumulative impact.

Table 4.99 Summary of CIA for the three proposed offshore wind farms in south-east Scotland on the regional population of little auks in the non-breeding (winter) period

Effect Assessed	Assumed amount of potential displacement realised*	Predicted impact from Neart na Gaoithe pOWF	Predicted impact from Inch Cape pOWF	Predicted impact from Outer Forth pOWF	Cumulative Impact
Displacement: effective loss of foraging habitat during the non-breeding period (%)	50%	0.9%, Negligible	0.5, Negligible	4.2, Low	5.6, Moderate
Collision: % increase in assumed annual adult mortality rate. All designs.	0%	No change, Negligible	No change, Negligible	No change, Negligible	No change, Negligible

The predicted effective loss of up to 5.6% of the foraging habitat of the regional little auk population in the non-breeding period is rated as moderate magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). Bearing in mind the low sensitivity of overwintering little auks to displacement effects, it is concluded that the cumulative impact of displacement caused by the proposed offshore wind farms on the regional little auk population in the non-breeding period is an effect of ***minor significance*** under the terms of the EIA Regulations.

The potential impact of collision mortality on the baseline mortality rate of the regional population of little auk in the non-breeding period is rated as negligible in magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative impact of collision mortality caused by the three proposed offshore wind farms on the regional little auk population in the non-breeding (winter) period is ***not significant*** under the terms of the EIA Regulations.

4.4.30.6 Mitigation measures for little auk

The assessment does not identify any significant adverse effects on the regional little auk population in the non-breeding period. Therefore no mitigation measures are required for this species.

4.4.31 Non-seabirds

A total of 1,209 birds of 19 species of non-seabird were recorded in the Neart na Gaoithe study area in Year 1. In Year 2, 424 birds of 22 species were recorded on surveys (Table 4.100).

Table 4.100 Comparison of numbers of non-seabird species recorded in the Neart na Gaoithe Study Area in Years 1 and 2

Species	Year 1			Year 2		
	Offshore Site	Buffer Area	Total	Offshore Site	Buffer Area	Total
Mute swan	0	2	2	0	0	0
Pink-footed goose	0	216	216	0	333	333
Barnacle goose	180	720	900	0	0	0
Wigeon	1	20	21	0	5	5
Shoveler	0	2	2	0	0	0
Tufted duck	0	0	0	0	1	1
Dabbling duck species	0	1	1	0	0	0
Merlin	0	0	0	0	1	1
Oystercatcher	0	2	2	0	2	2
Ringed plover	0	0	0	0	1	1
Golden plover	0	4	4	1	19	20
Sanderling	0	2	2	0	0	0
Purple sandpiper	0	1	1	0	0	0
Little stint	0	0	0	0	1	1
Dunlin	0	3	3	4	1	5

Bar-tailed godwit	0	1	1	0	0	0
Curlew	0	7	7	0	1	1
Redshank	0	2	2	0	2	2
Turnstone	0	0	0	0	2	2
Short-eared owl	0	0	0	0	1	1
Sand martin	0	1	1	0	0	0
Swallow	0	2	2	0	2	2
Skylark	3	0	3	0	0	0
Meadow pipit	1	32	33	8	30	38
Robin	0	0	0	1	1	2
Bluethroat	0	0	0	0	1	1
Wheatear	0	0	0	0	1	1
Song thrush	0	0	0	0	1	1
Blackbird	1	1	2	0	0	0
Fieldfare	0	0	0	0	1	1
Barred warbler	0	0	0	0	1	1
Starling	0	4	4	0	0	0
Carrion crow	0	0	0	1	0	1
Passerine species	0	0	0	0	1	1
Total numbers	**186**	**1,023**	**1,209**	**15**	**409**	**424**

A monthly breakdown of numbers of non-seabirds on surveys is given in Annex 1 of this report.

In Year 1, the three most frequently recorded species were barnacle goose, pink-footed goose and meadow pipit, accounting for 95.0% of all non-seabirds recorded. In Year 2, pink-footed goose, meadow pipit and golden plover were the three most frequently recorded species, accounting for 92.2% of all non-seabirds recorded (Table 4.100).

In both years, the majority of non-seabirds were recorded in the buffer area, with 15.4% of all non-seabirds recorded in the offshore site in Year 1, and 3.5% recorded in the offshore site in Year 2.

A brief summary of the non-seabird species recorded on baseline surveys is given below.

4.4.31.1 Mute swan

Mute swans are typically found on freshwater lakes and rivers and would be unlikely to occur regularly offshore. Two mute swans were recorded in the buffer area, flying below 7.5 m in April of Year 1 (Table 4.100).

4.4.31.2 Pink-footed goose

Scotland is a key wintering area for pink-footed geese breeding in Iceland and Greenland, with large roosting and feeding flocks occurring in autumn and winter months, particularly in eastern and central parts of the country (Forrester *et al.,* 2007).

A total of 216 pink-footed geese were recorded in the Neart na Gaoithe study area in Year 1, with 90 birds in November 2010, 34 in February 2010, 1 in March 2010 and 91 in September

2010 (Table 4.100). All sightings were recorded in the buffer area, to the north of the offshore site. In Year 2, 300 pink-footed geese were recorded flying through the buffer area in January, with a further 33 recorded in March.

Less than half of all pink-footed geese (42.6%) were recorded flying above 22.5 m in height i.e. within the rotor zone (Table 4.3).

Pink-footed goose is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

Langston (2010) assessed pink-footed goose as being at moderate risk of collision and displacement resulting from offshore wind farms, while barrier effects were rated as low risk. Overall, pink-footed goose was assessed as being at high risk from offshore wind developments (Table 4.4).

Pink-footed goose is listed as a qualifying interest species in autumn for eight SPAs, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.1 Barnacle goose

The Svalbard breeding population of Barnacle Geese spend the winter around the Solway Firth in south-west Scotland, with migrating birds flying over the North Sea in autumn, reaching the Scottish east coast, before continuing onto the Solway. Landfall is typically between Lothian and Northumberland, although landfall can occur anywhere on the east coast of Scotland or England, depending on wind and weather conditions.

A total of 900 barnacle geese were recorded in the Neart na Gaoithe study area on 12th October of Year 1 (Table 4.100). Three flocks totalling 720 birds were recorded in the buffer area, with a flock of 180 birds in the offshore site. All 900 birds were recorded flying below 7.5 m in height, and were flying south or south-west (Table 4.3). Barnacle goose was not recorded on surveys in Year 2.

Barnacle goose is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

Langston (2010) assessed barnacle goose as being at moderate risk of collision and displacement resulting from offshore wind farms, while barrier effects were rated as low risk. Overall, barnacle goose was assessed as being at high risk from offshore wind developments (Table 4.4).

Barnacle goose is listed as a qualifying interest species in autumn for two SPAs which could potentially be affected by the Neart na Gaoithe development (Loch of Strathbeg and Upper Solway Flats and Marshes) (Table 4.101). The Loch of Strathbeg is primarily an autumn arrival point for birds migrating to the wintering grounds, while the Upper Solway holds the entire wintering population of Svalbard barnacle geese (JNCC 2012).

Table 4.101 Qualifying non-seabird species for coastal SPAs recorded during Years 1 and 2 that could potentially be affected by the Neart na Gaoithe development

Species	SPA site	Distance to site (km)	Site total	% of national popn	% of biogeo graphic popn
Pink-footed goose	Fala Flow	58	6,719	3.5	3.0
	Firth of Forth	15	12,400	6.5	5.5
	Firth of Tay & Eden Estuary	35	3,769	2.0	1.7
	Gladhouse Reservoir	71	3,068	1.6	1.4
	Loch Leven	65	18,230	9.5	8.1
	Montrose Basin	40	31,622	16.5	14.1
	South Tayside Goose Roosts	50	43,300	22.6	19.2
	Ythan Estuary, Sands of Forvie & Meikle Loch	109	17,213	9.0	7.7
Barnacle goose	Loch of Strathbeg	160	226	1.3	1.9
	Upper Solway Flats & Marshes	190	13,595	c. 100	c. 100
Wigeon	Firth of Forth	15	2,139	0.78	0.2
	Lindisfarne	60	13,375	4.8	1.1
	Montrose Basin	40	4,340	1.6	0.4
	South Tayside Goose Roosts	50		5.0	
Shoveler	Loch Leven	65	520	5.2	1.3
Tufted duck	Loch Leven	65	3,362	5.6	0.3
Oystercatcher	Firth of Forth	15	8,931	2.5	1.0
	Firth of Tay & Eden Estuary	35	4,215	1.2	0.5
	Montrose Basin	40	2,368	0.7	0.3
Ringed plover	Firth of Forth	15	413	1.4	0.9
	Lindisfarne	60	527	1.8	1.0
Golden plover	Firth of Forth	15	2,970	1.2	0.2
	Lindisfarne	60	5,300	2.1	0.3
Sanderling	Firth of Tay & Eden Estuary	35	223	1.0	0.2
Dunlin	Firth of Forth	15	10,033	1.9	0.7

	Firth of Tay & Eden Estuary	35	5,479	1.0	0.4
	Montrose Basin	40	2,244	0.4	0.2
	Lindisfarne	60	7,703	1.5	0.6
Purple sandpiper	Northumbria coast	60	763	3.6	1.5
Bar-tailed godwit	Firth of Forth	15	2,600	5.0	2.3
	Firth of Tay & Eden Estuary	35	2,400	4.6	2.1
	Lindisfarne	60	2,946	5.6	2.6
Curlew	Firth of Forth	15	2,188	1.9	0.6
Redshank	Firth of Forth	15	3,700	3.3	2.1
	Firth of Tay & Eden Estuary	35	1,800	1.6	1.0
	Montrose Basin	40	2,259	2.0	1.3
	Lindisfarne	60	1,192	1.1	0.7
Turnstone	Firth of Forth	15	1,286	2.0	1.9
	Northumbria coast	60	1,456	2.3	2.2

Source: JNCC (2012) (Note, the data listed above are from the time of designation or review; more recent survey work has shown that in many cases numbers have since changed)

4.4.31.2 Wigeon

Wigeon are a common and widespread passage migrant and winter visitor to Scotland, with a small breeding population (Forrester *et al.,* 2007).

A total of 21 birds were recorded in the Neart na Gaoithe study area in Year 1, with 20 birds in the north west of the buffer area in December, and one bird in the south-east of the offshore site in January. A further five wigeon were recorded in the buffer area in March of Year 2 (Table 4.100).

Twenty birds (95.2%) were recorded flying below 22.5 m in height, with one bird recorded flying above 22.5 m and 47.5 m, i.e. within the rotor swept zone, at an estimated 30 m in height (Table 4.3).

Wigeon is listed as a qualifying interest species in the non-breeding period for four SPAs, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.3 Shoveler

Shoveler are typically found on freshwater lakes, coastal lagoons and estuaries (Forrester *et al.,* 2007). The species is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al.*, 2009). Two shoveler were recorded in the south-east buffer area, flying at 20 m in height in May of Year 1 (Table 4.100).

Shoveler is listed as a qualifying interest species in the non-breeding period for one SPA, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.4 Tufted duck

Tufted duck are typically found on freshwater lakes and coastal lagoons (Forrester *et al.,* 2007). The species is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009). One tufted duck was recorded in the buffer area, flying at 25 m in height in October of Year 2 (Table 4.100).

Tufted duck is listed as a qualifying interest species in the non-breeding period for one SPA, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.5 Merlin

Merlin is a scarce resident breeding species in upland areas and a passage and winter visitor to mainly low-lying, coastal areas (Forrester *et al.,* 2007). Merlin is listed on Annex I of the EU Bird Directive (2009/147/EEC), and is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009). One was recorded in the buffer area, flying below 7.5 m in height in October of Year 2 (Table 4.100).

4.4.31.6 Oystercatcher

Oystercatchers are typically found on wintering on rocky coasts and estuaries, and breed in inland lochs and river valleys (Forrester *et al.,* 2007). The species is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009). Two oystercatchers were recorded in the south of the buffer area, flying at 25 m in height in July of Year 1, while in Year 2, two were recorded in the buffer area, flying at 5 m in height in September (Table 4.100).

Oystercatcher is listed as a qualifying interest species in the non-breeding period for three SPAs, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.7 Ringed plover

Ringed plovers are typically found on wintering on rocky coasts and estuaries, and breed at inland reservoirs and river valleys (Forrester *et al.,* 2007). The species is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009). One ringed plover was recorded in the buffer area, flying at 15 m in height, in October of Year 2 (Table 4.100).

Ringed plover is listed as a qualifying interest species in the non-breeding period for two SPAs, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.8 Golden plover

An estimated 15,000 pairs of Golden plover breed in Scotland in upland areas. Outside the breeding season, birds occur on farmland and estuaries (Forrester *et al.,* 2007). Golden plover is listed on Annex I of the EU Bird Directive (2009/147/EEC), and is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

Four golden plovers were recorded in the south-east of the buffer area, in May of Year 1. In Year 2, one golden plover was recorded in the offshore site in October, with 5 in the buffer area in September, and 14 there in October (Table 4.100).

In total, 14 birds (58.3%) were recorded flying below 22.5 m in height, with 10 birds recorded flying above 22.5 m, i.e. within the rotor zone (Table 4.3).

Golden plover is listed as a qualifying interest species in the non-breeding period for two SPAs, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.9 Sanderling

Sanderlings are locally common and widespread passage migrant and winter visitor to Scotland, mostly on sandy beaches and estuaries (Forrester *et al.,* 2007). Single birds were recorded in the south of the buffer area in June and July of Year 1, flying below 7.5 m in height (Table 4.100).

Sanderling is listed as a qualifying interest species in the non-breeding period for one SPA, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.10 Little stint

Little stint are passage migrants to Scotland in spring and more regularly in autumn (Forrester *et al.,* 2007). One was recorded in the buffer area flying at 20 m in height, in July of Year 2 (Table 4.100).

4.4.31.11 Purple sandpiper

There is a very small breeding UK population of purple sandpipers in Scottish upland areas, while the species is common on rocky shores in winter and on passage, particularly on eastern and northern coasts of Scotland (Forrester *et al.,* 2007). Purple sandpiper is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

One purple sandpiper was recorded in the south of the Neart na Gaoithe buffer area, flying below 7.5 m in October of Year 1 (Table 4.100).

Purple sandpiper is listed as a qualifying interest species in the non-breeding period for one SPA, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.12 Dunlin

The Scottish breeding population of dunlin is estimated at between 8,000 and 10,000 pairs, with birds breeding in upland areas. In winter, dunlin are one of the commonest waders on Scottish coasts, with the main concentrations found on estuaries on the east coast (Forrester *et al.,* 2007). Dunlin is currently red-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

Three dunlin were recorded in the Neart na Gaoithe buffer area in Year 1, two in May and 1 in July. In Year 2, four were recorded in the offshore site in December, with nine in the buffer area in September (Table 4.100). All birds were recorded flying below 17.5 m in height.

Dunlin is listed as a qualifying interest species in the non-breeding period for four SPAs, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.13 Bar-tailed godwit

Bar-tailed godwit is a common passage migrant and winter visitor in Scotland, more common on the east coast, with small numbers of non-breeding immature birds present in the summer (Forrester *et al.,* 2007). Bar-tailed godwit is listed on Annex I of the EU Bird Directive (2009/147/EEC), and is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

One bar-tailed godwit was recorded in the north-east of the Neart na Gaoithe buffer area in July of Year 1, flying below 7.5 m in height (Table 4.100).

Bar-tailed godwit is listed as a qualifying interest species in the non-breeding period for three SPAs, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.14 Curlew

Curlews breed on farmland and in upland areas, and spend the winter months in coastal areas on farmland and estuaries (Forrester *et al.,* 2007). The species is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

Seven curlews were recorded in the Neart na Gaoithe buffer area in Year 1, with six birds recorded flying at an estimated 20 m in height in the south of the buffer area in July and one bird flying at 10 m in height, south of the offshore site in September. In Year 2, one was recorded in the buffer area, flying at 10 m in height in July of Year 2 (Table 4.100).

Curlew is listed as a qualifying interest species in the non-breeding period for one SPA, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.15 Redshank

Redshank is a common and widespread breeding species throughout Scotland. It is also a very common wintering species and passage migrant (Forrester *et al.,* 2007). The species is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009).

Two redshanks were recorded in the north of the Neart na Gaoithe buffer area in July of Year 1, flying below 7.5 m in height. Two were recorded in the buffer area, flying below 7.5 m in height in December of Year 2 (Table 4.100).

Redshank is listed as a qualifying interest species in the non-breeding period for four SPAs, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.16 Turnstone

Turnstone is a common and widespread winter visitor and passage migrant to coastal areas of Scotland (Forrester *et al.,* 2007). Two were recorded in the buffer area, flying below 7.5 m in height in August of Year 2 (Table 4.100).

Turnstone is listed as a qualifying interest species in the non-breeding period for two SPAs, which could potentially be affected by the Neart na Gaoithe development (Table 4.101).

4.4.31.17 Short-eared owl

Short-eared owl is a scarce resident breeding species in upland areas and a passage and winter visitor to mainly low-lying, coastal areas. Birds from continental Europe cross the North Sea on autumn migration and may remain throughout the winter (Forrester *et al.,* 2007). Short-eared owl is listed on Annex I of the EU Bird Directive (2009/147/EEC), and is currently amber-listed on the UK Birds of Conservation Concern list (Eaton *et al*., 2009). One was recorded in the buffer area, flying below 7.5 m in height in October of Year 2 (Table 4.100).

4.4.31.18 Passerines

Six species of passerines (or perching birds) were recorded in the Neart na Gaoithe study area in Year 1, with three species seen in the offshore site; one meadow pipit, one blackbird and three skylarks (Table 4.100). Meadow pipit was the most frequently recorded species, with a total of 31 birds, of which 18 were recorded in April and 11 in September. The remaining species were recorded occasionally in lower numbers.

In Year 2, eight species were recorded, with three species seen in the offshore site; eight meadow pipits, one robin and one carrion crow (Table 4.100). As in Year 1, meadow pipit was the most frequently recorded species, with a total of 38 birds, of which 17 were recorded in September and 11 in October. The remaining species were recorded occasionally in lower numbers.

Overall, meadow pipit was the only species of land bird with a sample size greater than 20 birds in Years 1 and 2. Of the 59 meadow pipits recorded, 58 (97.3%) were flying below 22.5 m in height (Table 4.3).

4.4.31.19 Assessment

The majority of the non-seabird species recorded on baseline surveys in the Neart na Gaoithe Survey Area were only recorded in low numbers, with greatest variety of species occurring in the autumn passage period and to a lesser extent the spring passage period.

Displacement

Displacement is defined as the potential for the wind farm and associated human activities to reduce or prevent birds, including flying birds, from using the offshore site and is therefore akin to habitat loss.

As wildfowl and waders are not seabirds, they are very unlikely to regularly use the offshore site for foraging or resting, although it is possible that geese and ducks could land on the sea to rest during migration. However, there are no species of wildfowl and wader likely to forage regularly in the offshore site. As such, it is concluded that the effects of displacement on non-seabirds such as wildfowl and waders is ***not significant*** under the terms of the EIA Regulations.

Barrier effect

The assessment of displacement of flying birds transiting around through the offshore site instead of through it is considered under barrier effects. As species of wildfowl and waders do not regularly forage at sea, the only possible barrier effect can occur during migration periods in spring and autumn, when birds are moving to or from breeding grounds to wintering areas. If birds avoided the wind farm area and flew around it instead of through it, then there could potentially be an impact on these birds as a result of the increased length of their journey.

However, the magnitude of any such barrier effect is likely to be negligible for species of wildfowl and waders on migration, as the migration journeys that they undertake are typically hundreds or thousands of kilometres long. Therefore, the potential incremental increase in distance as a result of having to fly around the wind farm rather than through it will be negligible compared to the overall distance flown during migration. Overall, it is concluded that the effects of barrier effect on non-seabirds such as wildfowl and waders is ***not significant*** under the terms of the EIA Regulations.

Collision Mortality

Collision risk modelling was undertaken for 15 species of geese and waders based on an assumed population of 1,000 birds of each species passing through the offshore site twice per year, on spring and autumn passage, with all birds flying at rotor height. The predicted number of collisions per year, based on a selection of avoidance rates are shown in Table 4.102. Further details are presented in Appendix 12.2.

Table 4.102 Predicted number of collisions per year for 15 species of geese and waders

Species	Annual no of collisions at different avoidance rates				
	No avoidance	95% avoidance	98% avoidance	99% avoidance	99.5% avoidance
Bean goose [1]	423	21	8	4	2
Pink-footed goose	410	20	8	4	2
Barnacle goose	390	19	8	4	2
Bar-tailed godit	321	16	6	3	2
Black-tailed godwit [1]	323	16	6	3	2
Knot [1]	297	15	6	3	1
Curlew	360	18	7	4	2
Dunlin	288	14	6	3	1
Sanderling	293	15	6	3	1
Grey plover [1]	307	15	6	3	2
Lapwing [1]	325	16	6	3	2
Ringed plover	286	14	6	3	1
Redshank	333	17	7	3	2
Turnstone	299	15	6	3	1
Oystercatcher	348	17	7	3	2

1 Species not recorded in Neart na Gaoithe Survey Area on baseline surveys

Using a 98% avoidance rate, and an assumed population of 1,000 birds passing through the wind farm twice a year at rotor height gives a peak predicted number of eight collisions per year for the three species of geese, which is equivalent to 0.8% of the assumed population (1,000 birds). Using the same parameters for the 12 species of waders gives a peak predicted number of seven collisons per year, which is equivalent to 0.7% of the assumed population (1,000 birds).

Overall, the potential effect of the collision mortality of these 15 species of geese and waders on the baseline mortality rate is rated as negligible in magnitude, temporally long-term and reversible. It is concluded that the effects of collision mortality on these 15 species of geese and waders is therefore ***not significant*** under the terms of the EIA Regulations.

4.4.31.20 Cumulative Impact Assessment for non-seabirds

CIA has not been undertaken for non-seabirds such as wildfowl and waders because the predicted effects of Neart na Gaoithe wind farm for these species were very close to no effect. Therefore it is not plausible that the offshore site could contribute to a significant cumulative impact for these species.

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Annex 1

Monthly survey effort in relation to sea state;

Monthly species totals;

Monthly survey routes

Table A1 Sea state during surveys in the Neart na Gaoithe Study Area in Years 1 and 2

Month	Year 1			Year 2		
	Sea State	Km travelled	Percentage	Sea State	Km travelled	Percentage
November	1	102.9	31.4	-	-	-
	2	224.6	68.6			
December	0	18.4	5.7	0	-	-
	1	113.8	35.0	1	-	-
	2	77.3	23.8	2	51.6	16.3 %
	3	62.9	19.4	3	115.0	36.2 %
	4	33.9	10.4	4	141.8	44.7 %
	5	18.8	5.8	5	8.9	2.9 %
January	2	112.4	34.6	2	42.2	13.0 %
	3	121.0	37.3	3	231.7	71.4 %
	4	78.9	24.3	4	50.6	15.6 %
	5	12.4	3.8	5	-	-
February	1	-	-	1	47.2	14.2 %
	2	129.7	39.6 %	2	99.9	30.1 %
	3	150.1	45.8 %	3	150.6	45.4 %
	4	35.7	10.9 %	4	34.2	10.3 %
	5	12.2	3.7 %	5	-	-
March	0	135.4	41.0 %	0	61.3	18.3 %
	1	60.0	18.2 %	1	101.1	30.2 %
	2	134.7	40.8 %	2	64.8	19.3 %
	3	-	-	3	93.5	27.9 %
	4	-	-	4	13.1	3.9 %
	5	-	-	5	1.2	0.4 %
April	0	54.5	16.7 %	0	45.9	13.9 %
	1	174.3	53.3 %	1	52.8	16.1 %
	2	98.5	30.1 %	2	103.3	31.4 %
	3	-	-	3	77.4	23.5 %
	4	-	-	4	45.4	13.8 %
	5	-	-	5	3.8	1.2 %

Table A1 Sea state during surveys in the Neart na Gaoithe Study Area in Years 1 and 2 (continued)

Month	Year 1			Year 2		
	Sea State	Km travelled	Percentage	Sea State	Km travelled	Percentage
May	1	37.3	11.5 %	1	13.2	3.8 %
	2	198.6	60.6 %	2	166.7	48.6 %
	3	76.4	23.3 %	3	72.2	21.0 %
	4	15.4	4.7 %	4	78.3	22.8 %
	5	-	-	5	0.3	0.1 %
June	0	13.7	4.2 %	0	-	-
	1	37.0	11.3 %	1	13.1	4.0 %
	2	50.6	15.5 %	2	78.1	23.7 %
	3	198.1	60.6 %	3	235.6	71.6 %
	4	27.4	8.4 %	4	2.1	0.6 %
July	1	-	-	1	64.6	19.6 %
	2	67.8	20.7	2	176.8	53.6 %
	3	146.5	44.6	3	88.5	26.8 %
	4	106.5	32.4	4	-	-
	5	7.8	2.4	5	-	-
August	0	16.8	5.1			
	1	55.2	16.8			
	2	136.3	41.3			
	3	121.6	36.9			
September	0	21.5	6.6			
	1	9.8	3.0			
	2	158.8	48.4			
	3	105.8	32.3			
	4	32.2	9.8			
October	0	57.0	17.3			
	1	164.5	49.9			
	2	95.5	29.0			
	3	12.5	3.8			

Table A2 Numbers of seabirds recorded in the Neart na Gaoithe offshore site in Year 1 (Raw numbers, all sea states)

Species	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Total
Fulmar	5	12	20	13	13	2	3	7	8	7	21	1	112
Sooty shearwater	1	0	0	0	0	0	0	0	0	0	2	81	84
Manx shearwater	0	0	0	0	0	0	0	4	0	8	3	1	16
Balearic shearwater	0	0	0	0	0	0	0	0	0	0	1	0	1
Gannet	10	1	2	14	133	61	273	339	222	129	310	99	1,593
Eider	2	7	0	0	0	0	0	0	0	0	0	0	9
Common scoter	0	0	0	0	0	0	0	0	0	0	0	5	5
Grey phalarope	1	0	0	0	0	0	0	0	0	0	0	0	1
Great skua	0	0	0	0	0	0	0	0	0	0	0	1	1
Little gull	0	0	0	0	0	0	0	0	0	2	0	30	32
Sabine's gull	0	0	0	0	0	0	0	0	0	1	0	0	1
Common gull	2	0	0	0	0	0	0	0	1	0	0	3	6
Lesser black-backed gull	0	1	0	0	0	1	1	4	2	1	0	0	10
Herring gull	4	10	6	3	10	1	1	12	0	2	0	1	50
Great black-backed gull	1	3	11	3	1	0	0	0	0	0	5	1	25
Large gull species	0	3	0	1	0	0	0	0	0	0	0	0	4
Kittiwake	11	6	0	1	3	10	29	25	16	14	291	395	801
Common tern	0	0	0	0	0	0	0	0	0	0	3	0	3
Arctic tern	0	0	0	0	0	0	0	0	0	205	0	0	205
Common/Arctic tern	0	0	0	0	0	0	0	0	0	1	0	0	1
Guillemot	88	22	2	19	56	55	89	69	15	1	131	705	1,252
Razorbill	27	11	1	2	58	3	13	4	10	123	61	283	596
Little auk	20	0	0	6	0	0	0	0	0	0	0	0	26
Puffin	1	0	0	0	35	271	29	41	135	574	44	176	1,306
Guillemot/razorbill	0	4	7	2	24	9	9	2	1	18	23	269	368
Unidentified auk species	1	1	0	0	0	1	1	4	0	115	32	0	155
Total numbers	**174**	**81**	**49**	**64**	**333**	**414**	**448**	**511**	**410**	**1,201**	**927**	**2,051**	**6,663**

Table A3 Numbers of seabirds recorded in the Neart na Gaoithe offshore site in Year 2 (Raw numbers, all sea states)

Species	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Total
Fulmar	-	40	17	10	15	11	10	16	17	21	30	2	189
Sooty shearwater	-	0	0	0	0	0	0	0	0	1	2	1	4
Manx shearwater	-	0	0	0	0	0	0	0	17	0	7	3	27
Gannet	-	8	6	176	91	595	418	410	462	361	437	158	3,122
Little gull	-	0	0	0	0	0	0	0	0	0	6	0	6
Common gull	-	10	0	1	0	1	0	0	0	0	0	0	12
Lesser black-backed gull	-	0	0	0	0	0	0	2	8	1	0	0	11
Herring gull	-	8	15	10	6	1	2	6	4	1	0	5	58
Great black-backed gull	-	8	4	2	1	1	0	1	0	0	1	2	20
Large gull species	-	0	0	0	0	0	0	0	0	1	0	0	1
Kittiwake	-	137	3	7	12	69	150	39	220	19	12	51	719
Common tern	-	0	0	0	0	0	0	0	0	0	13	0	13
Arctic tern	-	0	0	0	0	0	20	0	0	14	3	0	37
Common/Arctic tern	-	0	0	0	0	0	1	0	0	6	21	0	28
Guillemot	-	81	62	82	102	427	147	88	121	32	303	99	1,544
Razorbill	-	40	1	13	7	23	57	18	33	7	16	135	350
Little auk	-	10	6	0	0	0	0	0	0	0	0	0	16
Puffin	-	2	0	5	6	233	230	104	298	66	28	138	1,110
Guillemot/razorbill	-	14	3	48	3	20	1	0	0	0	65	14	168
Unidentified auk species	-	0	0	2	0	0	0	14	20	0	16	4	56
Total numbers	**-**	**358**	**117**	**356**	**243**	**1,381**	**1,036**	**698**	**1,200**	**530**	**960**	**612**	**7,491**

Table A4 Numbers of seabirds recorded in the Neart na Gaoithe buffer area in Year 1 (Raw numbers, all sea states)

Species	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Total
Red-throated diver	0	0	0	0	0	1	0	3	0	0	0	1	5
Fulmar	48	88	81	55	84	36	25	17	26	39	81	0	580
Sooty shearwater	0	0	0	0	0	0	0	0	0	1	25	117	143
Manx shearwater	0	1	0	0	0	0	7	15	8	14	6	5	56
Storm petrel	0	0	0	0	0	0	0	0	0	00	0	1	1
Gannet	128	16	5	191	973	538	1,263	2,478	1,559	1,732	1,905	582	11,370
Cormorant	0	0	0	0	1	0	0	0	0	0	0	0	1
Shag	0	0	0	0	4	3	0	0	0	0	0	4	11
Eider	10	1	0	0	0	0	0	0	0	0	0	0	11
Pomarine skua	0	0	0	0	0	0	0	0	0	0	0	6	6
Arctic skua	0	0	0	0	0	0	0	0	1	0	4	1	6
Great skua	2	0	0	0	0	1	0	0	0	2	16	2	23
Little gull	6	0	0	0	0	0	0	0	0	15	24	221	266
Black-headed gull	25	0	1	0	0	0	0	0	1	0	0	0	27
Common gull	14	0	18	0	8	1	0	0	0	0	3	28	72
Lesser black-backed gull	3	2	0	0	0	8	6	22	8	9	1	0	59
Herring gull	616	166	644	52	33	48	8	62	14	8	12	9	1,672
Great black-backed gull	84	11	307	8	6	3	1	4	1	0	32	46	503
Large gull species	0	20	2	1	0	32	0	5	0	1	1	93	155
Kittiwake	140	65	1	48	28	371	94	215	125	119	899	1,049	3,154
Common tern	1	0	0	0	0	0	0	0	2	0	7	0	10
Arctic tern	0	0	0	0	0	0	1	0	7	644	0	0	652
Common/Arctic tern	0	0	0	0	0	0	2	0	0	65	8	0	75
Tern species	0	0	0	0	0	0	0	0	0	34	0	0	34
Guillemot	295	143	136	254	552	653	619	822	134	133	1,052	1,853	6,646
Razorbill	83	47	4	55	209	75	67	78	57	427	239	2,043	3,384
Little auk	49	9	0	51	0	0	0	0	0	0	0	0	109
Puffin	5	15	1	1	0	750	703	624	768	5,862	60	1,002	9,791
Puffin/little auk	2	0		1	0	0	0	0	0	0	0	0	3
Guillemot/razorbill	77	26	28	77	165	66	205	52	8	34	249	1,968	2,955
Unidentified auk species	5	7	0	0	7	13	37	47	12	443	328	294	1,193
Total numbers	**1,593**	**617**	**1,228**	**794**	**2,070**	**2,599**	**3,038**	**4,444**	**2,731**	**9,582**	**4,952**	**9,325**	**42,973**

Table A5 Numbers of seabirds recorded in the Neart na Gaoithe buffer area in Year 2 (Raw numbers, all sea states)

Species	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Total
Fulmar	-	35	117	59	58	36	51	55	53	232	208	23	927
Sooty shearwater	-	0	0	0	0	0	0	0	0	2	58	115	175
Manx shearwater	-	0	0	0	0	2	8	9	46	0	180	14	259
Gannet	-	21	100	663	652	2,265	1,877	2,664	1,862	2,301	2,645	1,244	16,294
Cormorant	-	0	0	0	0	0	0	0	0	0	3	0	3
Shag	-	0	0	2	4	0	0	0	0	0	0	0	6
Eider	-	0	2	0	0	0	0	0	0	0	0	0	2
Common scoter	-	0	0	0	0	0	0	0	0	0	2	0	2
Red-necked phalarope	-	0	0	0	0	0	0	0	0	0	1	0	1
Grey phalarope	-	0	0	0	0	0	0	0	0	0	0	2	2
Arctic skua	-	0	0	0	0	0	0	0	0	2	5	11	18
Great skua	-	0	0	0	0	1	0	0	0	1	5	9	16
Little gull	-	0	0	1	0	0	0	0	0	0	92	121	214
Sabine's gull	-	0	0	0	0	0	0	1	0	0	0	0	1
Black-headed gull	-	0	0	0	1	0	0	0	0	6	0	4	11
Common gull	-	15	5	4	8	1	0	1	0	4	2	0	40
Small gull species	-	0	1	0	0	0	0	0	0	0	0	0	1
Lesser black-backed gull	-	0	0	0	0	6	2	11	39	6	120	0	184
Herring gull	-	199	191	76	56	714	12	38	68	9	11	1	1,375
Great black-backed gull	-	97	48	10	7	142	5	1	0	2	88	14	414
Large gull species	-	5	1	0	0	331	0	5	0	1	3	1	347
Kittiwake	-	86	231	59	196	163	246	313	838	120	121	1,031	3,404
Gull species	-	0	0	0	1	0	0	0	0	0	0	0	1
Common tern	-	0	0	0	0	0	2	0	0	0	35	0	37
Arctic tern	-	0	0	0	0	0	66	2	5	105	112	2	292
Common/Arctic tern	-	0	0	0	0	0	1	0	0	80	86	0	167
Guillemot	-	461	627	553	1,027	652	1,207	731	726	568	2,242	1,392	10,186
Razorbill	-	39	33	89	87	68	163	78	189	54	300	1,681	2,781
Little auk	-	54	40	3	0	0	0	0	0	0	0	0	97
Puffin	-	3	9	17	40	865	1,150	715	972	246	297	1,198	5,512
Guillemot/razorbill	-	49	63	246	230	26	32	9	6	73	478	152	1,364
Unidentified auk species	-	1	9	5	4	8	7	68	14	3	491	161	771
Total numbers	**-**	**1,065**	**1,477**	**1,787**	**2,371**	**5,280**	**4,829**	**4,701**	**4,818**	**3,815**	**7,585**	**7,176**	**44,904**

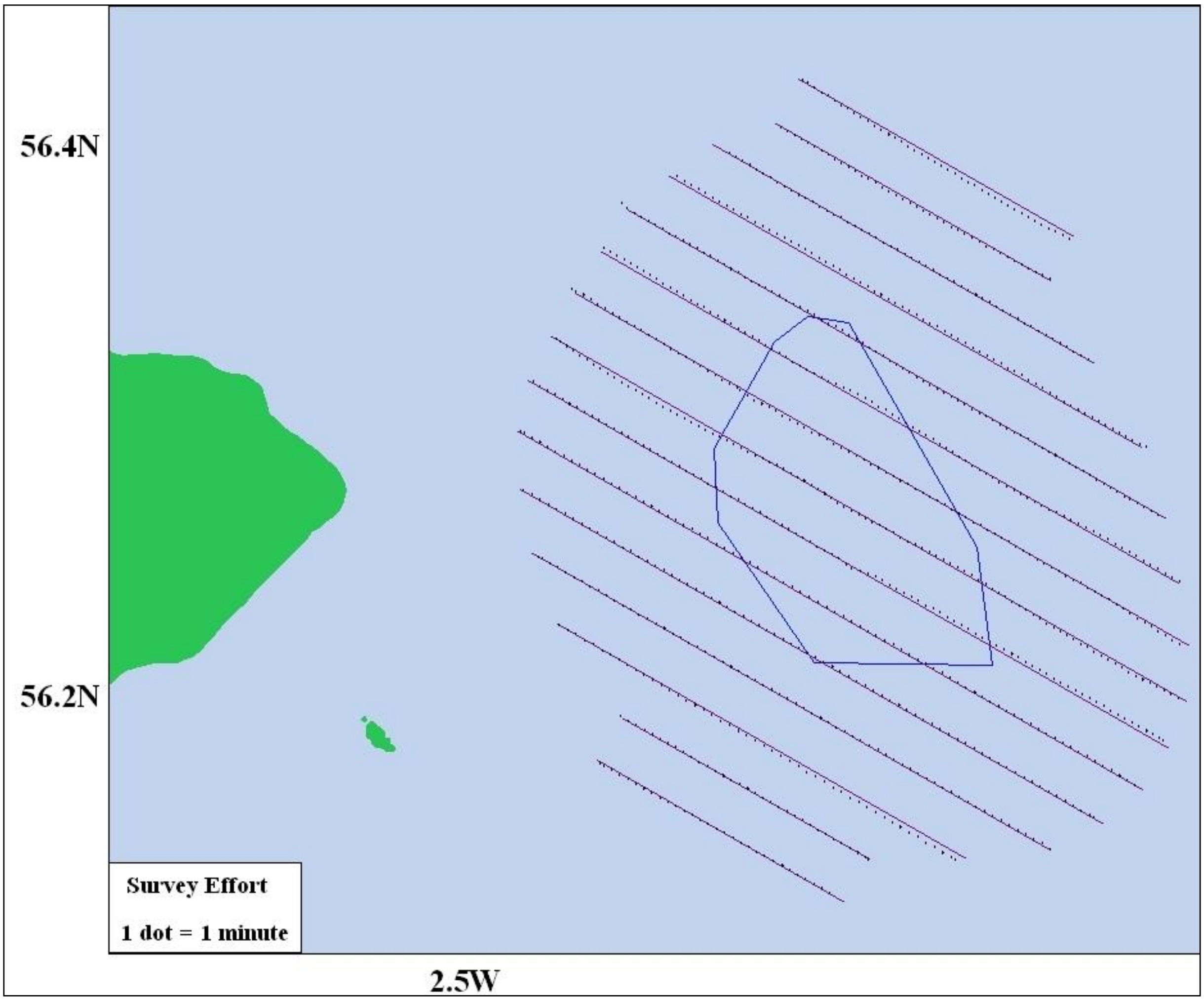
Table A6 Numbers of non-seabird species recorded in the Neart na Gaoithe Study Area in Year 1

Species	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Total
Mute swan	0	0	0	0	0	2	0	0	0	0	0	0	2
Pink-footed goose	90	0	0	34	1	0	0	0	0	0	91	0	216
Barnacle goose	0	0	0	0	0	0	0	0	0	0	0	900	900
Wigeon	0	20	1	0	0	0	0	0	0	0	0	0	21
Shoveler	0	0	0	0	0	0	2	0	0	0	0	0	2
Dabbling duck species	1	0	0	0	0	0	0	0	0	0	0	0	1
Oystercatcher	0	0	0	0	0	0	0	0	2	0	0	0	2
Golden plover	0	0	0	0	0	0	4	0	0	0	0	0	4
Sanderling	0	0	0	0	0	0	0	1	1	0	0	0	2
Dunlin	0	0	0	0	0	0	2	0	1	0	0	0	3
Purple sandpiper	0	0	0	0	0	0	0	0	0	0	0	1	1
Bar-tailed godwit	0	0	0	0	0	0	0	0	1	0	0	0	1
Curlew	0	0	0	0	0	0	0	0	6	0	1	0	7
Redshank	0	0	0	0	0	0	0	0	2	0	0	0	2
Sand martin	0	0	0	0	0	0	0	1	0	0	0	0	1
Swallow	0	0	0	0	0	0	0	1	0	0	1	0	2
Skylark	0	0	3	0	0	0	0	0	0	0	0	0	3
Meadow pipit	0	1	0	0	0	18	0	0	0	0	11	1	31
Blackbird	2	0	0	0	0	0	0	0	0	0	0	0	2
Starling	0	0	0	0	0	0	0	0	1	0	0	3	4
Total numbers	**93**	**21**	**4**	**34**	**1**	**20**	**8**	**3**	**14**	**0**	**104**	**905**	**1,207**

Table A7 Numbers of non-seabird species recorded in the Neart na Gaoithe Study Area in Year 2

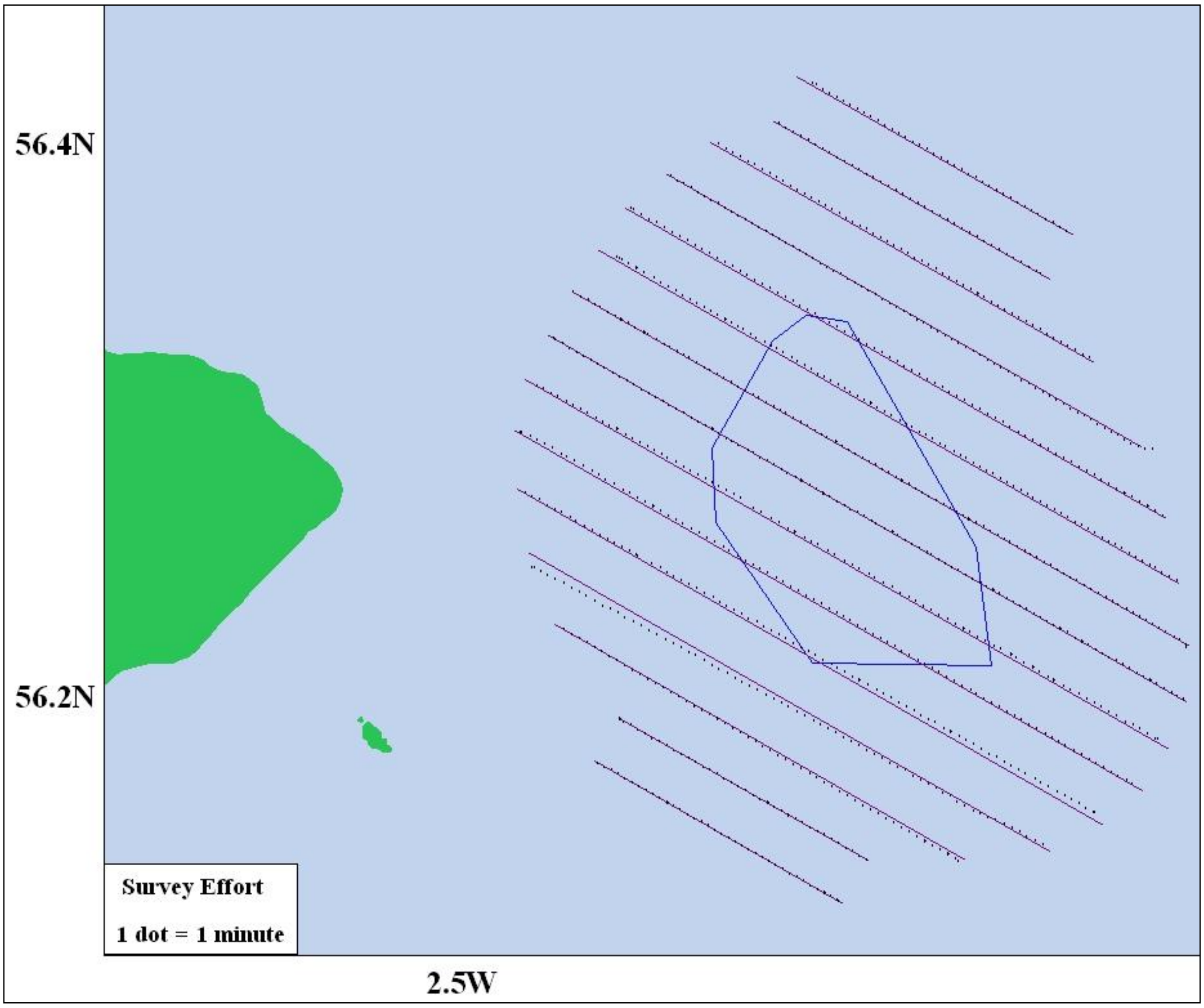
Species	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Total
Pink-footed goose	0	0	300	0	33	0	0	0	0	0	0	0	333
Wigeon	0	0	0	0	5	0	0	0	0	0	0	0	5
Tufted duck	0	0	0	0	0	0	0	0	0	0	0	1	1
Merlin	0	0	0	0	0	0	0	0	0	0	0	1	1
Oystercatcher	0	0	0	0	0	0	0	0	0	0	2	0	2
Ringed plover	0	0	0	0	0	0	0	0	0	0	0	1	1
Golden plover	0	0	0	0	0	0	0	0	0	0	5	15	20
Little stint	0	0	0	0	0	0	0	0	1	0	0	0	1
Dunlin	0	4	0	0	0	0	0	0	0	0	1	0	5
Curlew	0	0	0	0	0	0	0	0	1	0	0	0	1
Redshank	0	2	0	0	0	0	0	0	0	0	0	0	2
Turnstone	0	0	0	0	0	0	0	0	0	2	0	0	2
Short-eared owl	0	0	0	0	0	0	0	0	0	0	0	1	1
Swallow	0	0	0	0	0	0	1	0	0	0	1	0	2
Meadow pipit	0	0	0	0	1	9	0	0	0	0	17	11	38
Robin	0	0	0	0	0	0	0	0	0	0	0	2	2
Bluethroat	0	0	0	0	0	0	1	0	0	0	0	0	1
Wheatear	0	0	0	0	0	1	0	0	0	0	0	1	2
Fieldfare	0	0	1	0	0	0	0	0	0	0	0	0	1
Song thrush	0	0	0	0	0	0	0	0	0	0	0	1	1
Barred warbler	0	0	0	0	0	0	0	0	0	0	0	1	1
Carrion crow	0	0	0	0	0	0	1	0	0	0	0	0	1
Passerine species	0	0	0	0	0	0	0	0	0	0	0	1	1
Total numbers	**0**	**6**	**301**	**0**	**39**	**10**	**3**	**0**	**2**	**2**	**26**	**36**	**425**

November 2009 Survey Tracks



Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

December 2009 Survey Tracks



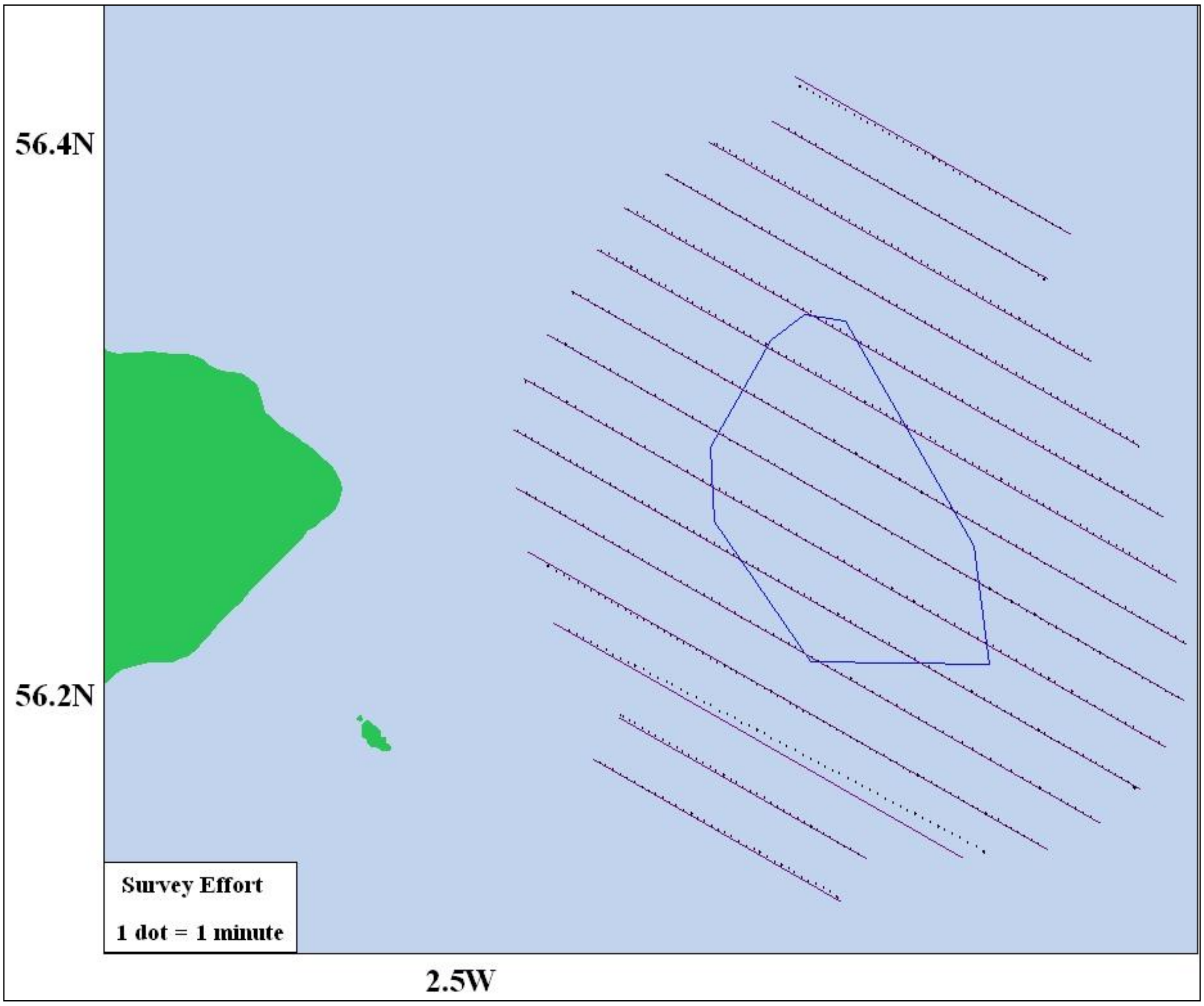
Key

Solid line – Target transect line

Dotted line – Actual vessel route

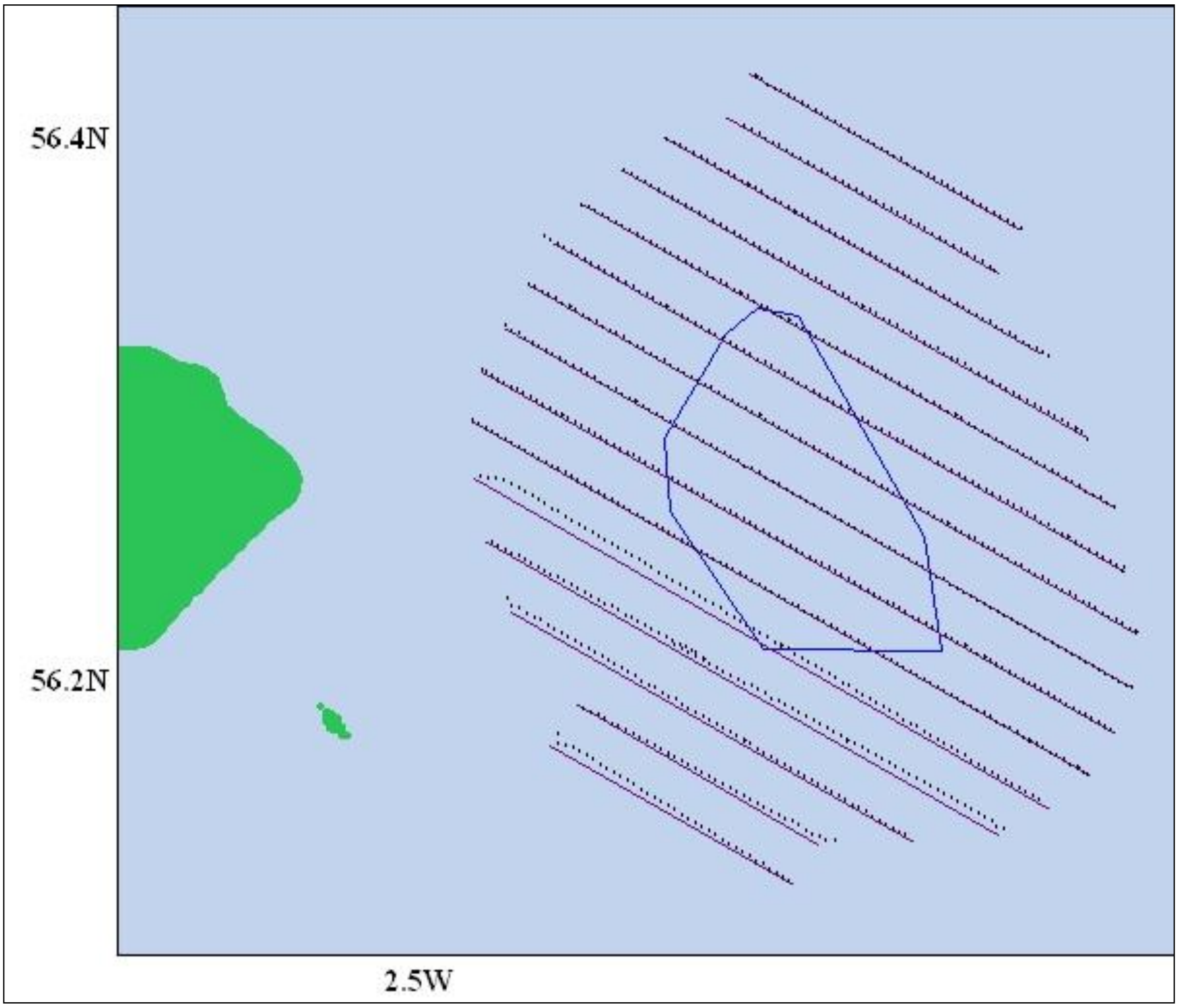
Blue line – Neart na Gaoithe development site

January 2010 Survey Tracks



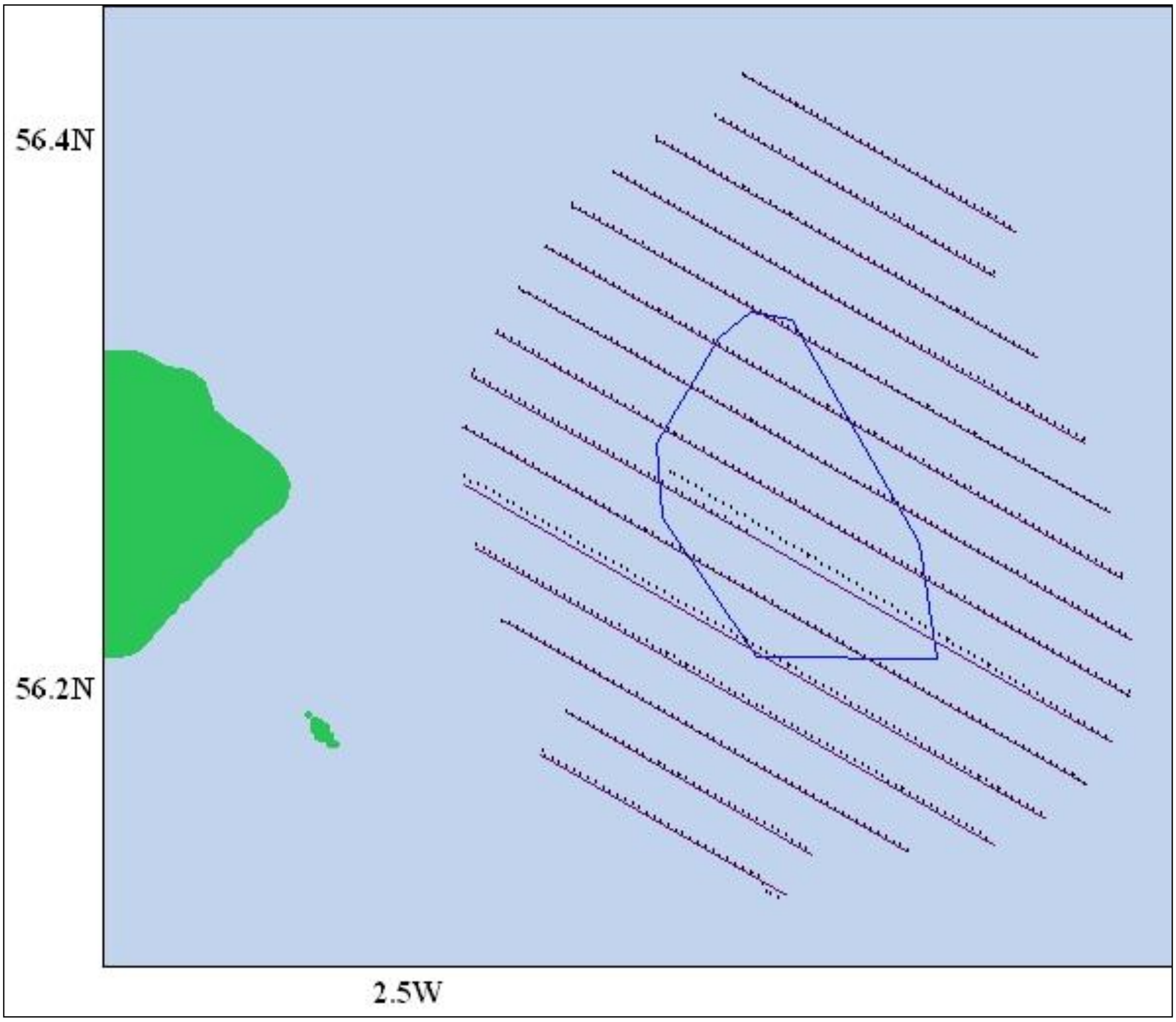
Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

February 2010 Survey Tracks



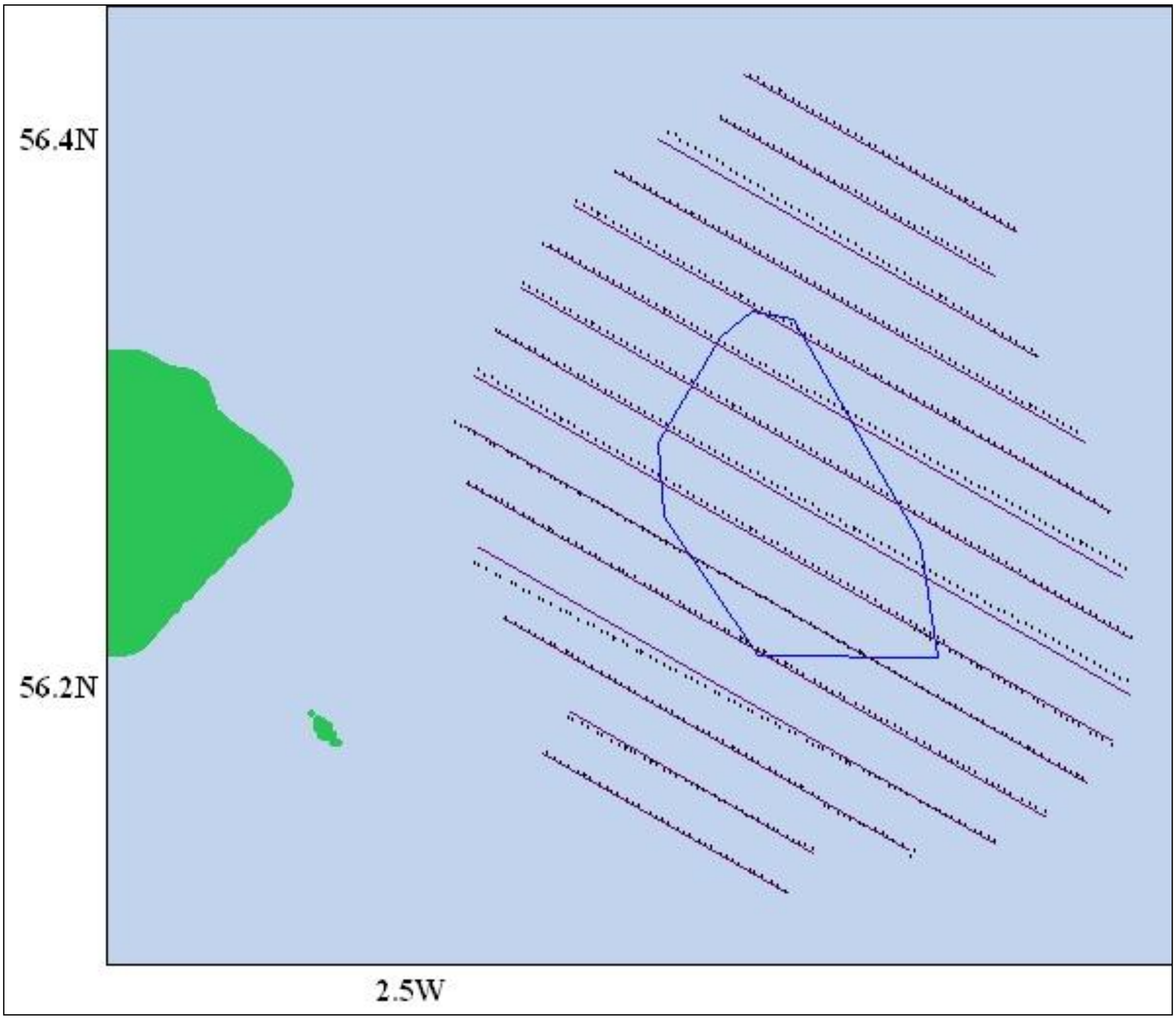
Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

March 2010 Survey Tracks



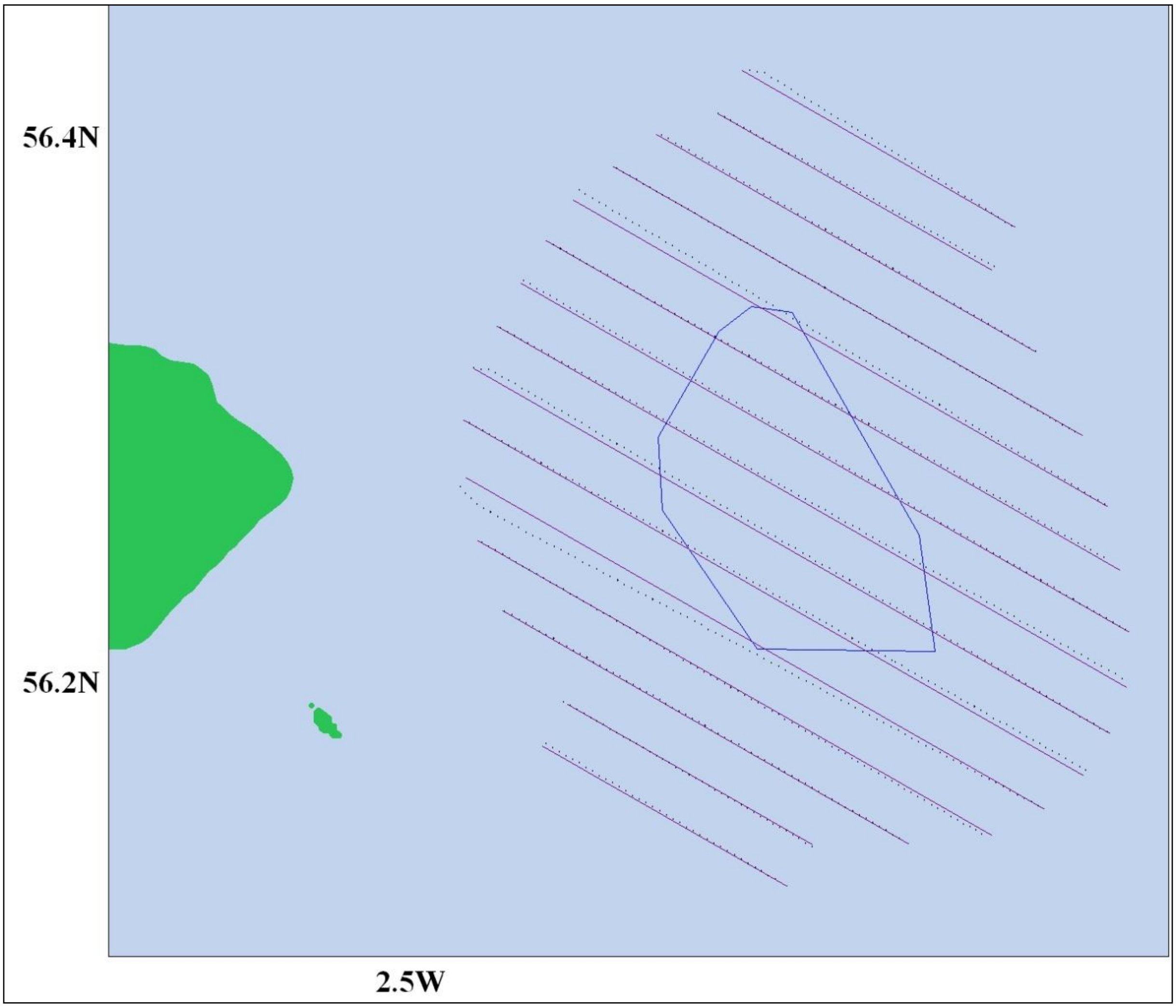
Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

April 2010 Survey Tracks



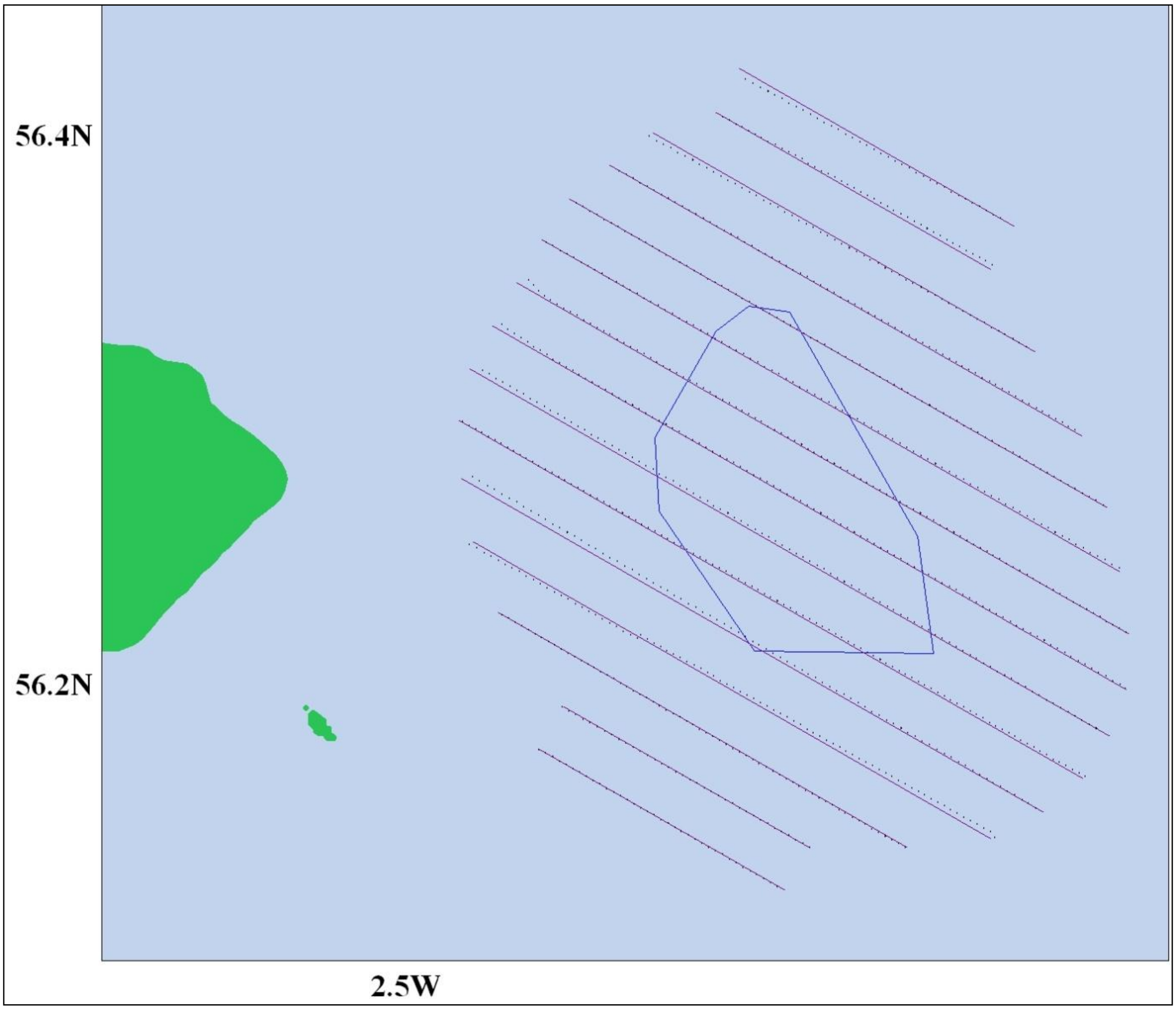
Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

May 2010 Survey Tracks



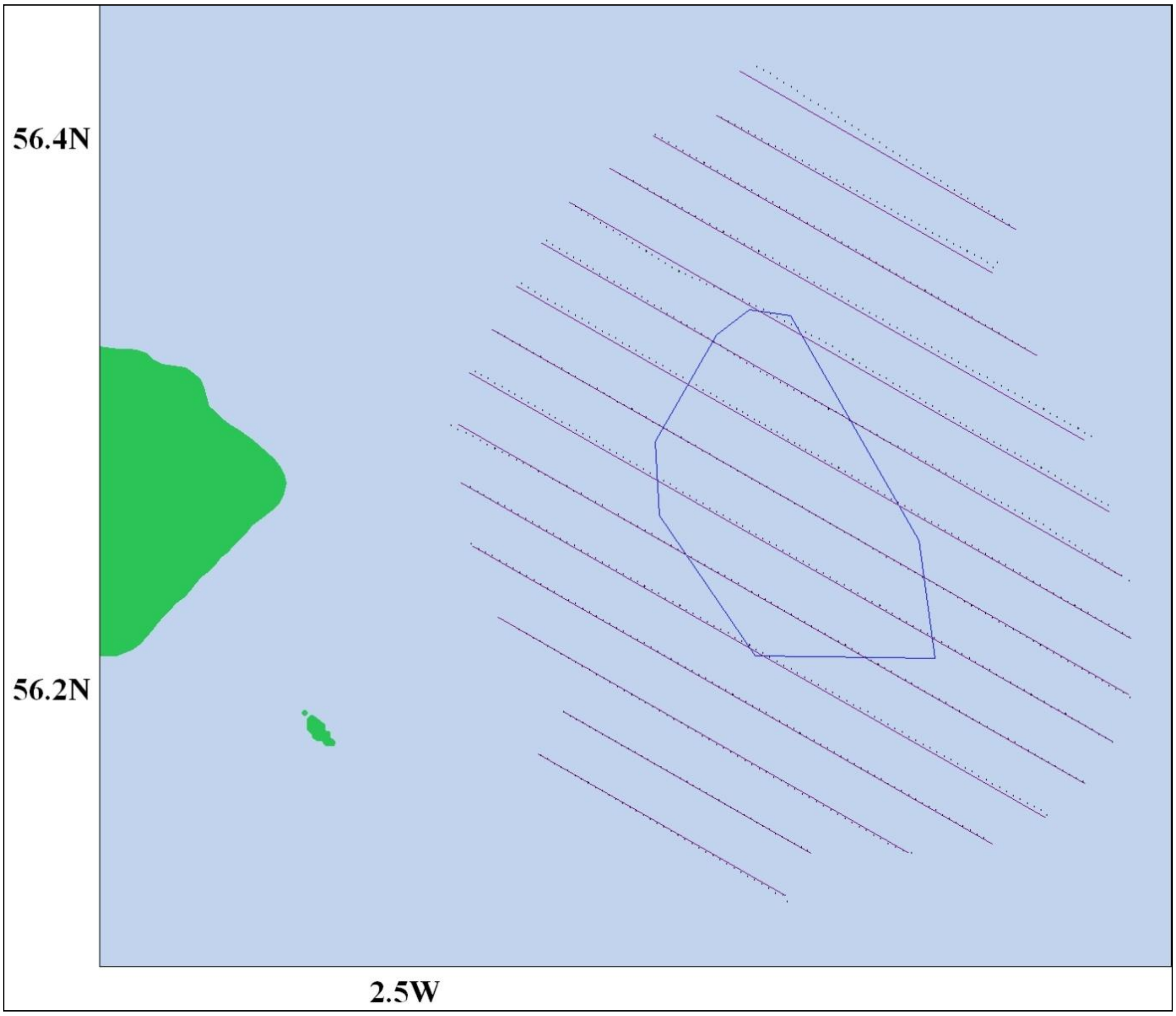
Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

June 2010 Survey Tracks



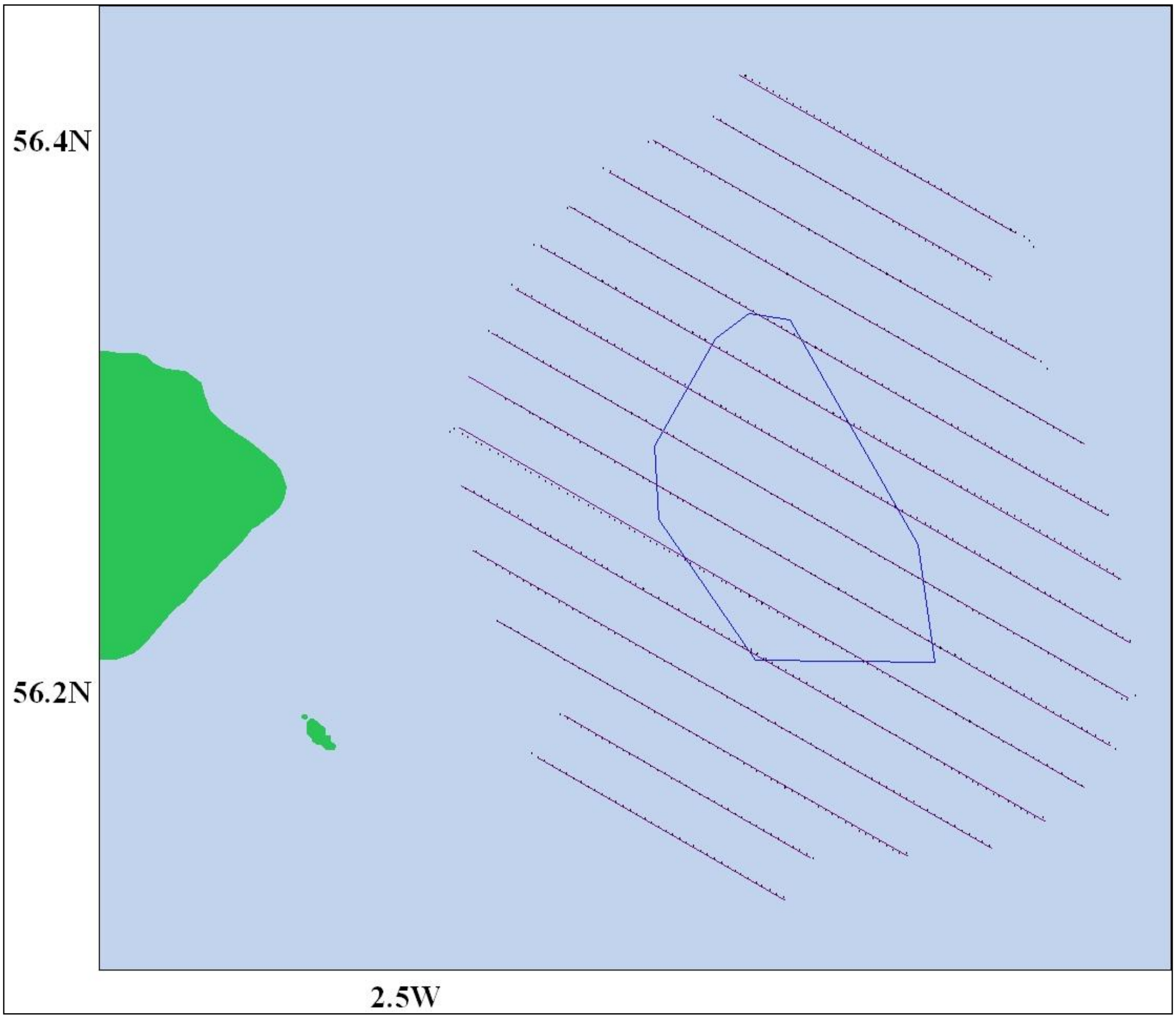
Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

July 2010 Survey Tracks



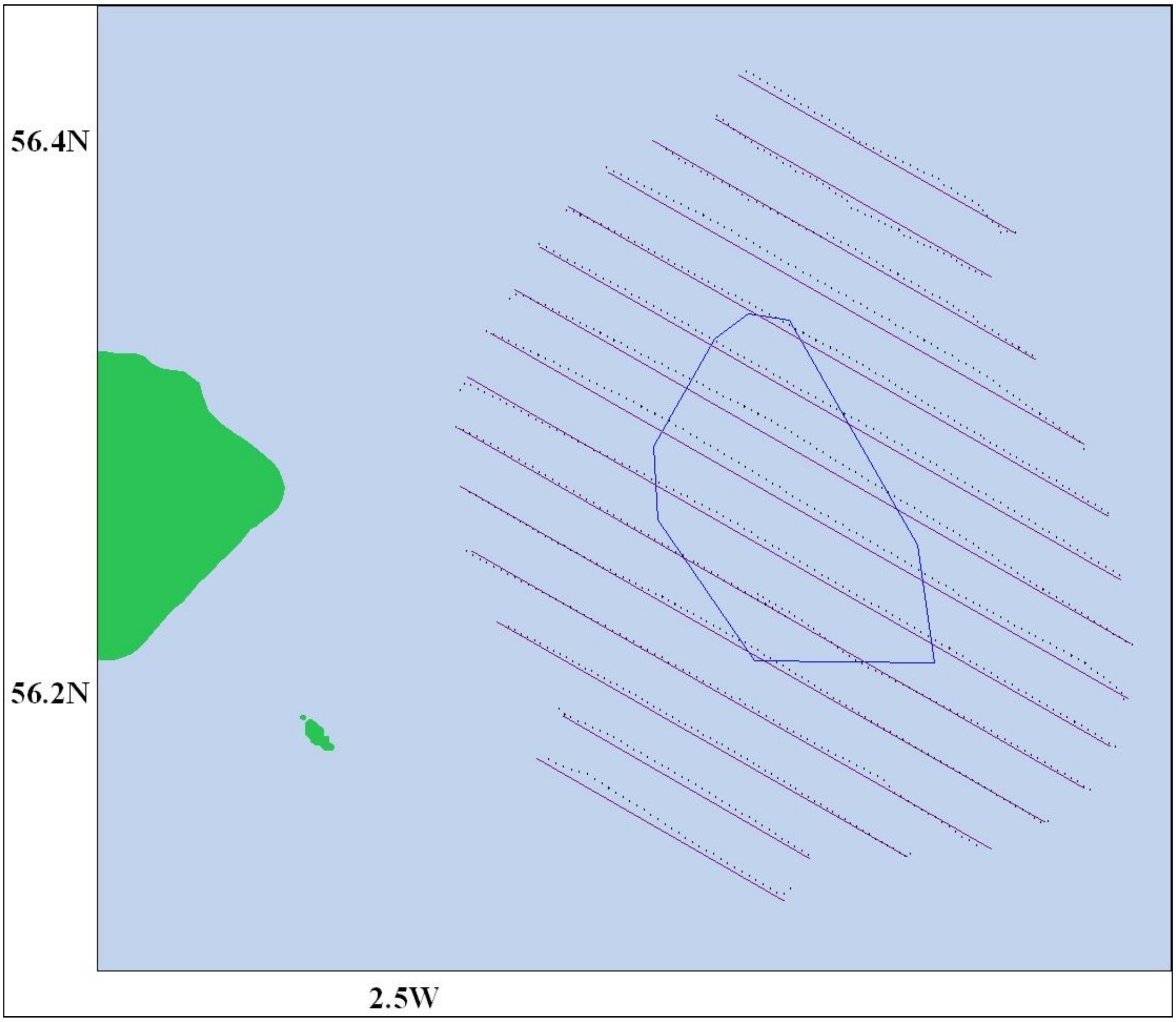
Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

August 2010 Survey Tracks



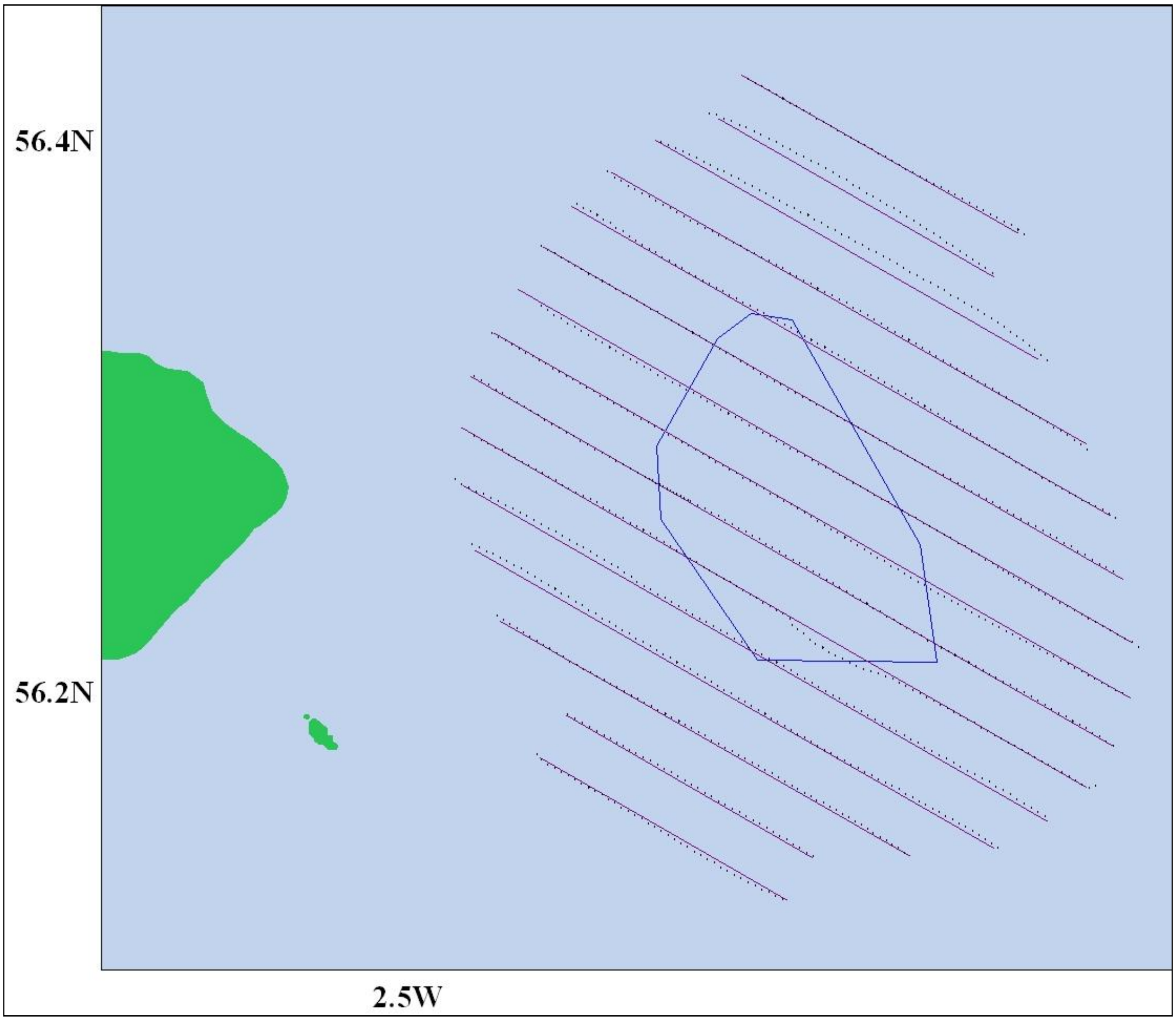
Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

September 2010 Survey Tracks



Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

October 2010 Survey Tracks



Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

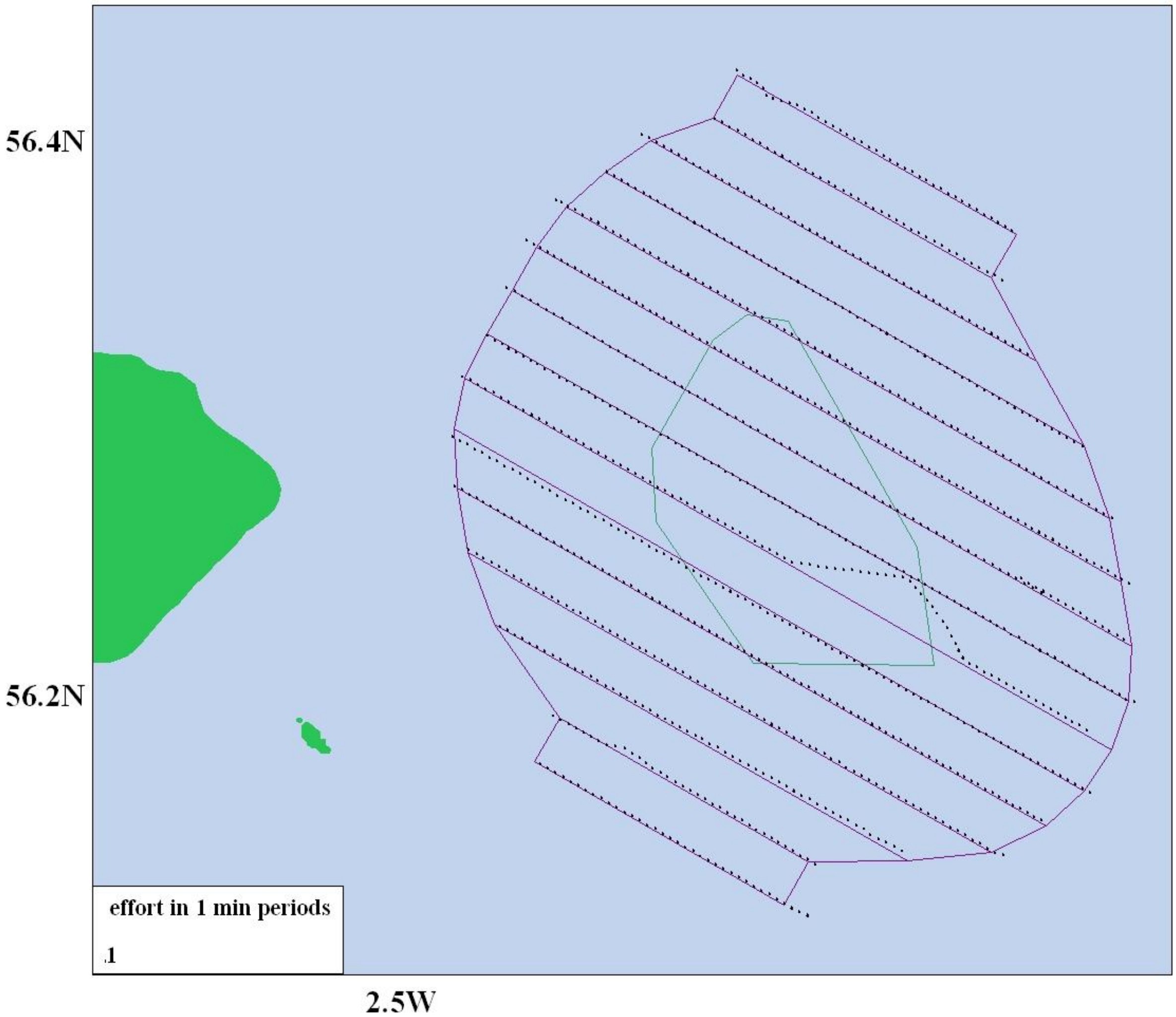
December 2010 Survey Tracks

Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

January 2011 Survey Tracks

Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

February 2011 Survey Tracks



Key

Solid line – Target transect line

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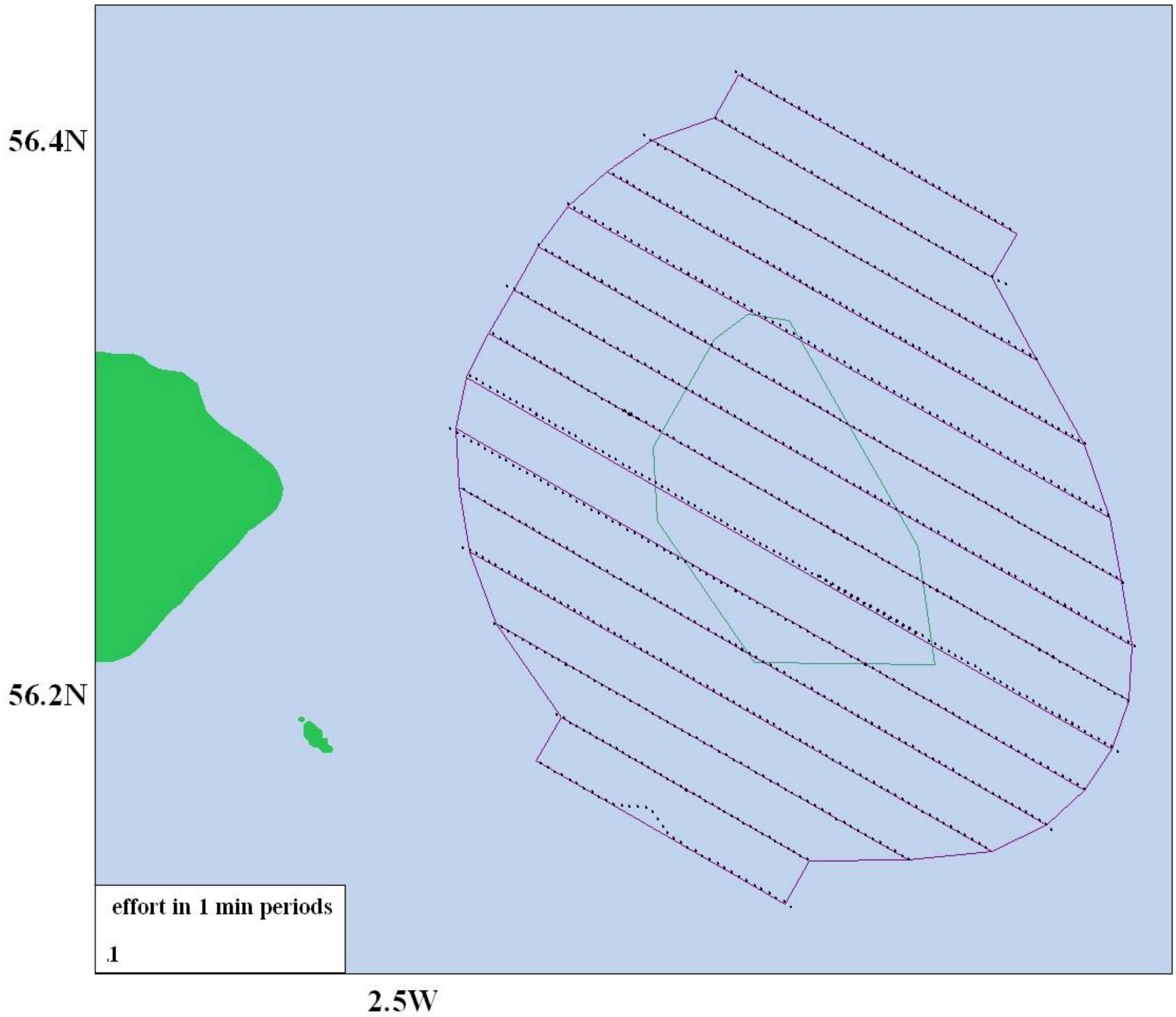
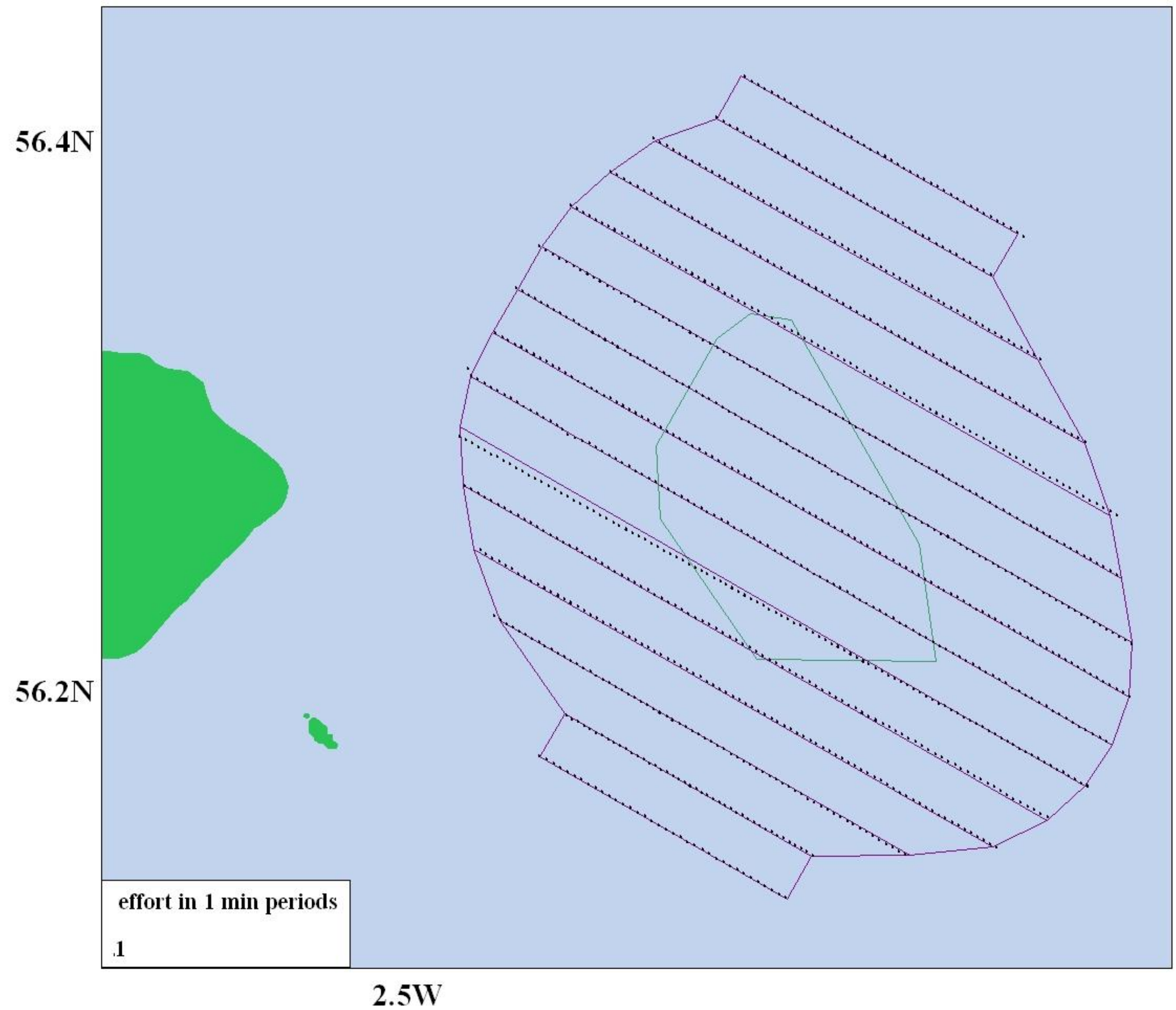
March 2011 Survey Tracks



Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

April 2011 Survey Tracks



Key
Solid line – Target transect line
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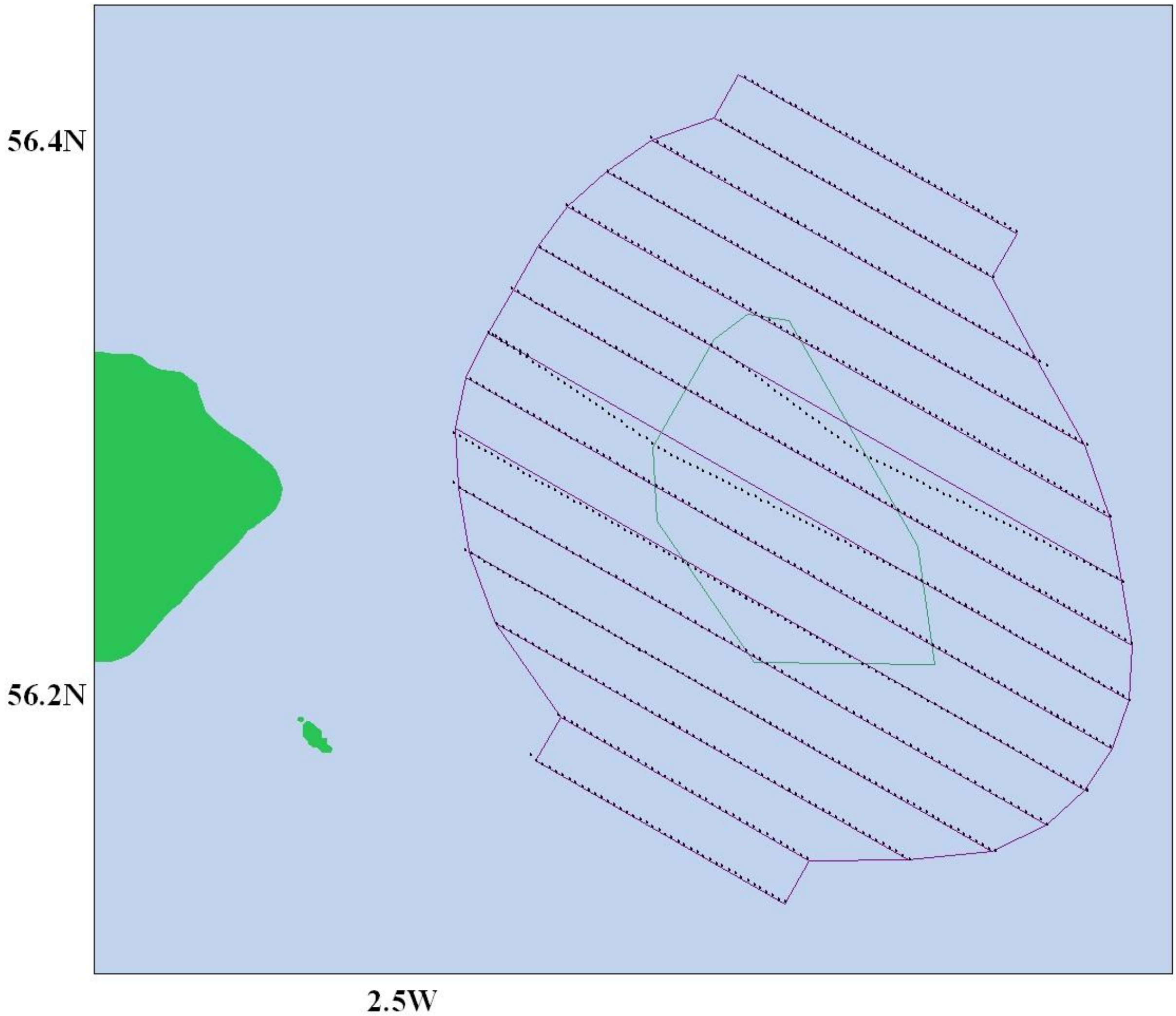
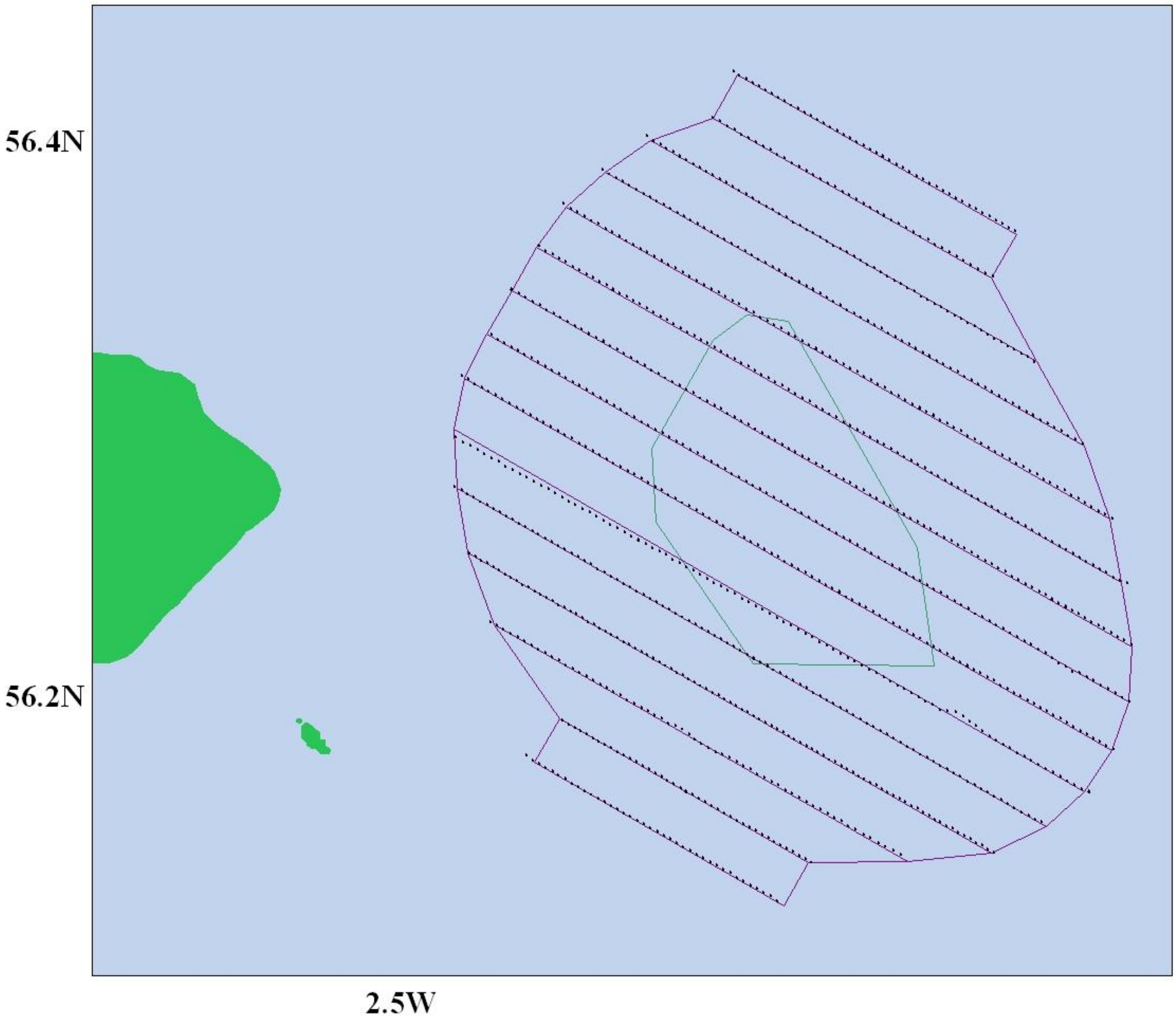
May 2011 Survey Tracks



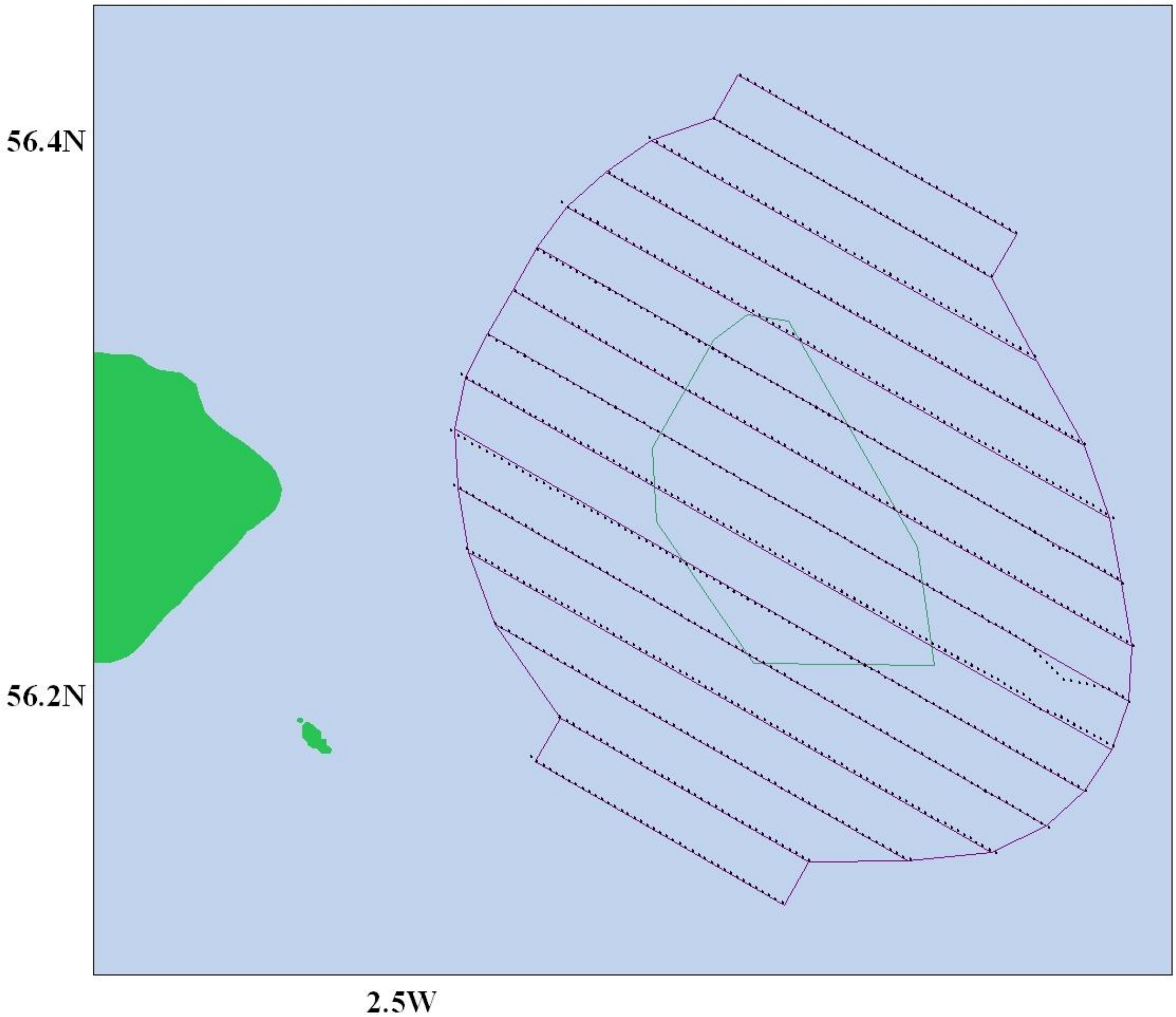
Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

June 2011 Survey Tracks



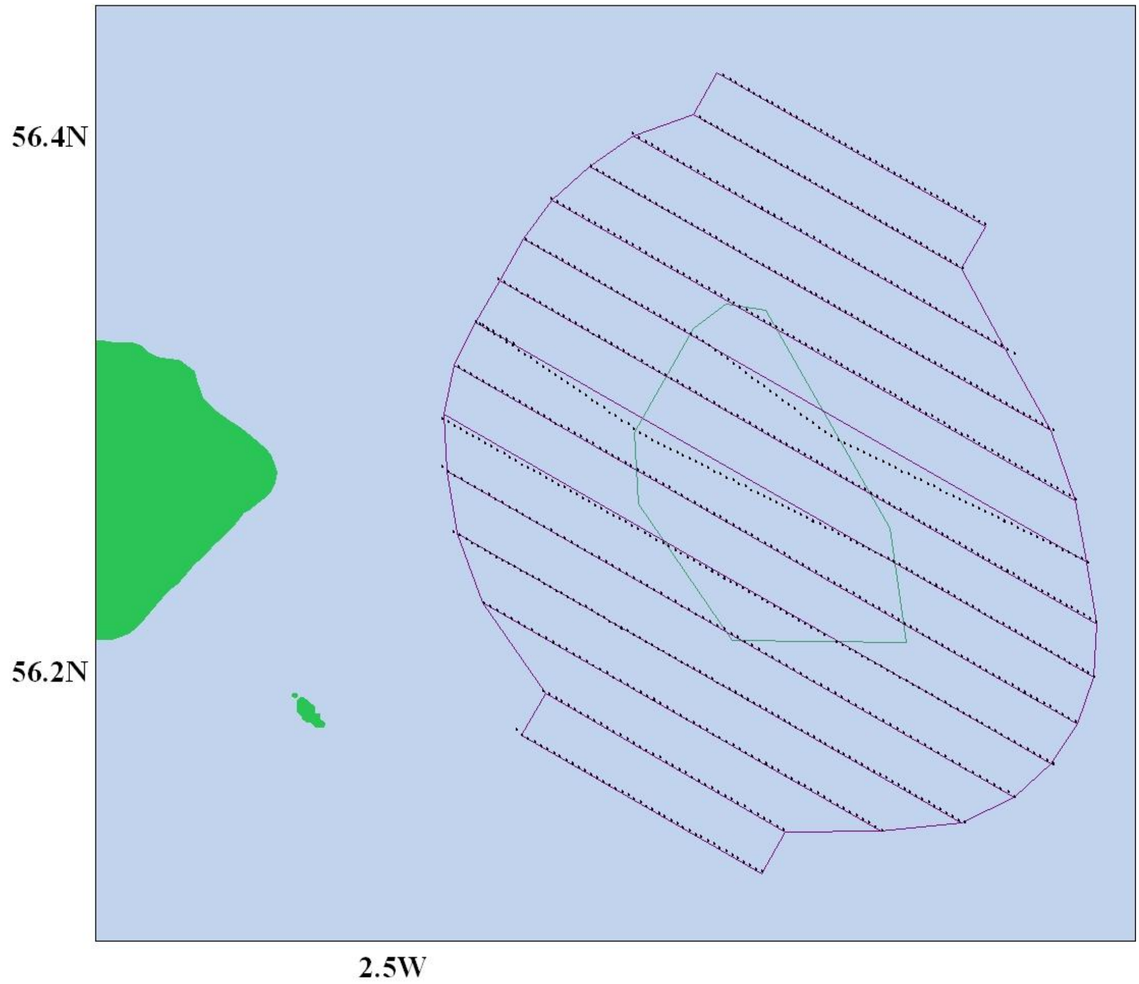
Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

July 2011 Survey Tracks



Key
Solid line – Target transect line
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

August 2011 Survey Tracks



Key
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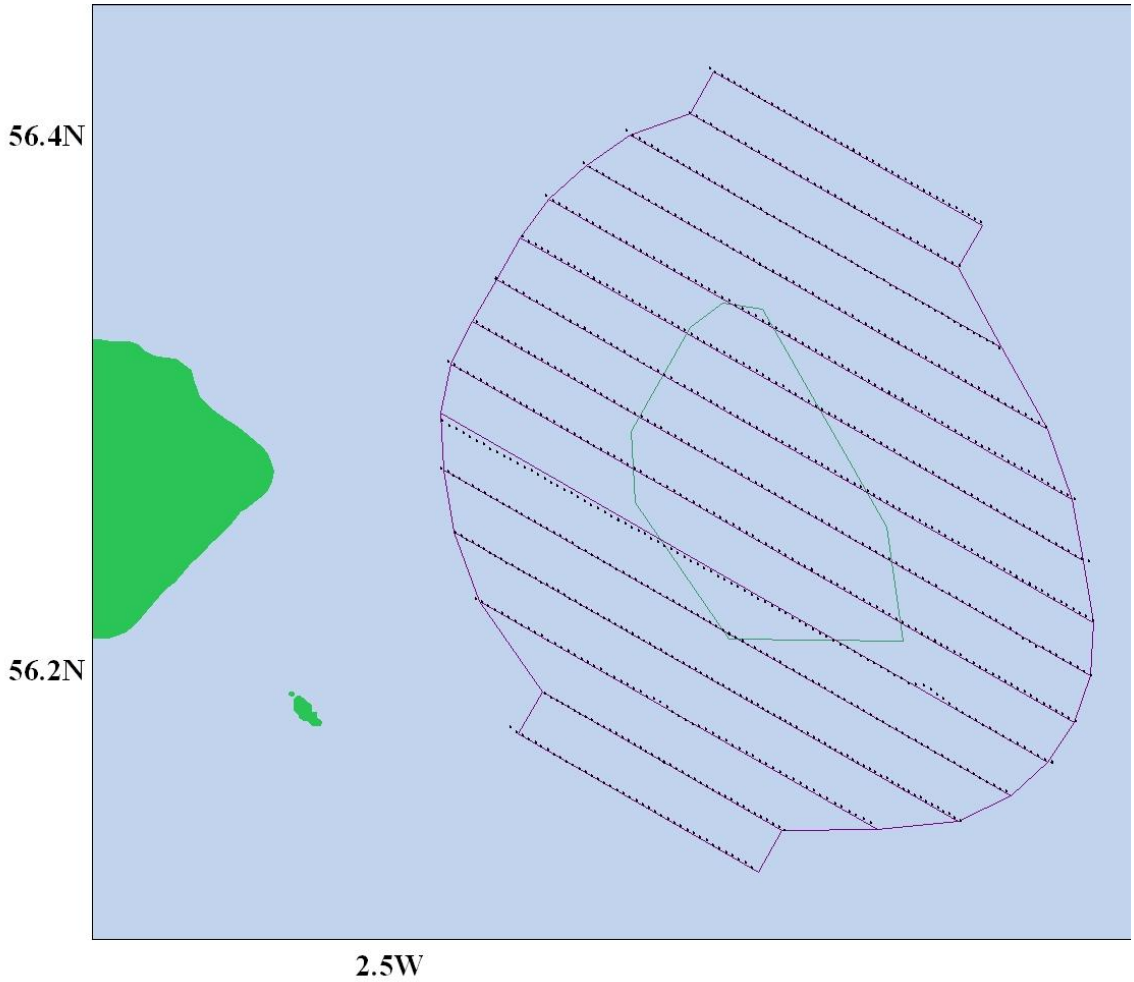
Dotted line – Actual vessel route
Blue line – Neart na Gaoithe development site

September 2011 Survey Tracks



Key
Solid line – Target transect line
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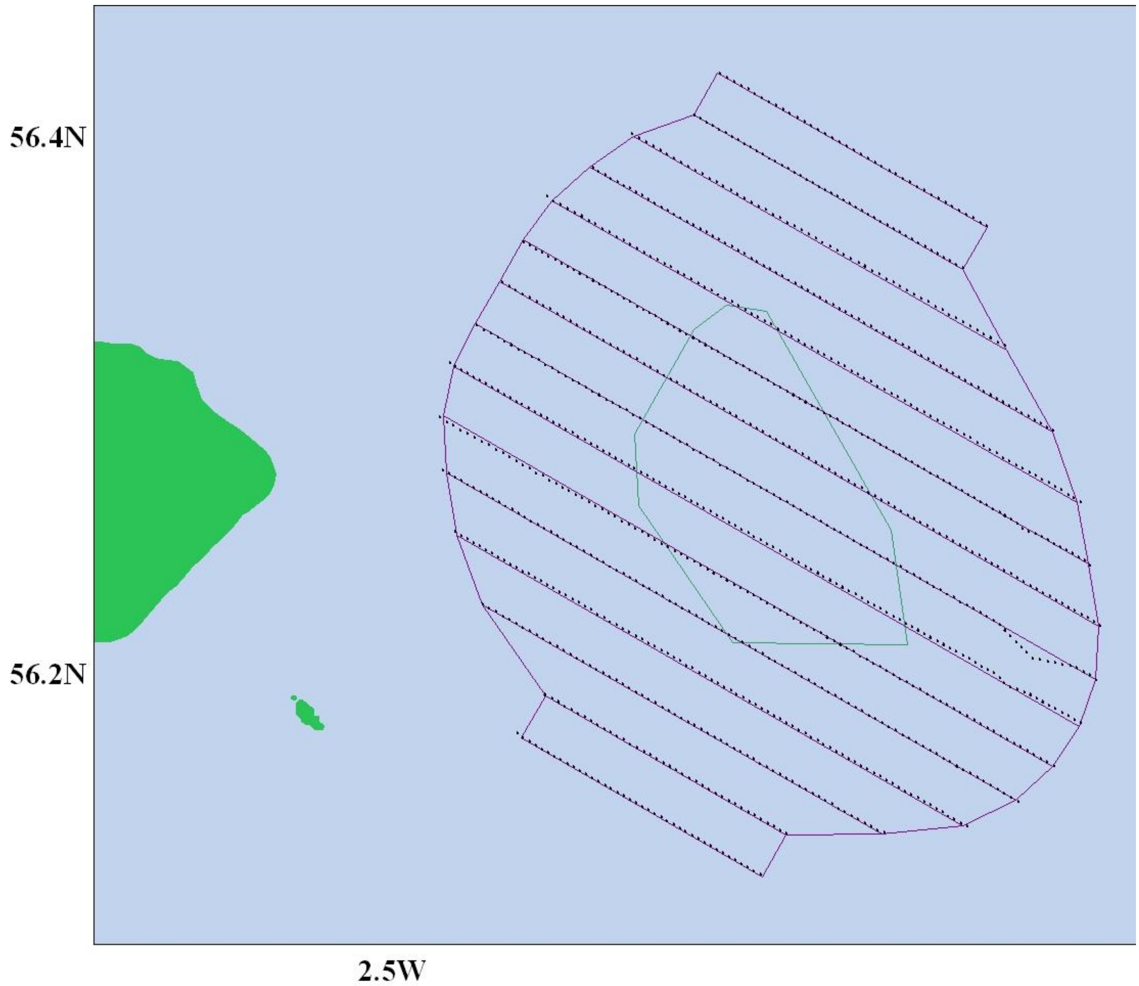
Blue line – Neart na Gaoithe development site

October 2011 Survey Tracks



Key
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