Mainstream Neart na Gaoithe

Offshore Wind Farm

Ornithology Technical Report

June 2013

CORK ECOLOGY

Long Strand, Castlefreke, Clonakilty, Co. Cork, Ireland

Based upon

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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, additional data from a third year of baseline surveys is also presented.

An environmental impact assessment, based on results from Years 1 to 3 is also presented.

This Technical Report is in support of the Neart na Gaoithe Addendum of Supplementary Environmental Information. Ornithology Appendix 2: Collision Rate Estimates of Seabirds at Neart na Gaoithe presents detailed information about Distance sampling methods and collision risk modelling techniques used in the impact assessment for the Neart na Gaoithe Addendum of Supplementary Environmental Information.

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. This was surveyed monthly using the European Seabirds at Sea (ESAS) boat-based survey method (Camphuysen *et al*., 2004) from November 2009 to October 2012.

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 the 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.

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The aims of this report are to support the Addendum of Supplementary Environmental Information. 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 report'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 Ornithology Appendix 3: Habitats Regulation Appraisal (Special Protection Areas). 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 surveys 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. 2012. Using a collision risk model to assess bird collision risks for offshore windfarms. Final version, August 2012. SOSS, The Crown Estate, 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 Design 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 Special Protection Area (SPAs) 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 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 (including 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 extensive 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 breeding 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 more distant, for example, Arctic terns and Arctic skuas 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 three 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 offshore site and 8 km buffer 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, and February of Year 3 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 IN Years 1 and 2*, which has a custom-built surveyor platform with an observer eye-height of greater than 5 m, as recommended for ESAS surveys (Webb & Durinck 1992, Camphuysen *et al.* 2004). In Year 3, surveys were conducted onboard *M.V. Eileen May*, which had a survey platform with a similar observer eye-height to the previous survey vessel.

Baseline 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, Jon Ford, Paul French, Jonathon Clarke, Bill Aspin, Phil Espin, Chris Rodger. All surveyors were ESAS-accredited.

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

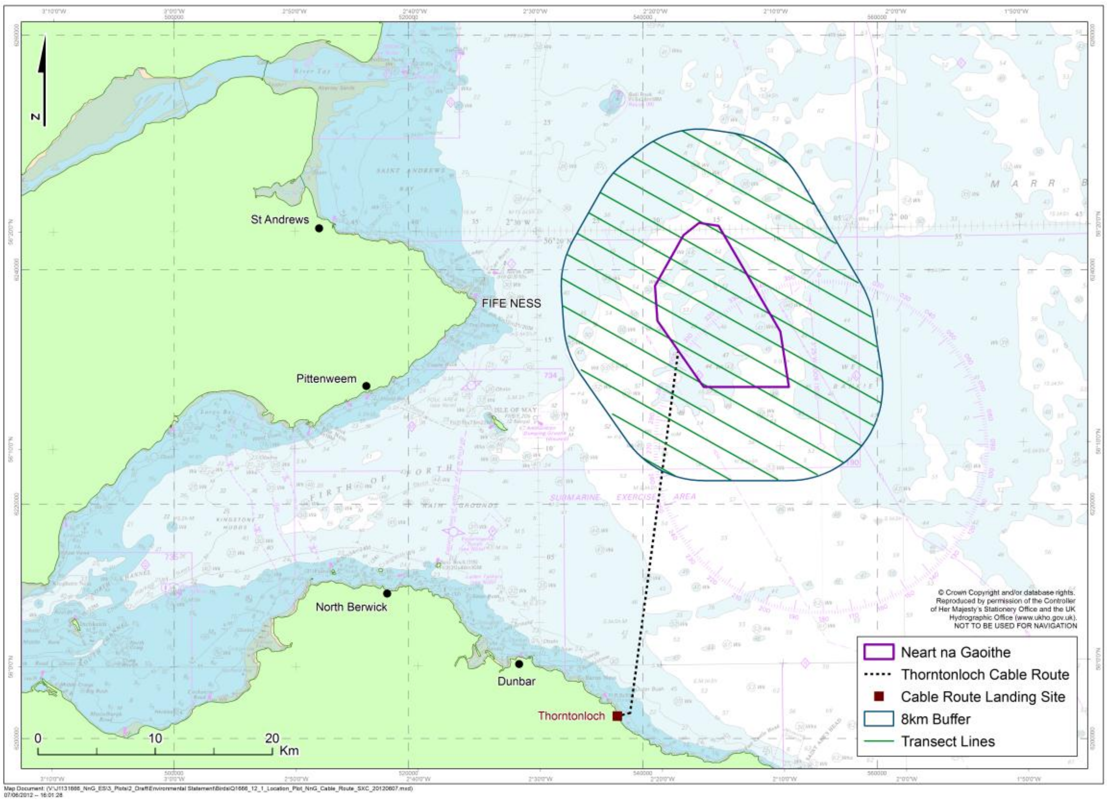


Figure 3.1 Offshore site and 8 km buffer area

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 Ornithology Appendix 2: Collision Rate Estimates of Seabirds at Neart na Gaoithe.

In addition, collision risk modelling was also carried out to inform this assessment. Details of this work are also presented in Ornithology Appendix 2: Collision Rate Estimates of Seabirds at Neart na Gaoithe.

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 were 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 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 the Forth and Tay Offshore Wind Developers Group (FTOWDG) 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, 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 published 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. Similarly, seabirds that are temporarily flightless whilst undergoing their annual wing moult e.g. auks, 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 (NCI) 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 individual 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, when considering 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, possibly 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 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 adult males are attending their dependent young on the sea; this was defined as July and August (based on Cramp and Simmons (1983), and the dates when dependent young were recorded on baseline 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 peak number of birds present in the period under consideration for each baseline survey year was used, and the mean calculated by dividing by three.

3.4.4 Estimation of Potential Collision Mortality

For various bird species, the level of collision-related mortality was estimated by collision rate modelling. The model used was that published by the Crown Estate Strategic Ornithological Support Services group in August 2012 (Band 2012; available from http://www.bto.org/science/wetland-and-marine/soss/projects). This model, based on the SNH/Band collision risk model (Band et al. 2007; available from http://www.snh.gov.uk/planning-and-development/renewable-energy/onshore-wind/assessing-bird-collision-risks/), has been extended to allow the direct input of density data and to allow the comparison of various avoidance rates on the estimated collision rates.

Based on the physical characteristics of both the turbine and species of bird, a turbine/species-specific probability of collision for a single bird crossing the rotor-swept area can be calculated. This probability is then applied to the number of birds crossing the rotor-swept area of an entire wind farm, which is estimated based on the density of flying birds and the size and number of turbines. Finally, an avoidance factor is applied that accounts for birds avoiding turbines, were they present.

Within this model, Option 1 (Basic model) was used to estimate the Overall Collision Risk.

Full details of the methods used to estimate collisions are presented in Ornithology Appendix 2: Collision Rate Estimates of Seabirds at Neart na Gaoithe.

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.

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 estimated the number of potential bird collisions per season for four wind turbine designs scenarios (Table 3.6).

Table 3.6 Wind turbine design scenarios

Scenarios	No of turbines	Megawatts	Rotation speed (rpm)	Rotor radius (m)	Maximum blade width (m)
Option 1	90	5 MW	10.2	67.5	4.8
Option 2	75	6 MW	8	77	5
Option 3	73	6.15 MW A	9.9	63	4.5
Option 4	73	6.15 MW B	8.25	76	4.5

Although Options 3 and 4 have the same number and same size of turbines, Option 3 has a narrower rotor radius and a slightly faster rotation speed than Option 4 (see Ornithology Appendix 2: Collision Rate Estimates of Seabirds at Neart na Gaoithe).

Results for all four design scenarios are considered in the assessments.

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 a 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. These species were selected from Cook et al (2012).

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 precautionary.

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.

The assessment of displacement effects was based on an Interim Advice Note from JNCC and Natural England (NE) (JNCC & NE, 2012).

Peak estimated numbers of birds in the offshore site per season (e.g. breeding season, post-breeding season and non-breeding season) for Years 1 to 3 were averaged to get the three-year mean peak per season. This was repeated for a 1 km and 2 km buffer around the offshore site.

As the displacement assessment is based on breeding adult birds, it was necessary to estimate the number of immature birds present during the breeding season, where possible. This was based on the ratio of immature to adult birds recorded on survey. For species where it was not possible to distinguish immature birds from adults (e.g. fulmar) it was assumed that all birds present in the breeding season were breeding birds. Outside the breeding season, all birds were assessed, regardless of age.

To calculate the mean percentage of immature birds in the breeding season (e.g. April to September), the mean percentage of immature birds per month was calculated by adding the number of immature birds in each month across the three years, and then dividing that by the total of aged birds per month, all years combined and multiplying by 100. The percentage for each month was then added, and this figure was divided by the number of months (in this example, six months).

This gave a mean percentage of immature birds in the breeding season, which was then subtracted from the three-year mean peak estimated number of gannets in the breeding season to get the estimated number of adults present.

The three-year mean peak estimated number of adult birds was then used to predict the estimated number of adult birds at potential risk of mortality following displacement by season in the offshore site (plus 1 km and 2 km buffers), as recommended in the draft guidance note on displacement (JNCC & NE, 2012).

The results were presented as a range of displacement and mortality values, and the most appropriate level for displacement and mortality were selected and discussed, based on available evidence from constructed offshore wind projects, tagging studies and expert opinion. Results were also compared against regional, national and international populations, as appropriate for the species and season.

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 some 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 is assessed here in terms of the effect it could have on the time and energy budget of foraging birds by causing them to make longer flights between their 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 3.7), 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 around the wind farm would be small compared to the overall length of the foraging trip.

Table 3.7 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 precautionary 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 3.7).

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 3.8). 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).

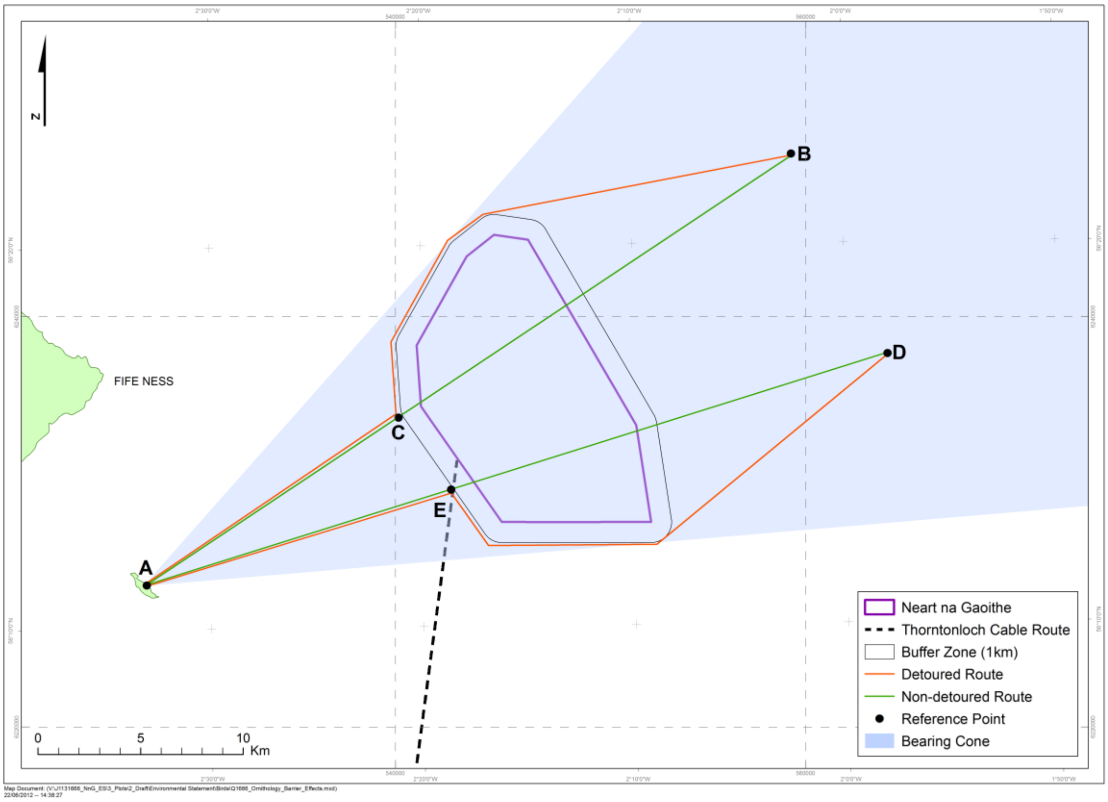


Figure 3.2 Schematic diagram showing how barrier effects were estimated

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 3.8 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%

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%

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

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 found 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 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.

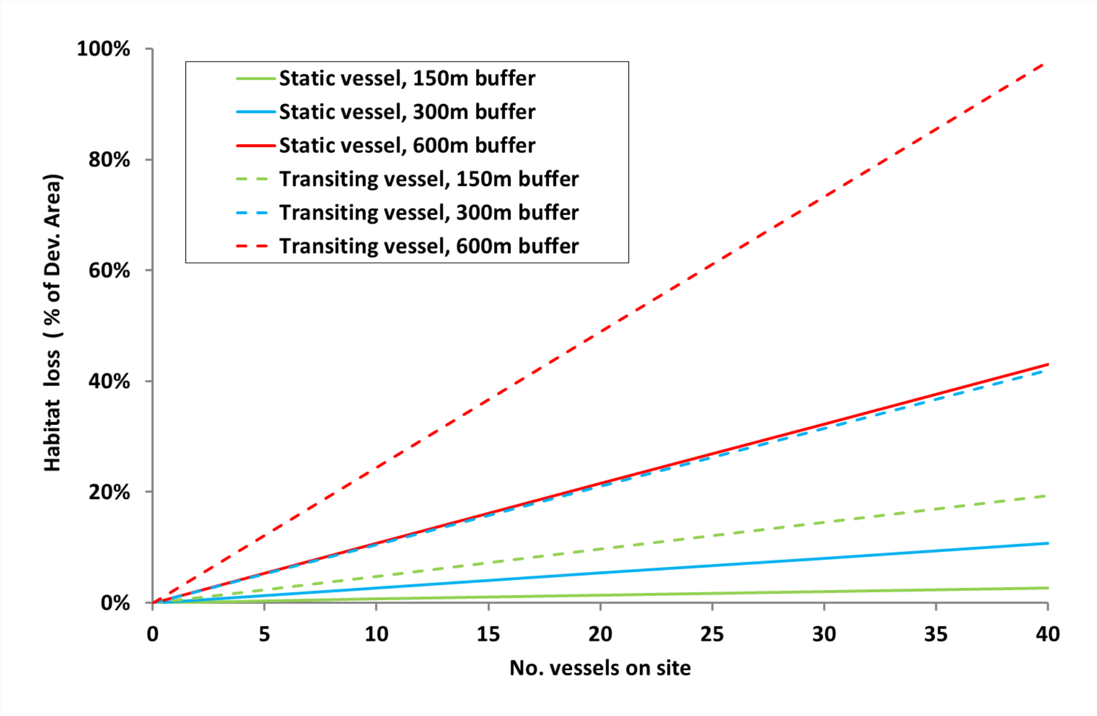


Figure 3.3 The hypothetical loss of seabird foraging habitat from the offshore site caused by disturbance from vessels

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 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 precautionary 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 three proposed offshore wind farms close to Neart na Gaoithe, in the wider Firth of Forth area. These are the Inch Cape Offshore Wind Farm (approximately 1,000 MW) and the Seagreen Project Alpha and Project Bravo Firth of Forth Round 3 Zone wind farms (approximately 3,700 MW). In addition, an application submitted for the proposed Aberdeen Offshore Wind Farm in Aberdeen Bay, approximately 100 km to the north of Neart na Gaoithe has recently been consented. This proposal is relatively small in scale (100 MW, 11 turbines). Information from the Environmental Statement states that the main species of concern were divers and scoter species (Technip UK 2012). 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 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.

Impacts on birds that might arise from the Inch Cape Offshore Limited development and Seagreen Projects Alpha and Bravo have been estimated using data provided in the Year One reports on baseline bird surveys for the Inch Cape projects (ICOL, 2012) and the Seagreen Project Alpha and Project Bravo ES (Seagreen 2012). In undertaking assessments of the available results for these projects various assumptions were required to overcome some information gaps, and these are highlighted in the accompanying text.

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 3,429.5 km surveyed. In Year 3, surveys were conducted over 32 days between November 2011 and October 2012, with a total of 3,237.2 km surveyed.

Complete coverage of both the offshore site and buffer area was achieved in all months in Year 1 (Table 4.1). In Year 2, there was no survey coverage in November due to bad weather, however full coverage was achieved in all other months. In Year 3, there was partial survey coverage in September and no survey coverage in December due to bad weather, however full coverage was achieved in all other months.

Table 4.1 Survey effort in the offshore site and 8 km buffer area in Years 1 to 3

Month	Offshore Site Km travelled			Buffer Area Km travelled			Proportion target coverage [1]		
	Yr 1	Yr 2	Yr 3	Yr 1	Yr 2	Yr 3	Yr 1	Yr 2	Yr 3
November	54.4	0	52.4	257.1	0	252.2	99.4%	0%	96.7%
December	54.7	54.9	0	254.7	246.5	0	98.7%	96.3%	0%
January	54.0	53.5	50.3	256.5	256.9	253.8	98.6%	98.5%	96.5%
February	53.9	55.0	52.4	259.7	257.0	253.9	99.6%	100.1%	97.2%
March	56.7	58.7	52.4	258.3	259.0	254.6	100%	101.7%	97.5%
April	51.9	55.0	52.6	258.3	256.0	255.2	99.5%	99.8%	97.7%
May	51.0	55.2	51.8	259.6	259.4	255.6	99.5%	100.4%	97.6%
June	55.1	56.6	52.4	256.8	256.0	254.3	99.2%	99.9%	97.4%
July	52.4	55.7	51.9	256.2	257.4	256.5	99.7%	100.1%	97.9%
August	48.2	52.8	51.9	263.6	258.3	270.8	100.1%	99.7%	102.4%
September	50.5	53.4	26.2	260.0	261.3	125.9	99.6%	100.8%	48.3%
October	48.7	52.2	51.6	262.3	258.8	256.7	100.1%	100.9%	97.9%
Total	**631.5**	**603.0**	**545.9**	**3,103.1**	**2,826.6**	**2,689.5**	**98.8%**	**90.7%**	**85.6%**

1 Although full coverage was achieved in all months except in November of Year 2 and September and December of Year 3, 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 (99.3%) were collected in Sea States 0 to 4, with only 0.7% conducted in Sea State 5 (Figure 4.1, Figure 4.2 & Figure 4.3). This data was excluded from further analyses.

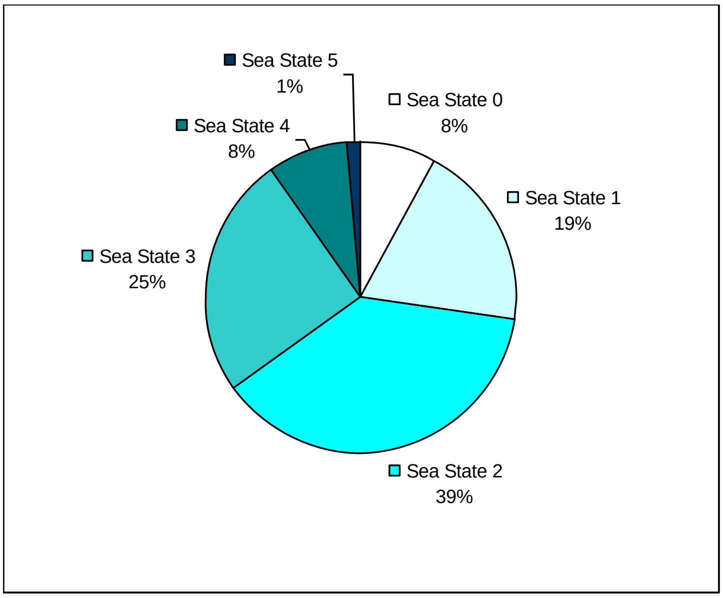


Figure 4.1 Survey effort in the offshore site and 8 km buffer area in relation to sea state during Year 1

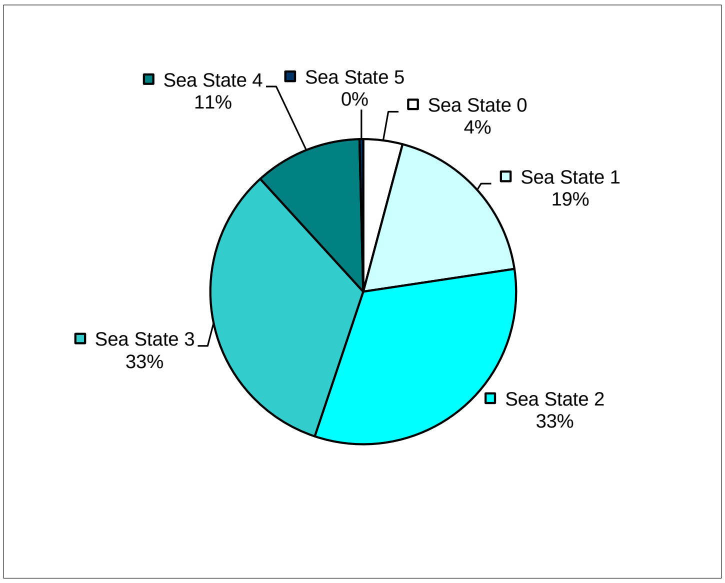


Figure 4.2 Survey effort in the offshore site and 8 km buffer area in relation to sea state during Year 2

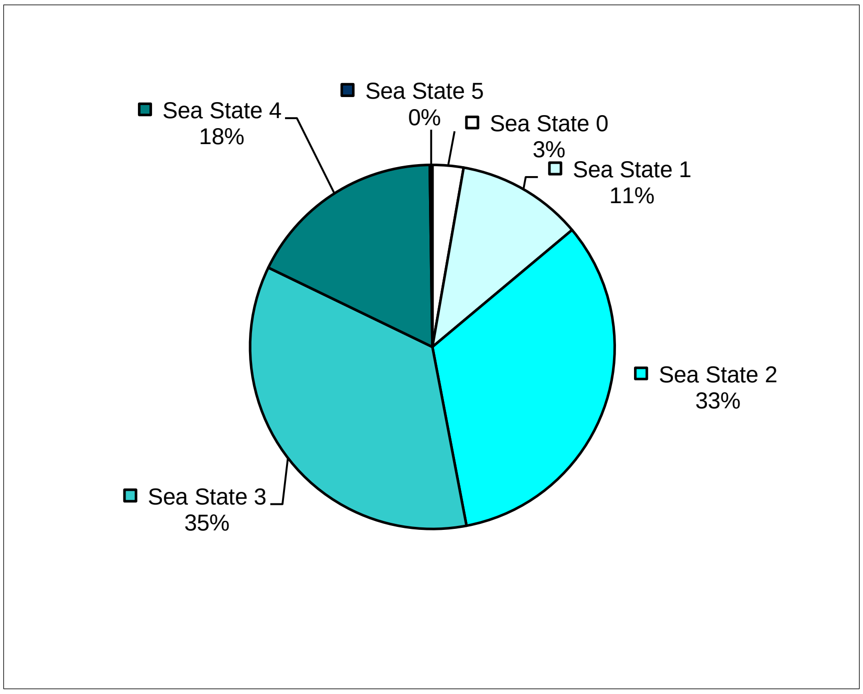


Figure 4.3 Survey effort in the offshore site and 8 km buffer area in relation to sea state during Year 3

The monthly breakdown of survey effort in relation to sea state for Years 1 to 3 is presented in Annex 1.

4.2 Raw numbers of seabirds in the offshore site and 8 km buffer area in Years 1 to 3

A total of 29 seabird species were identified on surveys in the offshore site and 8 km buffer area in Year 1 (November 2009 to October 2010) (Table 4.2). In Year 2, 26 seabird species were recorded in the offshore site and 8 km buffer area (November 2010 to October 2011). In Year 3, 28 seabird species were recorded in the offshore site and 8 km buffer area (November 2011 to October 2012).

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. In Year 3, 17 species were recorded in the offshore site, with gannet, guillemot and puffin again the three most frequently recorded species. These three species accounted for 72.9% of all birds recorded.

25 species were recorded in the buffer area in both Years 1 and 2, with 27 species recorded in Year 3 (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. In Year 3, 28 species were recorded in the buffer area, with gannet, guillemot and puffin accounting for 73.2% of all birds recorded.

Table 4.2 Comparison of seabird numbers in the offshore site and total Study Area in Years 1 to 3 (Raw numbers, all sea states)

Species	Year 1		Year 2		Year 3	
	Offshore Site	Study Area	Offshore Site	Study Area	Offshore Site	Study Area
Red-throated diver	0	5	0	0	0	0
Fulmar	112	692	189	1,116	87	491
Sooty shearwater	84	227	4	179	1	12
Manx shearwater	16	72	27	286	27	179
Balearic shearwater	1	1	0	0	0	1
Storm petrel	0	1	0	0	0	1
Gannet	1,649	13,021	3,122	19,416	2,134	14,825
Cormorant	0	1	0	3	0	3
Shag	0	11	0	6	0	3
Eider	9	20	0	2	0	7
Common scoter	5	5	0	2	0	1
Red-necked phalarope	0	0	0	1	0	0
Grey phalarope	1	1	0	2	0	3
Pomarine skua	0	6	0	0	3	22
Arctic skua	0	6	0	18	0	5
Great skua	1	24	0	16	2	20
Little gull	32	298	6	220	43	422
Sabine’s gull	1	1	0	1	0	0
Black-headed gull	0	27	0	11	0	1
Common gull	6	78	12	52	2	22
Lesser black-backed gull	10	66	11	195	37	171
Herring gull	50	1,723	58	1,433	54	800
Great black-backed gull	25	528	20	434	17	225
Large gull species	4	162	1	348	82	716
Kittiwake	801	3,955	719	4,123	838	4,300
Small gull species	0	0	0	1	0	0
Common tern	3	13	13	50	0	1
Arctic tern	205	877	37	329	90	549
Common/Arctic tern	1	76	28	195	0	6
Unidentified tern species	0	34	0	0	0	0
Guillemot	1,252	7,898	1,544	11,730	1,769	11,557
Razorbill	596	3,980	350	3,131	278	1,915
Little auk	26	135	16	113	415	2,710
Puffin	1,306	11,199	1,110	6,622	1,196	5,983
Puffin/little auk	0	3	0	0	0	0
Guillemot/razorbill	368	3,323	168	1,532	213	1,767
Unidentified auk species	155	1,348	56	827	7	186
Total numbers	**6,719**	**49,817**	**7,491**	**52,394**	**7,295**	**46,904**

Monthly summary tables are presented in Annex 1.

4.3 Flight height of birds

Information on the height of flying birds in Years 1 to 3 combined (November 2009 to October 2012) 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, 95.5% of all flying birds on baseline surveys were recorded flying below 27.5 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 27.5 m in height.

For other seabirds, a greater proportion of birds were recorded flying above 27.5 m i.e. in the wind turbine rotor swept zone, for example 4.8% of kittiwakes (n=6,945), 4.8% of gannets (n= 41,250), 9.2% of lesser black-backed gulls (n=358), 19.3% of great black-backed gulls (n=553) and 21.7% of herring gulls (n=1,646) were recorded flying above 27.5 m.

Two species of geese were recorded on baseline surveys in the offshore site and 8 km buffer area, with 45.4% of pink-footed goose recorded flying above 27.5 m (n=577), while 100% of barnacle goose sightings were recorded below 27.5 m (n=900).

Golden plover was the only species of wader for which more than 20 individuals were recorded, with 41.7% recorded flying above 27.5 m (n=24). The majority of all other wader species combined (83.6%, n=61) were recorded flying below 27.5 m in height.

Meadow pipit was the only species of land bird for which more than 20 individuals were recorded, with 1.7% recorded flying above 27.5 m (n=58). All other passerine species combined (n=33) were recorded flying below 27.5 m in height.

Table 4.3 Flight heights of birds in the offshore site and 8 km buffer area in Years 1 to 3 (November 2009 to October 2012) [1]

Species	Height bands in metres						Total in flight	% above 27.5m
	0 – 7.5	7.5 – 12.5	12.5 –17.5	17.5 – 22.5	22.5 – 27.5	Above 27.5		
Fulmar	1,806	23	0	0	0	2	1,831	0.1
Sooty shearwater	90	0	0	0	0	0	90	0
Manx shearwater	206	1	0	0	0	0	207	0
Gannet	32,704	2,231	925	2,797	604	1,989	41,250	4.8
Pink-footed goose	301	0	0	14	0	262	577	45.4
Barnacle goose	900	0	0	0	0	0	900	0
Wigeon	0	0	0	20	0	1	21	4.8
Eider	13	12	0	2	2	0	29	0
Golden plover	14	0	0	0	0	10	24	41.7
Waders (combined)	43	2	3	1	2	10	61	16.4
Arctic skua	10	5	1	5	0	1	22	4.5
Great skua	34	6	4	1	2	2	49	4.1
Little gull	216	23	3	1	2	82	327	25.1
Black-headed gull	5	12	6	5	1	9	38	23.7
Common gull	28	21	10	28	9	27	123	22.0
Lesser black-backed gull	164	59	26	65	11	33	358	9.2
Herring gull	542	282	109	270	86	357	1,646	21.7

Species	Height bands in metres						Total in flight	% above 27.5m
	0 – 7.5	7.5 – 12.5	12.5 –17.5	17.5 – 22.5	22.5 – 27.5	Above 27.5		
Great black-backed gull	224	81	33	87	21	107	553	19.3
Large gull species	159	35	26	56	5	159	445	35.7
Kittiwake	3,326	1,706	504	920	156	333	6,945	4.8
Common tern	30	5	1	0	0	0	36	0
Arctic tern	938	178	31	36	1	2	1,186	0.2
Common/Arctic tern	137	62	16	1	0	0	216	0
Unidentified tern species	34	0	0	0	0	0	34	0
Guillemot	6,716	77	7	9	1	1	6,812	0.01
Razorbill	1,913	21	8	7	0	0	1,949	0
Little auk	630	1	0	0	0	0	631	0
Puffin	6,962	68	7	6	2	3	7,049	0.04
Guillemot/razorbill	1,439	13	1	0	0	0	1,453	0
Unidentified auk species	141	0	0	0	0	0	141	0
Meadow pipit	22	23	9	2	1	1	58	1.7
Passerines combined	24	6	2	0	1	0	33	0
Total numbers	**59,771**	**4,953**	**1,732**	**4,333**	**907**	**3,391**	**75,094**	**4.5**

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

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 and Arctic skua were only recorded in the 8 km buffer area around the offshore site. For this reason they were 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 Offshore site and 8 km buffer 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 4.4).

No red-throated divers were recorded in the offshore site and 8 km buffer area in Year 2 or Year 3.

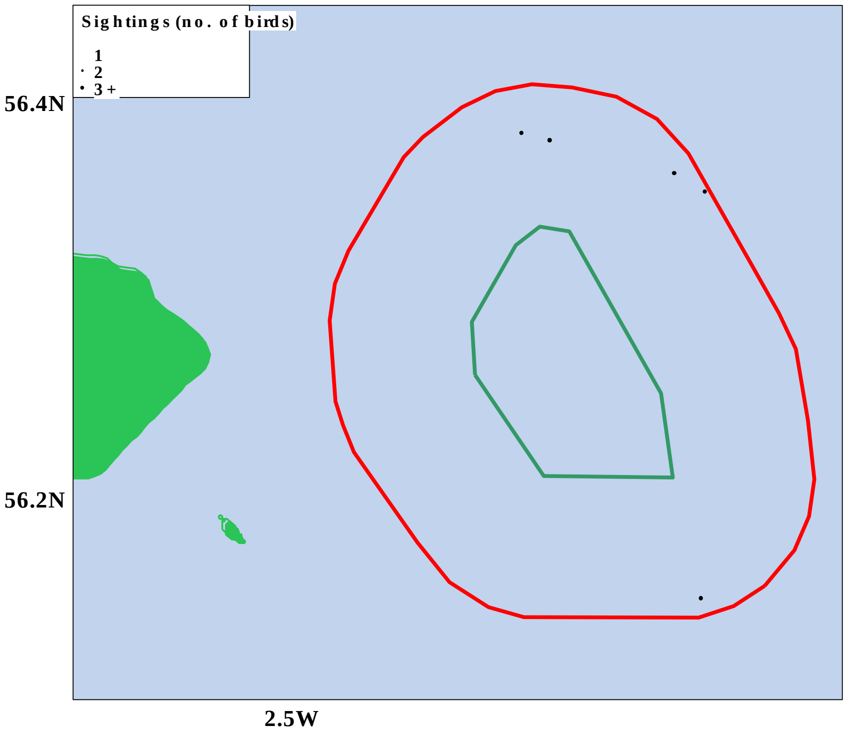


Figure 4.4 Red-throated diver sightings in Year 1

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).

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 or Year 3.

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 offshore site and 8 km buffer 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 Offshore site and 8 km buffer area

Fulmars were regularly recorded on baseline surveys in the offshore site and buffer area, with a total of 692 fulmars were recorded in Year 1, 1,116 birds in Year 2 and 491 birds in Year 3 (raw numbers, all sea states). The majority of fulmars were recorded in the buffer area (Table 4.2).

During the Year 1 breeding season (April to September), the mean estimated number of fulmars in the offshore site was 13 birds, with a peak of 47 birds in September (Table 4.4). In the same period of Year 2, the mean estimated number of fulmars in the offshore site was 34 birds, with a peak of 69 birds in September. In Year 3, the mean estimated number of fulmars during the breeding season was 40 birds, with a peak of 104 birds in September.

In the Year 1 non-breeding season (October to March), the mean estimated number of fulmars in the offshore site was 40 birds, with a peak of 60 birds in December (Table 4.4). In the same period of Year 2, the mean estimated number of fulmars in the offshore site was 41 birds, with a peak of 77 birds in both December and January. In Year 3, the mean estimated number of fulmars during the non-breeding season was 21 birds, with a peak of 53 birds in January.

Mean monthly estimated numbers of fulmars in the offshore site and the buffer area, based on three years, are shown in Figure 4.5.

Table 4.4 Estimated numbers of fulmars in the offshore site (plus 1, 2 and 8 km buffer) in Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 8 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	0	0	0	14	14	14	20	198
Yr1 Dec	33	6	192	27	60	94	115	245
Yr1 Jan	0	0	0	48	48	54	74	166
Yr1 Feb	42	9	199	7	49	49	88	169
Yr1 Mar	45	18	114	13	59	83	83	321
Yr1 Apr	0	0	0	14	14	14	41	123
Yr1 May	0	0	0	0	0	0	0	52
Yr1 Jun	0	0	0	0	0	0	0	18
Yr1 Jul	0	0	0	0	0	0	0	27
Yr1 Aug	0	0	0	14	14	25	39	75
Yr1 Sep	40	16	103	7	47	87	100	295
Yr1 Oct	0	0	0	7	7	7	7	7
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	0	0	0	77	77	77	154	233
Yr2 Jan	9	4	23	68	77	120	140	505
Yr2 Feb	11	2	65	20	32	32	38	93
Yr2 Mar	0	0	0	13	13	13	13	124
Yr2 Apr	45	18	113	7	52	52	70	169
Yr2 May	0	0	0	13	13	27	27	116
Yr2 Jun	0	0	0	34	34	34	47	102
Yr2 Jul	9	3	28	13	23	68	68	153
Yr2 Aug	0	0	0	14	14	37	84	877
Yr2 Sep	42	22	79	27	69	125	252	962
Yr2 Oct	0	0	0	7	7	7	7	83
Yr3 Nov	0	0	0	0	0	0	0	32
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	21	11	42	14	35	53	85	250
Yr3 Feb	0	0	0	0	0	0	31	122
Yr3 Mar	28	11	68	14	41	51	57	179
Yr3 Apr	58	19	176	0	58	76	76	140
Yr3 May	0	0	0	7	7	7	7	32
Yr3 Jun	0	0	0	0	0	0	0	34
Yr3 Jul	0	0	0	28	28	53	66	116
Yr3 Aug	24	6	98	20	43	43	50	217
Yr3 Sep	31	16	60	35	66	104	111	264
Yr3 Oct	20	5	77	7	27	27	27	44

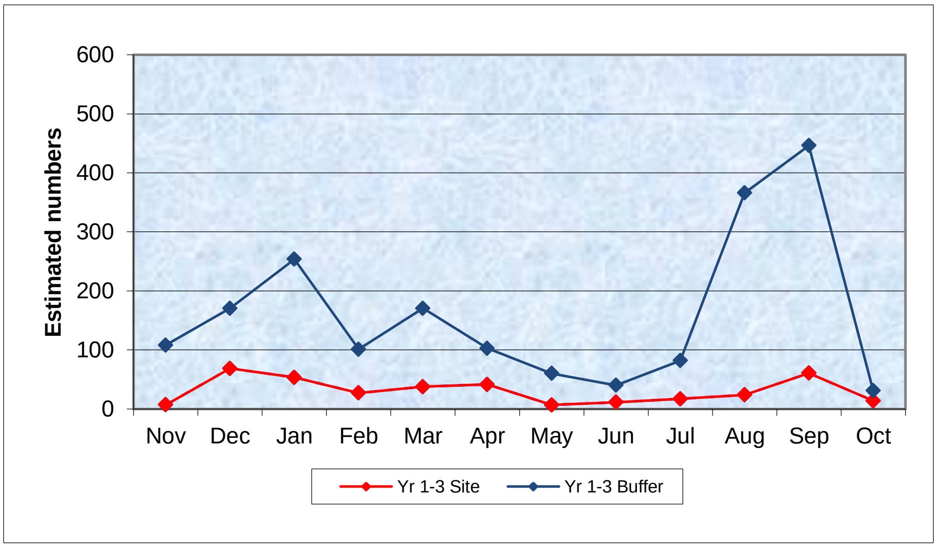


Figure 4.5 Mean monthly estimated numbers of fulmars in the Neart na Gaoithe Development & buffer areas in Years 1 to 3 (Three-year mean)

Between October and March of Year 1, fulmars were widespread at low to moderate densities across most of the offshore site and 8 km buffer area, although fewer birds were present in the west (Figure 4.6).

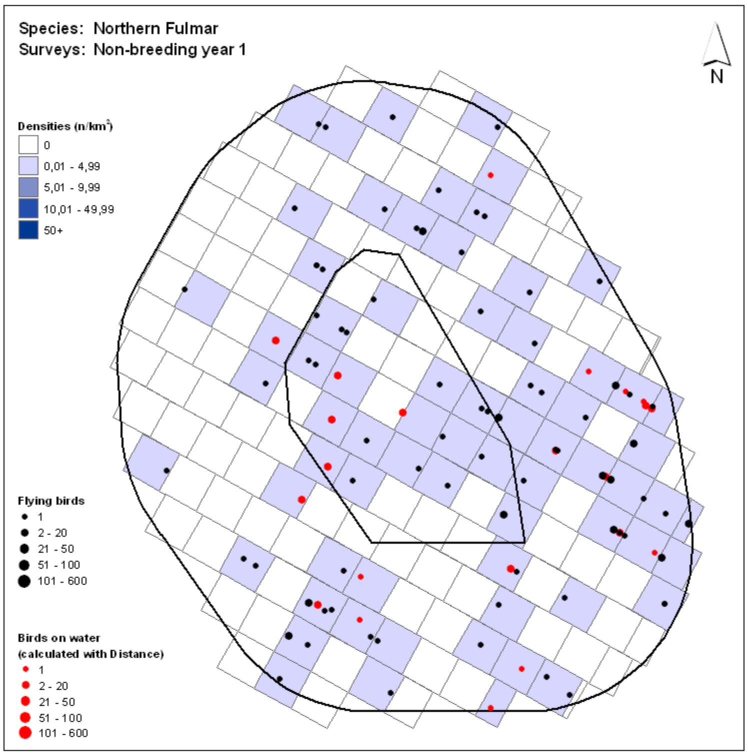


Figure 4.6 Fulmar density between October and March, Year 1

Between October and March of Year 2, fulmars were widespread at mostly low densities across the offshore site and 8 km buffer area (Figure 4.7). Highest densities were recorded to the south of the offshore site at this time.

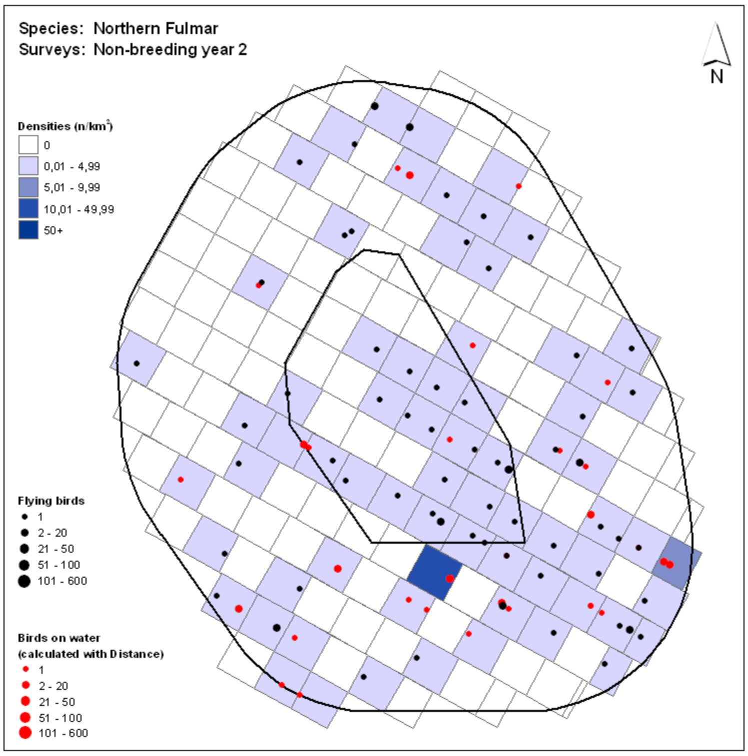


Figure 4.7 Fulmar density between October and March, Year 2

Between October and March of Year 3, fulmars were scattered at mostly low densities across the offshore site and 8 km buffer area, with a more restricted distribution than in the previous years (Figure 4.8).

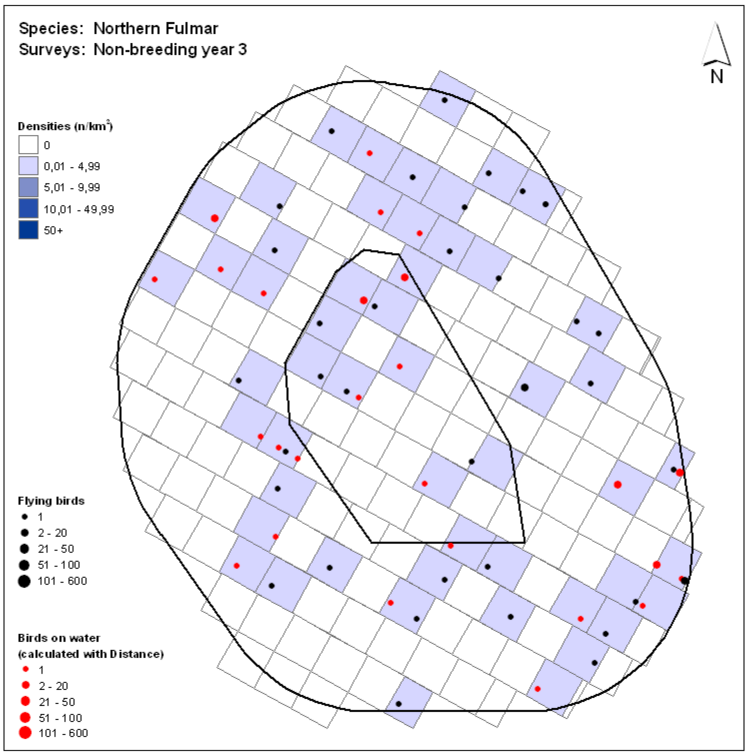


Figure 4.8 Fulmar density between October and March, Year 3

Fulmar density during the Year 1 breeding season (April to September) was low across the offshore site and 8 km buffer area. Few birds were recorded in the offshore site at this time (Figure 4.9).

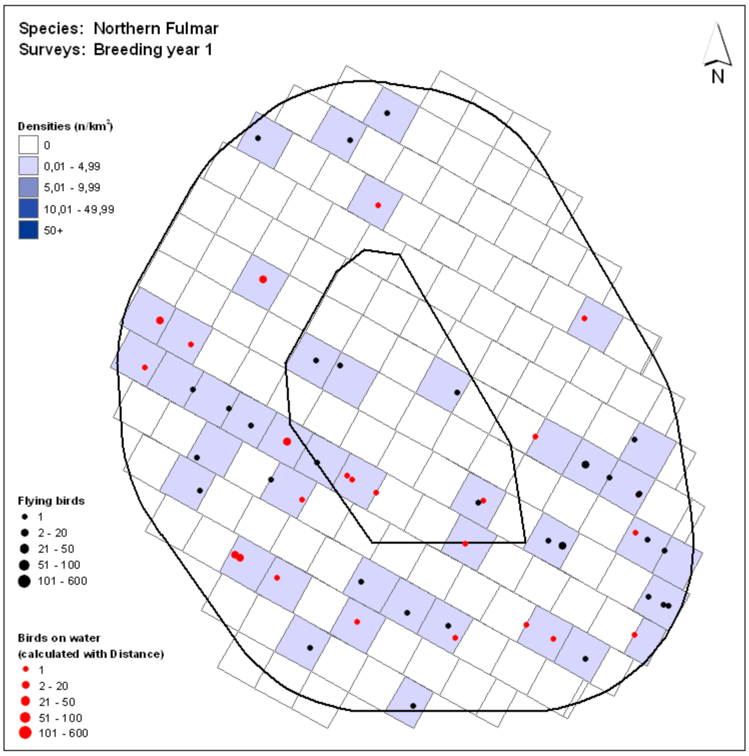


Figure 4.9 Fulmar density between April and September, Year 1

In the Year 2 breeding season, fulmars were more widespread than in the same period in Year 1. Densities generally remained low throughout the offshore site and buffer area (Figure 4.10).

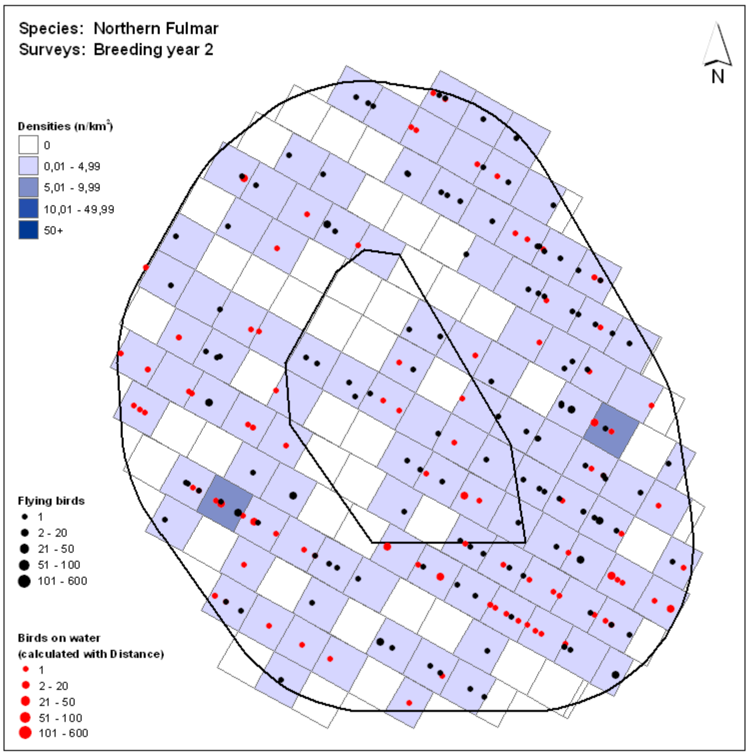


Figure 4.10 Fulmar density between April and September, Year 2

Fulmar distribution in the Year 3 breeding season was similar to the same period in Year 1, with birds generally scattered at low densities across the offshore site and the buffer area (Figure 4.11).

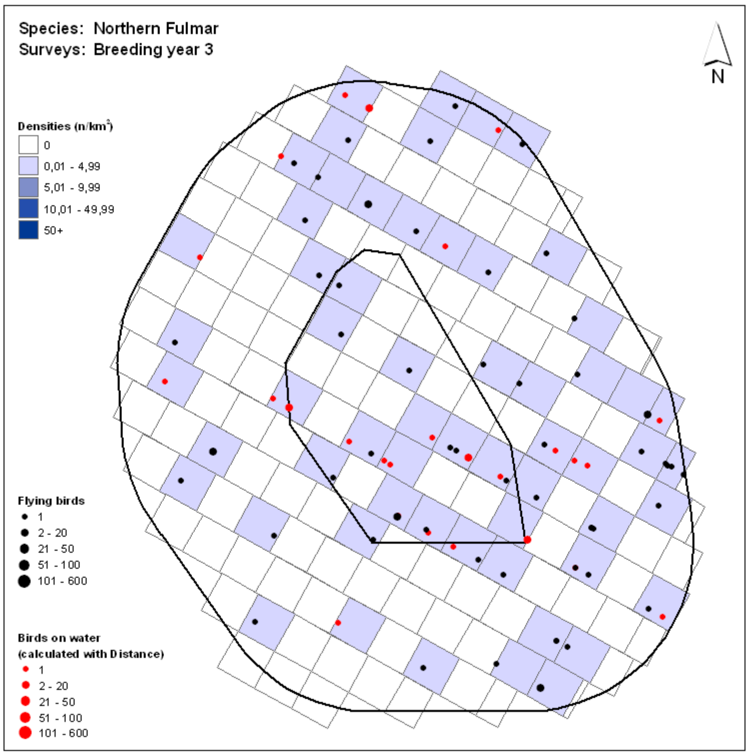
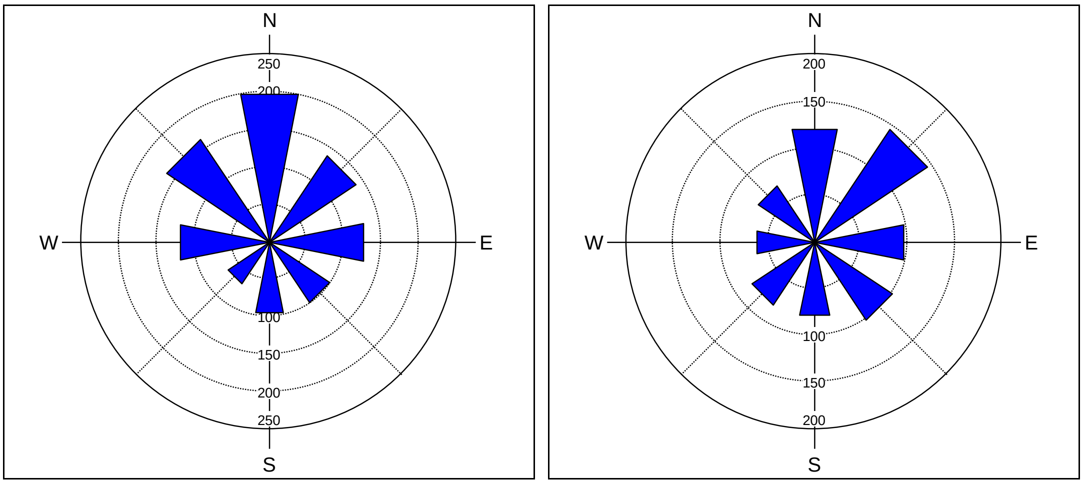


Figure 4.11 Fulmar density between April and September, Year 3

A total of 1,831 fulmars were recorded in flight in Years 1 to 3, with the majority of all birds (98.6%) recorded flying below 7.5 m in height (Table 4.3). Just two birds (0.1 %) were recorded flying above 27.5 m, i.e. within the rotor swept zone of the turbines, at an estimated height of 30 m.



April to September (n=1,010 birds) October to March (n=757 birds)

Numbers shown on figures are numbers of birds recorded

Figure 4.12 Flight direction of fulmars in the offshore site and 8 km buffer area in Years 1 to 3

Flight direction was recorded for 1,010 fulmars in the breeding season (April to September), with direction recorded for 757 fulmars in the non-breeding season (October to March) (Figure 4.12). An additional 48 birds were recorded as circling (not shown).

In the breeding season, there was a slight pattern of fulmars flying north (19.8%), north-west (16.3%) and north-east (13.8%), with fewer birds recorded flying in other directions. In the non-breeding season, 19.2% of birds were recorded flying north-east, with 16.2% flying north.

Four types of foraging behaviour were recorded for fulmars in Years 1 to 3, with surface pecking the most frequently recorded behaviour, although the sample size was very small (Table 4.5).

Table 4.5 Fulmar foraging behaviour in the offshore site and 8 km buffer area in Years 1 to 3

Behaviour	Number of birds
Actively searching	5
Dipping	2
Scavenging	4
Surface pecking	10

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).

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 4.6). 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 4.13.

Table 4.6 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**	**29,649**	**2011**
Fetlar	*477*	*9,800*	*0.1*	*1.8*	*9,203*	*1999-2001*
Forth Islands	**16**	**1,600**	**<0.1**	**0.3**	**4,245**	**2012**
Foula	*424*	*46,800*	*0.6*	*8.7*	*21,106*	*2000*
Fowlsheugh	**62**	**1,170**	**<0.1**	**0.2**	**119**	**2012**
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*	*5,248*	*2011*
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 %**	**150,176**	**-**

Sources: 1 JNCC (2013) – SPA online species accounts. 2 SMP (2013) – 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).

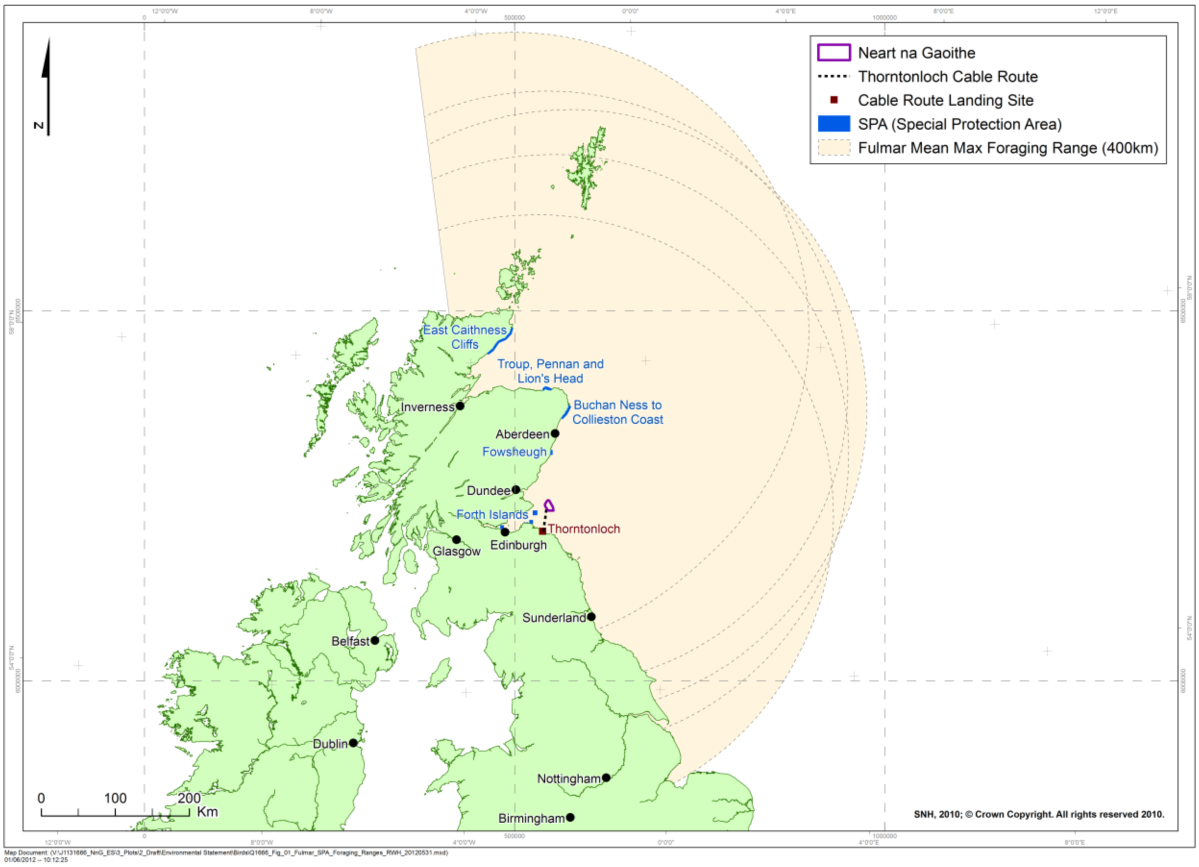


Figure 4.13 Fulmar foraging range from the five nearest SPAs in relation to the Development

4.4.2.3 Assessment

Definition of seasons

The annual cycle for fulmar was 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, was defined as April 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 was defined as October to March and broadly corresponds to the period when fulmars are in their over-wintering areas. 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).

Populations

The regional breeding population within mean maximum foraging range (400 km) (Thaxter *et al.,* 2012) is estimated to be 319,878 birds (or 159,939 pairs). This figure is based on the Seabird 2000 counts from Sumburgh Head (Shetland) to Spurn Point (Humberside) (Mitchell *et al.,* 2004).

The regional breeding SPA population within mean maximum foraging range (400 km) (Thaxter *et al.,* 2012) is estimated to be 206,950 birds (or 103,475 pairs). This figure is based on most recent available counts from these SPAs (Table 4.6).

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

The nature conservation importance (NCI) of fulmars using the offshore site was rated at moderate throughout the year, 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. The three-year mean peak estimated number of fulmars present in the breeding season in the offshore site (31 birds) is 0.01% of the regional breeding SPA population (103,475 pairs) within mean maximum foraging range (400 km) (Thaxter *et al.,* 2012).

Offshore wind farm studies

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 fulmars. 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).

Construction Phase

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.

Operational Phase

Displacement

In order to assess the potential impacts from displacement, the approach presented in the interim guidance note on displacement issued by the JNCC and NE has been followed (JNCC & NE 2012).

Peak estimated numbers of fulmars in the offshore site in the breeding season (April to September) and non-breeding season (October to March) for Years 1 to 3 were averaged to get the three-year mean peak per season. Where peak numbers occurred in different months within the same season across different years, the peak month was used. This was repeated for a 1 km and 2 km buffer around the offshore site (Table 4.7).

Table 4.7 Seasonal three-year mean peak estimated numbers of fulmars in the offshore site (plus 1 & 2 km buffer)

Year	Offshore site		Offshore site + 1 km		Offshore site + 2 km	
	Breeding	Non-breeding	Breeding	Non-breeding	Breeding	Non-breeding
Year 1	47	60	87	94	100	115
Year 2	69	77	125	120	252	154
Year 3	66	41	104	53	111	85
3-year mean peak	61	59	105	89	154	118

For the purposes of assessment it was assumed that all birds in the breeding season were breeding birds. This is precautionary, as it is likely than some immature birds were also present.

Guidance recommends presenting a range of potential displacement and mortality rates and wherever possible selecting a suitable impact based on empirical evidence. Where, there is little evidence to support the assessment a precautionary approach should be taken.

Likely impacts of displacement

Assuming 10% of all fulmars were to be displaced from the offshore site during the breeding season, this would affect an estimated six birds, (Table 4.8), increasing to 15 birds for the offshore site plus 2 km buffer (Table 4.10). As fulmars have a very large foraging range, it was considered that the majority of displaced birds would be able to find other suitable foraging areas. A conservative level of 2% mortality was therefore used in this assessment, resulting in zero fulmars predicted to die as a result of being displaced from the offshore site and 2 km buffer area in the breeding season.

Table 4.8 Estimated number of fulmars predicted to be at risk of mortality following displacement from offshore site in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	0	1	1	2	2	3	4	4	5	5	6
	20%	0	1	1	2	4	5	6	7	9	10	11	12
	30%	0	1	2	4	5	7	9	11	13	15	16	18
	40%	0	1	2	5	7	10	12	15	17	20	22	24
	50%	1	2	3	6	9	12	15	18	21	24	27	31
	60%	1	2	4	7	11	15	18	22	26	29	33	37
	70%	1	2	4	9	13	17	21	26	30	34	38	43
	80%	1	2	5	10	15	20	24	29	34	39	44	49
	90%	1	3	5	11	16	22	27	33	38	44	49	55
	100%	1	3	6	12	18	24	31	37	43	49	55	61
Three-year mean peak of 61 fulmars in the offshore site in the breeding season (April to Sept)													
SPA Population within mean max foraging range (400 km) = 103,475 pairs (SMP 2013)													
Regional population within mean max foraging range = 159,969 pairs (SMP 2013)													

Table 4.9 Estimated number of fulmars predicted to be at risk of mortality following displacement from offshore site plus 1 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	1	1	2	3	4	5	6	7	8	9	11
	20%	0	1	2	4	6	8	11	13	15	17	19	21
	30%	1	2	3	6	9	13	16	19	22	25	28	32
	40%	1	2	4	8	13	17	21	25	29	34	38	42
	50%	1	3	5	11	16	21	26	32	37	42	47	53
	60%	1	3	6	13	19	25	32	38	44	50	57	63
	70%	1	4	7	15	22	29	37	44	51	59	66	74
	80%	2	4	8	17	25	34	42	50	59	67	76	84
	90%	2	5	9	19	28	38	47	57	66	76	85	95
	100%	2	5	11	21	32	42	53	63	74	84	95	105
Three-year mean peak of 105 fulmars in the offshore site & 1 km buffer in the breeding season (April to Sept)													
SPA Population within mean max foraging range (400 km) = 103,475 pairs (SMP 2013)													
Regional population within mean max foraging range = 159,969 pairs (SMP 2013)													

Table 4.10 Estimated number of fulmars predicted to be at risk of mortality following displacement from offshore site plus 2 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	1	2	3	5	6	8	9	11	12	14	15
	20%	1	2	3	6	9	12	15	18	22	25	28	31
	30%	1	2	5	9	14	18	23	28	32	37	42	46
	40%	1	3	6	12	18	25	31	37	43	49	55	62
	50%	2	4	8	15	23	31	39	46	54	62	69	77
	60%	2	5	9	18	28	37	46	55	65	74	83	92
	70%	2	5	11	22	32	43	54	65	75	86	97	108
	80%	2	6	12	25	37	49	62	74	86	99	111	123
	90%	3	7	14	28	42	55	69	83	97	111	125	139
	100%	3	8	15	31	46	62	77	92	108	123	139	154
Three-year mean peak of 154 fulmars in the offshore site & 2 km buffer in the breeding season (April to Sept)													
SPA Population within mean max foraging range (400 km) = 103,475 pairs (SMP 2013)													
Regional population within mean max foraging range = 159,969 pairs (SMP 2013)													

For any displaced breeding fulmars, there could potentially be a detrimental impact on their breeding success, as a result of having to travel further on each trip to forage elsewhere. However, comparing the distribution of fulmars in the offshore site from Years 1 to 3 in the breeding season shows that birds were present in the offshore site at mostly low densities, with few birds recorded in the offshore site at this time (Figure 4.9, Figure 4.10 and Table 4.12). This indicates that the offshore site is not an important foraging area for fulmars in the breeding season. In addition, as fulmars have a very large foraging range, it was considered that should displacement occur, it would not cause a significant detrimental impact on fulmar breeding success.

Assuming 10% of all fulmars were displaced from the offshore site during the non-breeding season, this would affect an estimated six birds (Table 4.11), increasing to 12 birds for the offshore site plus 2 km buffer (Table 4.13). However, given that fulmars are not tied to a colony in the non-breeding season, and are therefore free to forage further afield, any additional mortality arising from displacement from the offshore site is likely to be minimal. It was concluded that any displaced fulmars would move to alternative foraging areas over the winter months.

Based on the distribution and densities of fulmars recorded from baseline studies, and the large foraging range of fulmars (Thaxter *et al.,* 2012), the regional fulmar population in the breeding and non-breeding seasons is considered to have low sensitivity to displacement effects and it is therefore likely that should displacement occur, it will not result in any discernible population effects on the regional population throughout the year.

This impact is categorised as having 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 fulmar population in the breeding and non-breeding seasons are ***not significant*** under the EIA Regulations.

Table 4.11 Estimated number of fulmars predicted to be at risk of mortality following displacement from offshore site in the non-breeding season

		Mortality (%)											
Displacement level (%)		0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	0	1	1	2	2	3	4	4	5	5	6
	20%	0	1	1	2	4	5	6	7	8	9	11	12
	30%	0	1	2	4	5	7	9	11	12	14	16	18
	40%	0	1	2	5	7	9	12	14	17	19	21	24
	50%	1	1	3	6	9	12	15	18	21	24	27	30
	60%	1	2	4	7	11	14	18	21	25	28	32	35
	70%	1	2	4	8	12	17	21	25	29	33	37	41
	80%	1	2	5	9	14	19	24	28	33	38	42	47
	90%	1	3	5	11	16	21	27	32	37	42	48	53
	100%	1	3	6	12	18	24	30	35	41	47	53	59
Three-year mean peak of 59 fulmars in the offshore site in the non-breeding season (Oct to Mar)													
Regional population in the non-breeding season = 400,000 birds (Skov *et al*., 1995)													

Table 4.12 Estimated number of fulmars predicted to be at risk of mortality following displacement from offshore site plus 1 km buffer in the non-breeding season

	Mortality (%)												
	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
Displacement level (%)	10%	0	0	1	2	3	4	4	5	6	7	8	9
	20%	0	1	2	4	5	7	9	11	12	14	16	18
	30%	1	1	3	5	8	11	13	16	19	21	24	27
	40%	1	2	4	7	11	14	18	21	25	28	32	36
	50%	1	2	4	9	13	18	22	27	31	36	40	45
	60%	1	3	5	11	16	21	27	32	37	43	48	53
	70%	1	3	6	12	19	25	31	37	44	50	56	62
	80%	1	4	7	14	21	28	36	43	50	57	64	71
	90%	2	4	8	16	24	32	40	48	56	64	72	80
	100%	2	4	9	18	27	36	45	53	62	71	80	89
Three-year mean peak of 89 fulmars in the offshore site in the non-breeding season (Oct to Mar)													
Regional population in the non-breeding season = 400,000 birds (Skov *et al*., 1995)													

Table 4.13 Estimated number of fulmars predicted to be at risk of mortality following displacement from offshore site plus 2 km buffer in the non-breeding season

	Mortality (%)												
	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
Displacement level (%)	10%	0	1	1	2	4	5	6	7	8	9	11	12
	20%	0	1	2	5	7	9	12	14	17	19	21	24
	30%	1	2	4	7	11	14	18	21	25	28	32	35
	40%	1	2	5	9	14	19	24	28	33	38	42	47
	50%	1	3	6	12	18	24	30	35	41	47	53	59
	60%	1	4	7	14	21	28	35	42	50	57	64	71
	70%	2	4	8	17	25	33	41	50	58	66	74	83
	80%	2	5	9	19	28	38	47	57	66	76	85	94
	90%	2	5	11	21	32	42	53	64	74	85	96	106
	100%	2	6	12	24	35	47	59	71	83	94	106	118
Three-year mean peak of 118 fulmars in the offshore site in the non-breeding season (Oct to Mar)													
Regional population in the non-breeding season = 400,000 birds (Skov *et al*., 1995)													

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 fulmars 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 fulmars 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 3.7 and Table 3.8). 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 3.7). 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 3.7). 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 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 3.8).

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 3.8) 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 fulmars 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 therefore categorised as having 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). 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 collision mortality on fulmars was rated as being 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.

Decommissioning Phase

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). 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.

Summary of combined effects

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 4.14).

Table 4.14 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***

Similarly, 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 4.15).

Table 4.15 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

There were no significant impacts predicted for the regional population of fulmars in either the breeding or non-breeding seasons from the proposed Neart na Gaoithe development. The predicted effects of the development on the regional population of fulmars in the breeding or non-breeding seasons arising from construction, operation and decommissioning are very close to no effect.

Based on these findings, it was considered that it is not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for this population, and therefore no further cumulative impact assessment was undertaken for this species.

4.4.2.5 Mitigation measures

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 Offshore site and 8 km buffer area

Sooty shearwaters were regularly recorded on autumn passage on baseline surveys in the offshore site and 8 km buffer area although numbers varied between years. In Year 1, a total of 227 birds were recorded, with 179 birds in Year 2 and 12 birds in Year 3 (raw numbers, all sea states). The majority of birds were recorded in the buffer area (Table 4.2).

During the Year 1 autumn period (September and October), the mean estimated number of sooty shearwaters in the offshore site was 69 birds, with a peak of 130 birds in October (Table 4.16). In the same period of Year 2, the mean estimated number of sooty shearwaters in the offshore site was seven birds, with seven birds estimated for both September and October. No sooty shearwaters were recorded in the offshore site in Year 3.

Table 4.16 Estimated numbers of sooty shearwaters in the offshore site (plus 1 & 2 km buffer) in Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 8 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	0	0	0	7	7	7	7	7
Yr1 Dec	0	0	0	0	0	0	0	0
Yr1 Jan	0	0	0	0	0	0	0	0
Yr1 Feb	0	0	0	0	0	0	0	0
Yr1 Mar	0	0	0	0	0	0	0	0
Yr1 Apr	0	0	0	0	0	0	0	0
Yr1 May	0	0	0	0	0	0	0	0
Yr1 Jun	0	0	0	0	0	0	0	0
Yr1 Jul	0	0	0	0	0	0	0	0
Yr1 Aug	0	0	0	0	0	0	14	14
Yr1 Sep	0	0	0	7	7	30	47	266
Yr1 Oct	130	76	225	0	130	174	243	633
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	0	0	0	0	0	0	0	0
Yr2 Jan	0	0	0	0	0	0	0	0
Yr2 Feb	0	0	0	0	0	0	0	0
Yr2 Mar	0	0	0	0	0	0	0	0
Yr2 Apr	0	0	0	0	0	0	0	0
Yr2 May	0	0	0	0	0	0	0	0
Yr2 Jun	0	0	0	0	0	0	0	0
Yr2 Jul	0	0	0	0	0	0	0	0
Yr2 Aug	0	0	0	0	0	0	0	14
Yr2 Sep	0	0	0	7	7	26	104	204
Yr2 Oct	0	0	0	7	7	21	21	448
Yr3 Nov	0	0	0	0	0	0	0	0
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	0	0	0	0	0	0	0	0
Yr3 Feb	0	0	0	0	0	0	0	0
Yr3 Mar	0	0	0	0	0	0	0	0
Yr3 Apr	0	0	0	0	0	0	0	0
Yr3 May	0	0	0	0	0	0	0	0
Yr3 Jun	0	0	0	0	0	0	0	0
Yr3 Jul	0	0	0	0	0	0	0	19
Yr3 Aug	0	0	0	0	0	0	0	51
Yr3 Sep	0	0	0	0	0	0	19	19
Yr3 Oct	0	0	0	0	0	0	0	0

Mean monthly estimated numbers of sooty shearwaters in the offshore site and the buffer area, based on three years, are shown in Figure 4.14.

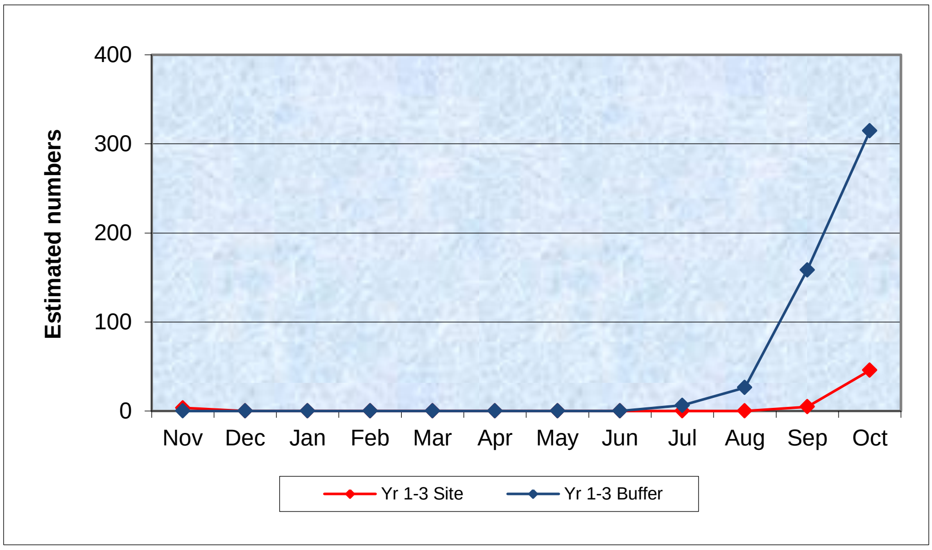


Figure 4.14 Mean monthly estimated numbers of sooty shearwaters in the Neart na Gaoithe Development & buffer areas in Years 1 to 3 (Three-year mean)

Sooty shearwaters were scattered across the offshore site in moderate to high densities in October of Year 1 (Figure 4.15). Distribution in the 8 km buffer area was also quite restricted at this time, with few birds in the east of the buffer area.

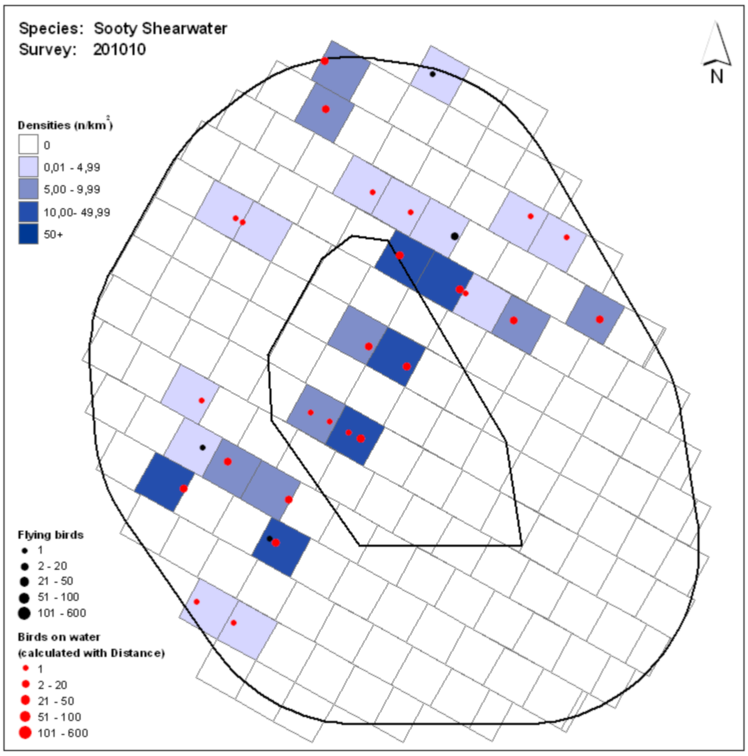


Figure 4.15 Sooty shearwater density in October, Year 1

In October of Year 2, sooty shearwaters were mostly recorded in the south of the offshore site and 8 km buffer area at moderate to high densities (Figure 4.16).

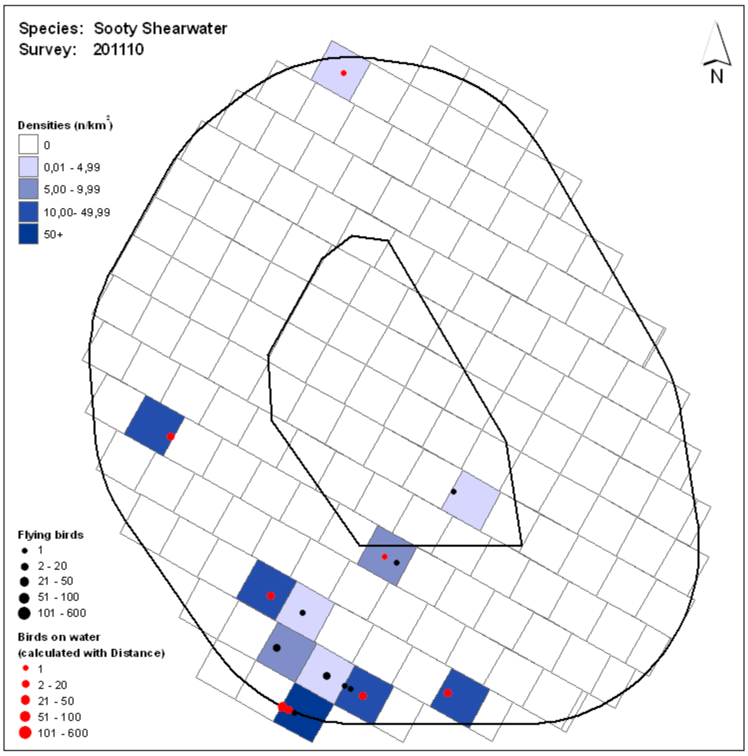


Figure 4.16 Sooty shearwater density in October, Year 2

Sooty shearwaters were not recorded in the offshore site or 8 km buffer area in October of Year 3.

A total of 90 sooty shearwaters were recorded in flight in Years 1 to 3, with all birds flying below 7.5 m in height (Table 4.3).

4.4.3.3 Species sensitivity

A review assessed sooty shearwater as being at low risk of collision and displacement resulting from offshore wind farms (Langston 2010). 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

Almost all sooty shearwaters were recorded in the offshore site in September and October, therefore, this was the period considered for assessment. 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).

Populations

The size of the regional autumn passage population for sooty shearwaters is imprecisely known and is likely to vary year-to-year, which presents a difficulty for undertaking the assessment. The regional population during the autumn passage period was assumed to be

2,500 birds, based on peak autumn passage counts from Scottish east coast headlands, where "hundreds" may be recorded in a day (Forrester *et al.,* 2007) and from published ESAS data (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; ICOL, 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

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.

Construction Phase

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.

Operational phase

Displacement

In order to assess the potential impacts from displacement, the approach presented in the interim guidance note on displacement issued by the JNCC and NE has been followed (JNCC & NE 2012).

Peak estimated numbers of sooty shearwaters in the offshore site in the autumn passage period (September and October) for Years 1 to 3 were averaged to get the three-year mean peak per season. Where peak numbers occurred in different months within the same season across different years, the peak month was used. This was repeated for a 1 km and 2 km buffer around the offshore site (Table 4.17).

Table 4.17 Seasonal three-year mean peak estimated numbers of sooty shearwaters in the offshore site (plus 1 & 2 km buffer)

Year	Offshore site	Offshore site + 1 km	Offshore site + 2 km
	Autumn passage	Autumn passage	Autumn passage
Year 1	130	174	243
Year 2	7	26	104
Year 3	0	0	19
3-year mean peak	46	67	122

Guidance recommends presenting a range of potential displacement and mortality rates and wherever possible selecting a suitable impact based on empirical evidence. Where, there is little evidence to support the assessment a precautionary approach should be taken.

Likely impacts of displacement

Assuming 10% of all sooty shearwaters were to be displaced from the offshore site during the autumn passage period, this would affect an estimated five birds, (Table 4.18), increasing to 12 birds for the offshore site plus 2 km buffer (Table 4.20). This is approximately 0.5% of the assumed regional population in the autumn passage period (2,500 birds) (Forrester *et al.,* 2007, Stone *et al.,* 1995). As sooty shearwaters are on migration at this time, they are moving through the area and it was considered that any displaced birds would be able to find other suitable foraging areas, and that no sooty shearwaters would die as a result of being displaced from the offshore site and 2 km buffer area in the autumn passage period.

Table 4.18 Estimated number of sooty shearwaters predicted to be at risk of mortality following displacement from offshore site in the autumn passage season

	Mortality (%)												
	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
Displacement level (%)	10%	0	0	0	1	1	2	2	3	3	4	4	5
	20%	0	0	1	2	3	4	5	6	6	7	8	9
	30%	0	1	1	3	4	6	7	8	10	11	12	14
	40%	0	1	2	4	6	7	9	11	13	15	17	18
	50%	0	1	2	5	7	9	12	14	16	18	21	23
	60%	1	1	3	6	8	11	14	17	19	22	25	28
	70%	1	2	3	6	10	13	16	19	23	26	29	32
	80%	1	2	4	7	11	15	18	22	26	29	33	37
	90%	1	2	4	8	12	17	21	25	29	33	37	41
	100%	1	2	5	9	14	18	23	28	32	37	41	46
Three-year mean peak of 46 sooty shearwaters in the offshore site in the autumn period (Sep to Oct)													
Regional population in the autumn period = c. 2,500 birds (Stone *et al*., 1995)													

Table 4.19 Estimated number of sooty shearwaters predicted to be at risk of mortality following displacement from offshore site plus 1 km buffer in the autumn passage season

	Mortality (%)												
	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
Displacement level (%)	10%	0	0	1	1	2	3	3	4	5	5	6	7
	20%	0	1	1	3	4	5	7	8	9	11	12	13
	30%	0	1	2	4	6	8	10	12	14	16	18	20
	40%	1	1	3	5	8	11	13	16	19	21	24	27
	50%	1	2	3	7	10	13	17	20	23	27	30	34
	60%	1	2	4	8	12	16	20	24	28	32	36	40
	70%	1	2	5	9	14	19	23	28	33	38	42	47
	80%	1	3	5	11	16	21	27	32	38	43	48	54
	90%	1	3	6	12	18	24	30	36	42	48	54	60
	100%	1	3	7	13	20	27	34	40	47	54	60	67
Three-year mean peak of 67 sooty shearwaters in the offshore site in the autumn period (Sep to Oct)													
Regional population in the autumn period = c. 2,500 birds (Stone *et al*., 1995)													

Table 4.20 Estimated number of sooty shearwaters predicted to be at risk of mortality following displacement from offshore site plus 2 km buffer in the autumn passage season

	Mortality (%)												
	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
Displacement level (%)	10%	0	1	1	2	4	5	6	7	9	10	11	12
	20%	0	1	2	5	7	10	12	15	17	20	22	24
	30%	1	2	4	7	11	15	18	22	26	29	33	37
	40%	1	2	5	10	15	20	24	29	34	39	44	49
	50%	1	3	6	12	18	24	31	37	43	49	55	61
	60%	1	4	7	15	22	29	37	44	51	59	66	73
	70%	2	4	9	17	26	34	43	51	60	68	77	85
	80%	2	5	10	20	29	39	49	59	68	78	88	98
	90%	2	5	11	22	33	44	55	66	77	88	99	110
	100%	2	6	12	24	37	49	61	73	85	98	110	122
Three-year mean peak of 122 sooty shearwaters in the offshore site in the autumn period (Sep to Oct)													
Regional population in the autumn period = c. 2,500 birds (Stone *et al*., 1995)													

On autumn passage sooty shearwaters 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. It was considered that the regional sooty shearwater population in the autumn passage period would have low sensitivity to displacement effects. It is therefore likely that should displacement occur, it will not result in any discernible effects on the regional population in this period.

This impact is categorised as having 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 sooty shearwater population in the autumn passage period are ***not significant*** under the EIA Regulations.

Barrier effects

Sooty shearwaters occur off East Scotland only as a passage migrant (Forrester *et al.*, 2007). 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 the 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

CRM was not undertaken for sooty shearwaters because all of the 90 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 collision mortality on sooty shearwaters was rated as being 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.

Decommissioning Phase

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.

Summary of combined effects

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 sooty shearwaters in the autumn passage period is negligible. As 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 4.21).

Table 4.21 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	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***

4.4.3.5 Cumulative Impact Assessment

There were no significant impacts predicted for the regional population of sooty shearwaters in the autumn passage period from the proposed Neart na Gaoithe development. The predicted effects of the development on the regional population of sooty shearwaters in the autumn passage period arising from construction, operation and decommissioning are very close to no effect.

Based on these findings, it was considered that it is not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for this population, and therefore no further cumulative impact assessment was undertaken for this species.

4.4.3.6 Mitigation measures

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 off 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 Offshore site and 8 km buffer area

Manx shearwaters were only recorded in the offshore site between July and October on baseline surveys, although numbers varied between years. In Year 1, a total of 72 birds were recorded, with 16 birds seen in the offshore site (raw numbers, all sea states) (Table 4.2). In Year 2, a total of 286 birds were recorded, with 27 birds seen in the offshore site. In Year 3, a total of 179 birds were recorded, with 27 birds seen in the offshore site.

During the Year 1 autumn migration period (July to October), the mean estimated number of Manx shearwaters in the offshore site was 26 birds, with a peak of 75 birds in August (Table 4.22). In the same period of Year 2, the mean estimated number of Manx shearwaters in the offshore site was 12 birds, with 23 birds estimated for both July and September. In Year 3, the mean estimated number of Manx shearwaters in the offshore site was 45 birds, with 107 birds estimated in August.

Table 4.22 Estimated numbers of Manx shearwaters in the offshore site (plus 1 & 2 km buffer) in Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 8 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	0	0	0	0	0	0	0	0
Yr1 Dec	0	0	0	0	0	0	0	0
Yr1 Jan	0	0	0	0	0	0	0	0
Yr1 Feb	0	0	0	0	0	0	0	0
Yr1 Mar	0	0	0	0	0	0	0	0
Yr1 Apr	0	0	0	0	0	0	0	0
Yr1 May	0	0	0	0	0	0	0	45
Yr1 Jun	0	0	0	0	0	0	0	14
Yr1 Jul	0	0	0	0	0	0	0	22
Yr1 Aug	75	32	178	0	75	119	144	165
Yr1 Sep	15	4	54	0	15	15	22	43
Yr1 Oct	15	6	36	0	15	30	44	59
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	0	0	0	0	0	0	0	0
Yr2 Jan	0	0	0	0	0	0	0	0
Yr2 Feb	0	0	0	0	0	0	0	0
Yr2 Mar	0	0	0	0	0	0	0	0
Yr2 Apr	0	0	0	0	0	0	0	14
Yr2 May	0	0	0	0	0	0	0	13
Yr2 Jun	0	0	0	0	0	0	0	22
Yr2 Jul	23	6	89	0	23	23	29	230
Yr2 Aug	0	0	0	0	0	0	0	0
Yr2 Sep	23	5	95	0	23	30	953	1,048
Yr2 Oct	0	0	0	0	0	0	0	92
Yr3 Nov	0	0	0	0	0	0	0	7
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	0	0	0	0	0	0	0	0
Yr3 Feb	0	0	0	0	0	0	0	0
Yr3 Mar	0	0	0	0	0	0	0	0
Yr3 Apr	0	0	0	0	0	0	0	0
Yr3 May	0	0	0	0	0	0	0	0
Yr3 Jun	0	0	0	0	0	0	0	7
Yr3 Jul	59	17	197	14	72	72	96	121
Yr3 Aug	107	43	266	0	107	142	211	1,031
Yr3 Sep	0	0	0	0	0	0	0	0
Yr3 Oct	0	0	0	0	0	0	0	0

Mean monthly estimated numbers of Manx shearwaters in the offshore site and the buffer area, based on three years, are shown in Figure 4.17.

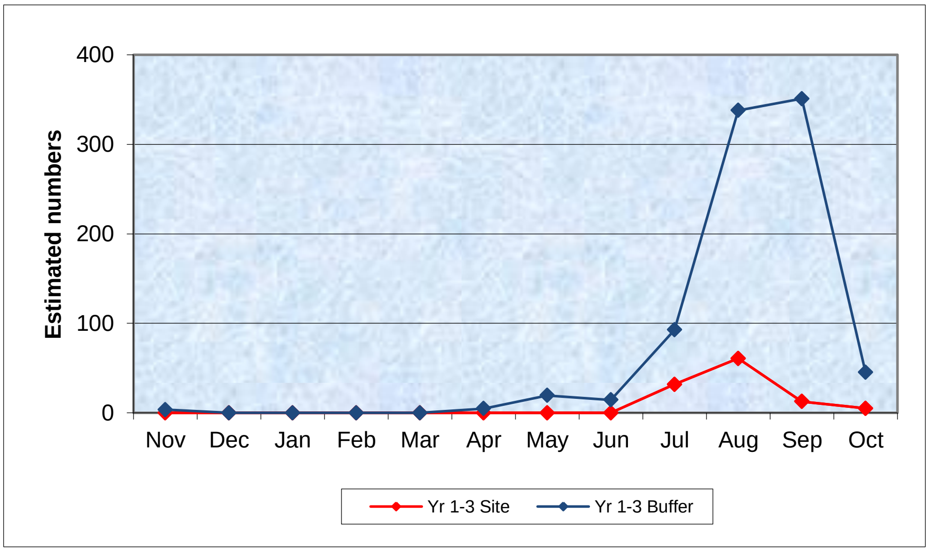


Figure 4.17 Mean monthly estimated numbers of Manx shearwaters in the Neart na Gaoithe Development & buffer areas in Years 1 to 3 (Three-year mean)

Manx shearwaters were scattered across the offshore site and 8 km buffer area in low densities, between July and October of Year 1 (Figure 4.18). Few birds were recorded in the offshore site over the period.

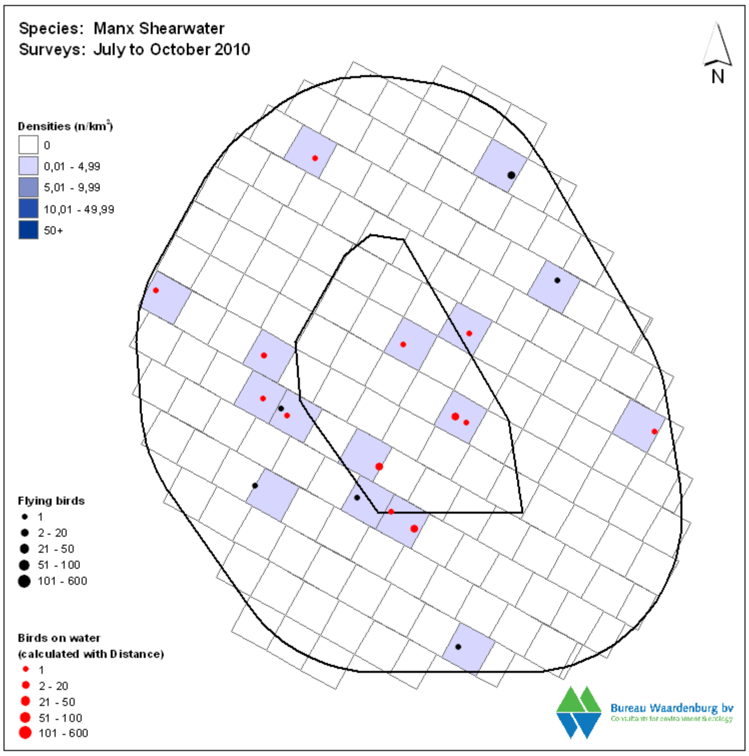


Figure 4.18 Manx shearwater density between July and October, Year 1

In Year 2, Manx shearwaters were scattered across the offshore site and 8 km buffer area at mostly low abundances, between July and October, although few were recorded in the offshore site at this time (Figure 4.19). Highest densities were recorded to the west of the offshore site over the period.

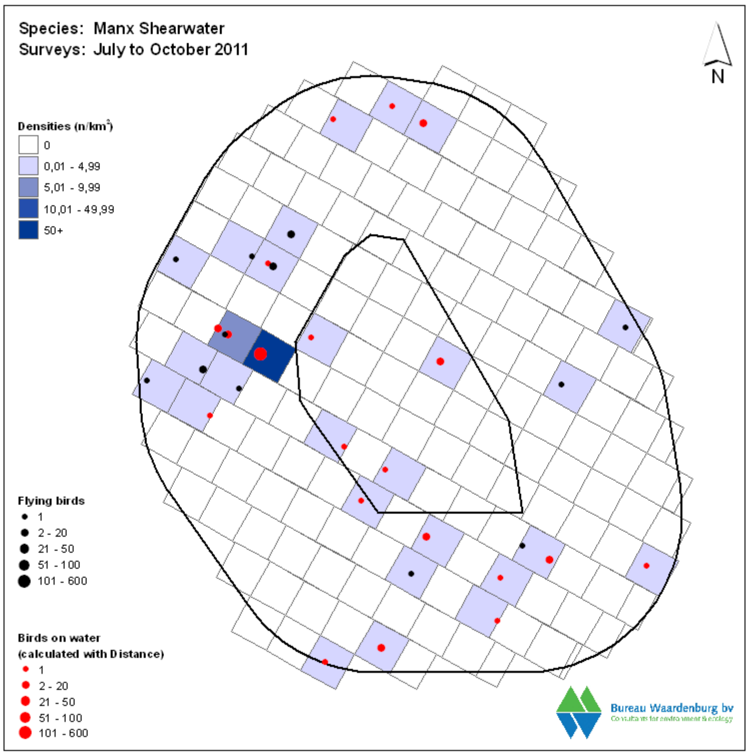


Figure 4.19 Manx shearwater density between July and October, Year 2

In Year 3, Manx shearwaters were scattered across the southern half of theoffshore site and 8 km buffer area at low to moderate densities, between July and October , although few birds were recorded in the offshore site over the period (Figure 4.20). Highest density at this time was recorded to the south of the offshore site.

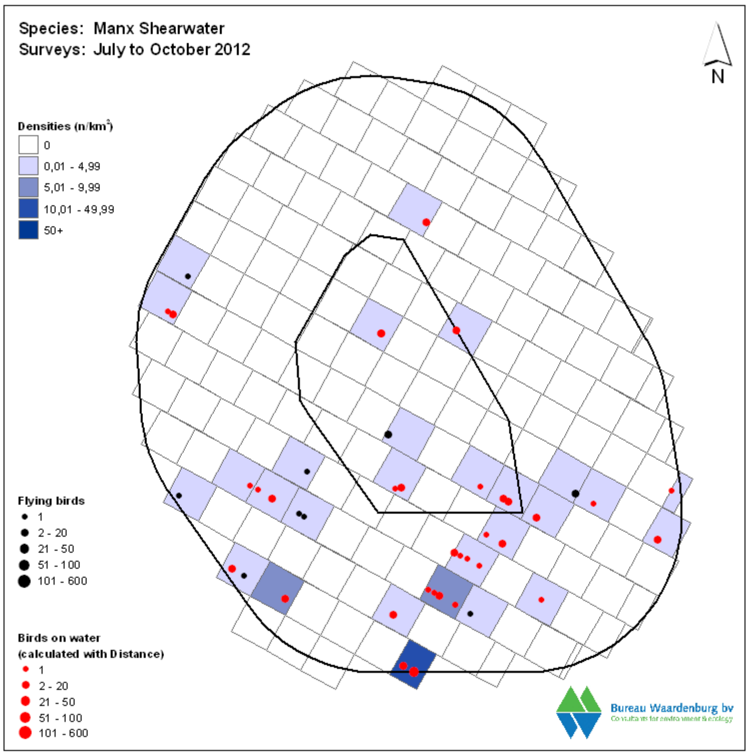


Figure 4.20 Manx shearwater density between July and October, Year 3

A total of 207 Manx shearwaters were recorded in flight on baseline surveys, with all birds flying below 27.5 m in height (Table 4.3). Almost all birds were recorded flying below 7.5 m, with one bird flying between 7.5 m and 12.5 m in height.

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).

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 the offshore site in Years 1 to 3. 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 in the offshore site between July and October, with the majority of birds seen in August and September. 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 Manx shearwaters (330 km) (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 as all birds (n=207) 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 collision mortality on Manx shearwaters was rated as being 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.

Summary of combined effects

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 Manx shearwaters between May and October is negligible. As the population has low sensitivity to all effects, it is concluded that the overall impact on the regional population of Manx shearwaters between May and October is ***not significant*** under the EIA regulations

4.4.4.5 Cumulative Impact Assessment

There were no significant impacts predicted for the regional population of Manx shearwaters in the autumn passage period from the proposed Neart na Gaoithe development. The predicted effects of the development on the regional population of Manx shearwaters in the autumn passage period arising from construction, operation and decommissioning are very close to no effect.

Based on these findings, it was considered that it is not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for this population, and therefore no further cumulative impact assessment was undertaken for this species.

4.4.4.6 Mitigation measures

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

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 Offshore site and 8 km buffer area

One Balearic shearwater was recorded 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 4.21). No Balearic shearwaters were recorded in the Survey Area during Year 2 surveys. One Balearic shearwater was recorded in October of Year 3, flying below 7.5 m in height, in the buffer area (Figure 4.21).

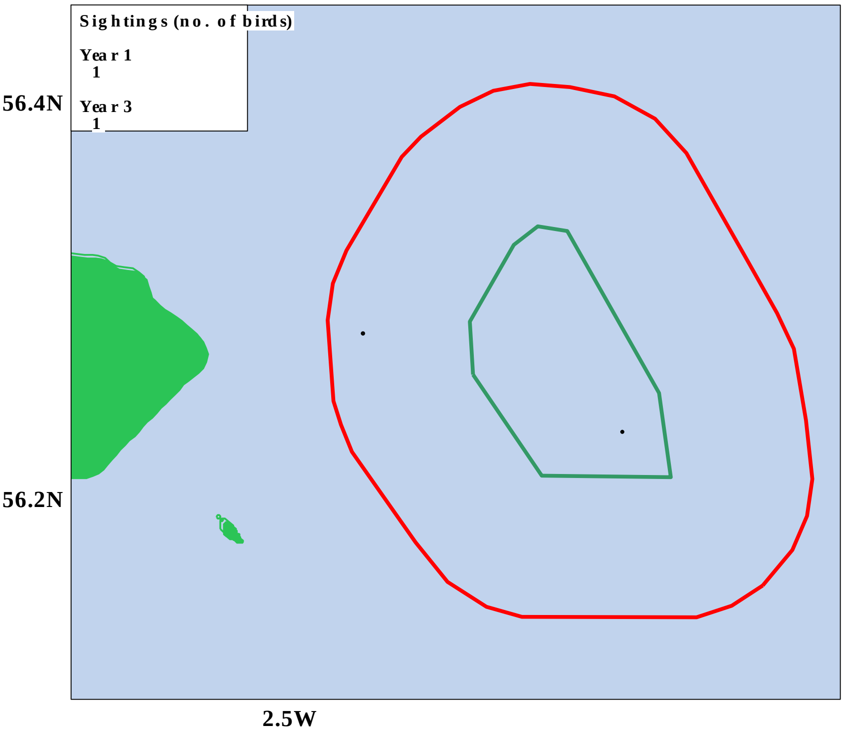


Figure 4.21 Balearic shearwater sightings in Year 1 and 3

4.4.5.3 Assessment

As only one Balearic shearwater was recorded in the offshore site on baseline surveys, 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 Offshore site and 8 km buffer area

One storm petrel was recorded in October of Year 1, in the north-east of the Neart na Gaoithe buffer area, flying north-west, below 7.5 m in height (Table 4.2). The species was not recorded on Year 2 surveys. In Year 3, one storm petrel was recorded flying below 7.5 m in height in June, in the north-east of the buffer area (Figure 4.22).

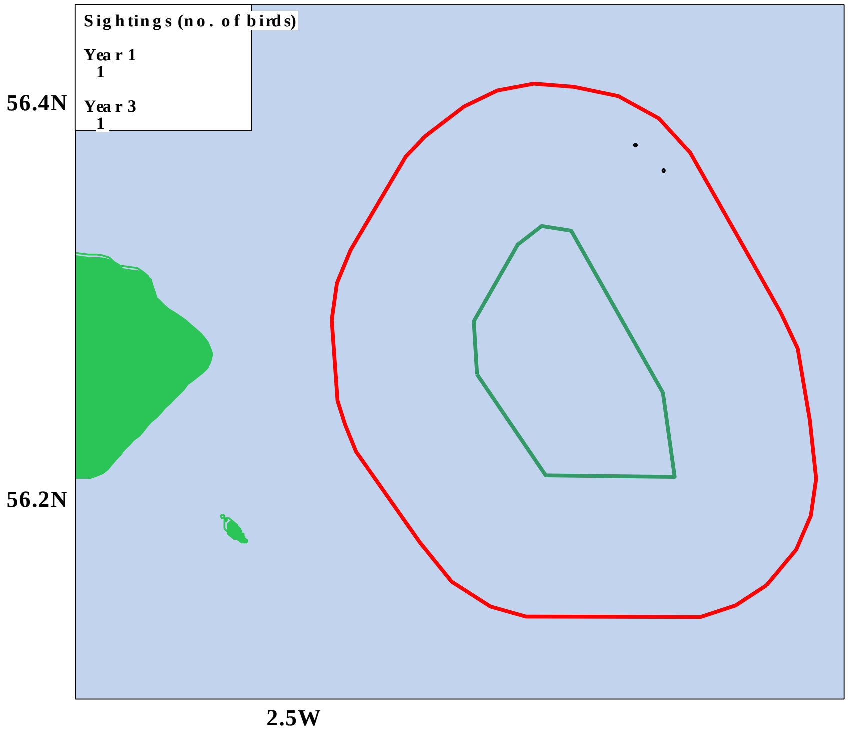


Figure 4.22 Storm petrel sightings in Year 1 and 3

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).

4.4.6.4 Assessment

As storm petrels were not recorded in the offshore site on baseline surveys, and only two birds were recorded in the buffer area in Years 1 and 3, with no birds seen in Year 2, it is considered that storm petrel is unlikely to be affected by the Development.

4.4.1 Northern Gannet *Morus bassanus*

4.4.1.1 Status

Gannets breed in a few, typically very large, colonies around the UK. The UK breeding population is 218,546 pairs (Wanless *et al.,* 2005) and 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).

During the breeding season (April to September) birds from the Bass Rock colony 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 of the colony with concentrations of feeding locations off North-east Scotland (Hamer *et al*., 2011). Outwith the breeding period, gannets disperse widely across the North Sea and move southward with birds wintering in the Bay of Biscay and off West Africa.

Gannets feed by plunge diving for fish, typically from around 25 to 30 m above the surface (BTO, 2012).

4.4.1.2 Offshore site and 8 km buffer area

Gannet was the most frequently recorded seabird during baseline surveys in the offshore site and 8 km buffer area, with a total of 13,021 birds recorded in Year 1, 19,416 birds in Year 2 and 14,825 birds in Year 3 (raw numbers, all sea states). The majority of gannets were recorded in the buffer area. Highest numbers were recorded during the breeding season (April to September) (Table 4.2).

During the Year 1 breeding season (April to September), the mean estimated number of gannets in the offshore site was 286 birds, with a peak of 734 birds in September (Table 4.23). In the Year 2 breeding season, the mean estimated number of gannets in the offshore site was 1,137 birds, with a peak of 2,113 birds in April. In Year 3, the mean estimated number of gannets during the breeding period was 503 birds, with a peak of 611 gannets in September.

In the Year 1 non-breeding season (October to March), the mean estimated number of gannets in the offshore site was 187 birds, with a peak of 477 birds in March (Table 4.23). In the same period of Year 2, the mean number of gannets in the offshore site was 187 birds, with a peak of 410 gannets in October. In Year 3, the mean number of gannets recorded during the non-breeding period was 158, with a peak of 327 birds in October.

Table 4.23 Estimated numbers of gannets in the offshore site (and 1, 2 & 8 km buffer) in Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 8 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	0	0	0	20	20	20	27	435
Yr1 Dec	0	0	0	0	0	9	9	30
Yr1 Jan	0	0	0	7	7	7	7	20
Yr1 Feb	0	0	0	20	20	20	49	394
Yr1 Mar	15	7	33	462	477	504	538	1,562
Yr1 Apr	7	4	14	129	137	137	177	959
Yr1 May	7	2	23	150	157	327	484	1,522
Yr1 Jun	118	55	254	237	355	404	530	1,714
Yr1 Jul	0	0	0	89	89	159	276	1,217
Yr1 Aug	64	33	123	177	241	445	692	3,113
Yr1 Sep	381	277	523	353	734	1,159	1,648	4,081
Yr1 Oct	7	4	14	163	170	248	360	1,234
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	0	0	0	7	7	7	7	77
Yr2 Jan	0	0	0	14	14	75	75	562
Yr2 Feb	35	15	81	374	409	526	696	955
Yr2 Mar	14	6	30	79	93	263	323	1,279
Yr2 Apr	1,936	782	4,790	177	2,113	2,614	2,867	6,060
Yr2 May	15	5	41	800	814	1,003	1,407	4,151
Yr2 Jun	45	22	92	603	648	1,003	1,346	5,918
Yr2 Jul	291	196	432	1,248	1,539	1,764	2,089	4,402
Yr2 Aug	44	21	92	731	775	1,027	1,376	6,009
Yr2 Sep	303	191	483	628	932	1,765	3,113	6,588
Yr2 Oct	106	53	211	305	410	513	626	3,926
Yr3 Nov	109	57	206	42	150	244	850	1,953
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	0	0	0	7	7	55	55	417
Yr3 Feb	16	5	49	21	37	59	97	239
Yr3 Mar	21	4	112	248	269	359	744	2,909
Yr3 Apr	23	10	53	89	113	254	537	2,107
Yr3 May	67	37	122	467	534	769	970	2,838
Yr3 Jun	165	53	518	248	413	606	1,172	5,076
Yr3 Jul	37	15	89	282	319	494	735	4,347
Yr3 Aug	119	47	301	406	524	598	881	3,495
Yr3 Sep	77	50	118	533	611	907	1,200	3,184
Yr3 Oct	67	43	103	260	327	503	670	1,852

Overall, estimated numbers of gannets in the offshore site in Years 1 to 3 were lower than estimated numbers in the buffer area throughout the year (Figure 4.23). Estimated numbers in the offshore site peaked in September, while estimated numbers in the buffer area peaked in June and September.

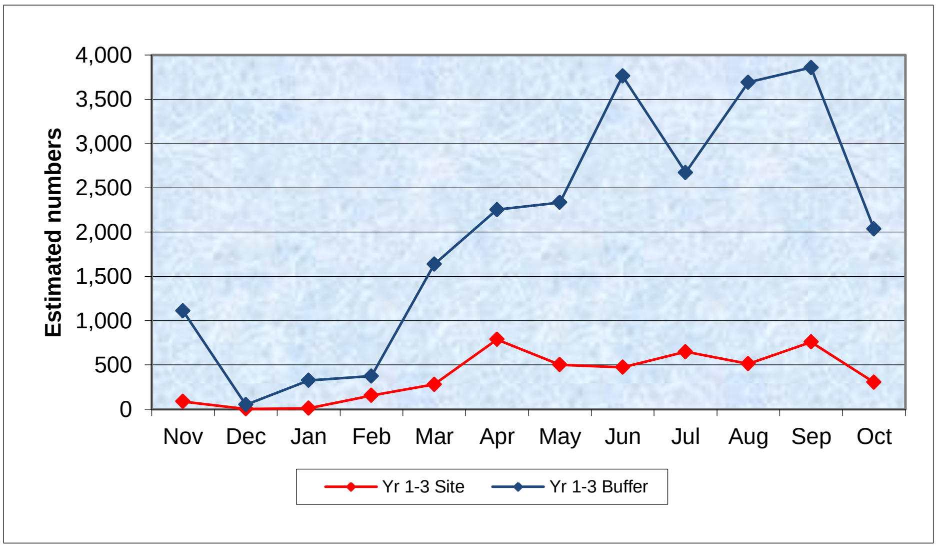


Figure 4.23 Estimated numbers of gannets in the offshore site & buffer areas in Years 1 to 3 (Three-year mean)

In the Year 1 non-breeding season, (October to March), gannets were widespread at mostly low densities across the offshore site and 8 km buffer area (Figure 4.24).

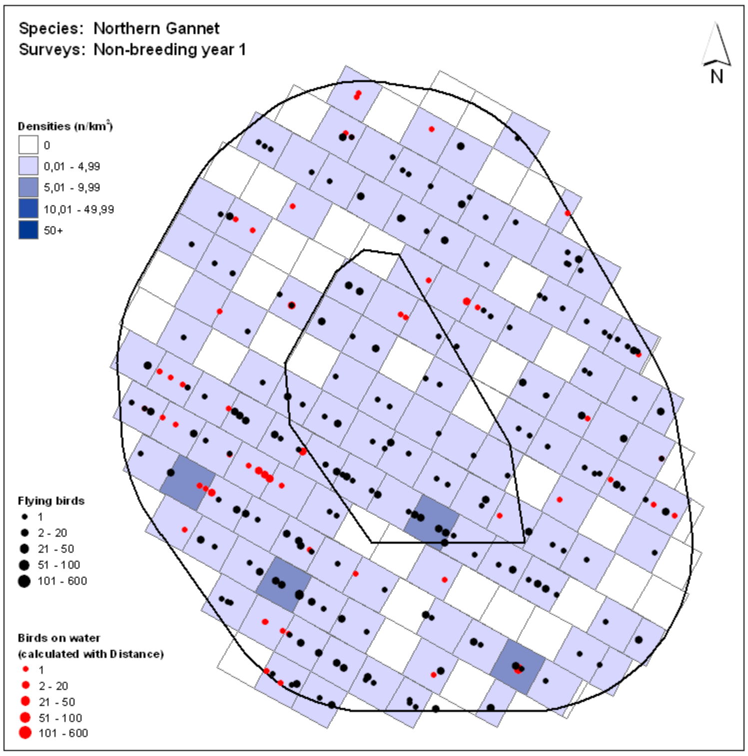


Figure 4.24 Gannet density between October and March, Year 1

Between October and March of Year 2, gannets were slightly more widespread at low to moderate densities across the offshore site and 8 km buffer area than in Year 1 (Figure 4.25). Highest densities were recorded in the south-east of the buffer area at this time.

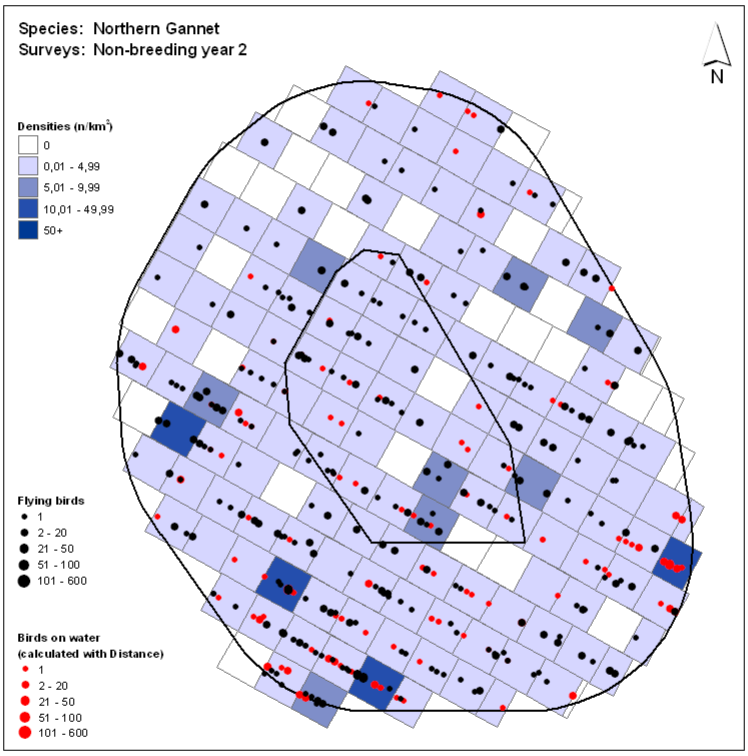


Figure 4.25 Gannet density between October and March, Year 2

Gannet distribution between October and March of Year 3 was similar to the two previous years, with mostly low densities across the offshore site and 8 km buffer area (Figure 4.26).

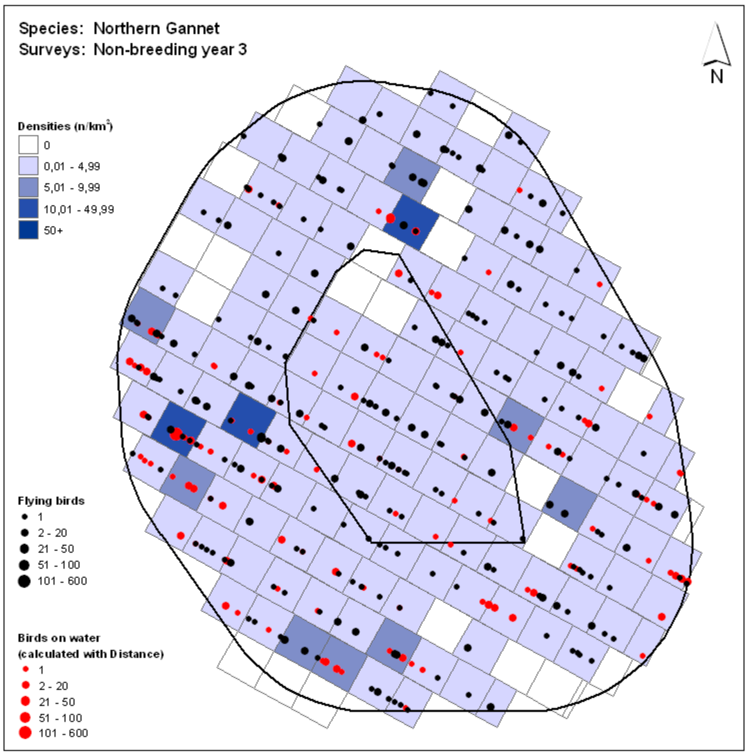


Figure 4.26 Gannet density between October and March, Year 3

Gannets were more numerous and widespread in the Year 1 breeding season (April to September), with low to moderate, occasionally high densities recorded across the offshore site and 8 km buffer area over the period (Figure 4.27).

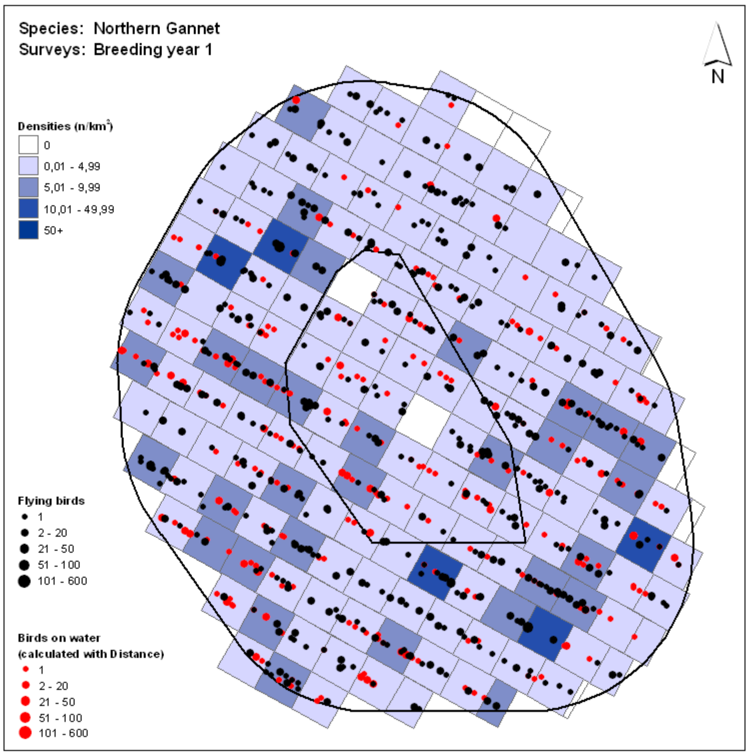


Figure 4.27 Gannet density between April and September, Year 1

Gannet density in the offshore site and 8 km buffer 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 western half of the offshore site and buffer area, and mostly low to moderate densities recorded in the eastern half (Figure 4.28).

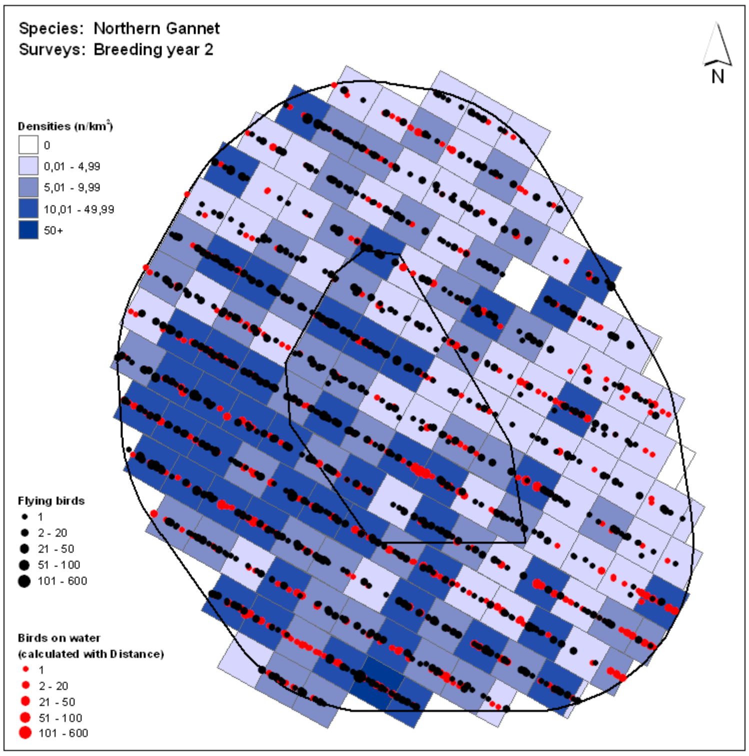


Figure 4.28 Gannet density between April and September, Year 2

Gannet density in the offshore site and 8 km buffer area in the Year 3 breeding season was slightly lower than in the same period in Year 2, with mostly low to moderate densities recorded in the offshore site. Higher densities were recorded in the west and south of the buffer area at this time(Figure 4.29).

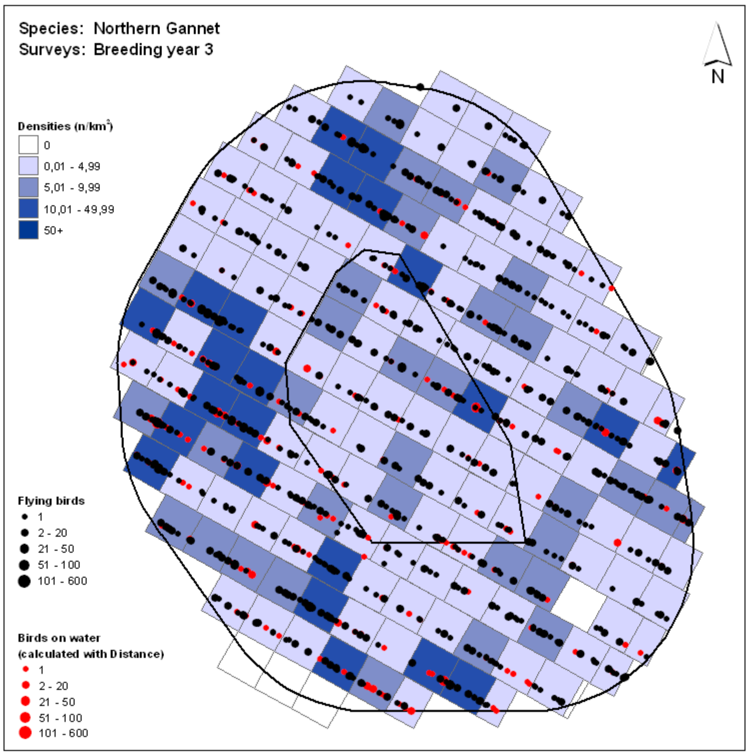
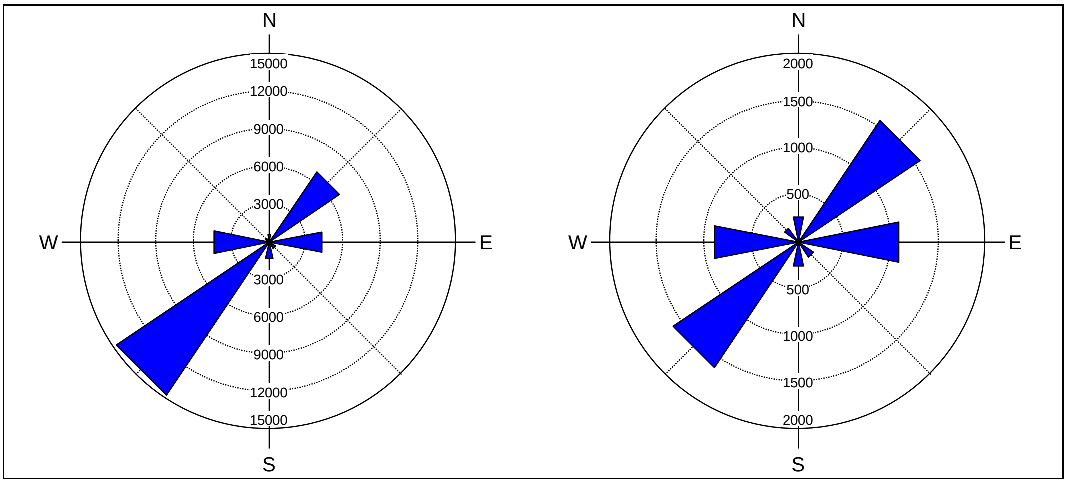


Figure 4.29 Gannet density between April and September, Year 3

A total of 41,250 gannets were recorded in flight on baseline surveys, with 95.2% of birds flying below 27.5 m in height i.e. below the turbine swept zone (Table 4.3). A total of 1,989 birds (4.8%) were recorded flying above 27.5 m, i.e. within the rotor swept zone, with the majority of these birds 95.5% recorded at an estimated 40 m or less. The maximum height estimated was 70 m.



April to September (n=33,053 birds) October to March (n=6,074 birds)
Numbers shown on figures are number of birds recorded

Figure 4.30 Flight direction of gannets in the offshore site and 8 km buffer area in Years 1 to 3

Flight direction was recorded for 33,053 gannets in the breeding season (April to September), with direction recorded for 6,074 birds in the non-breeding season (October to March) (Figure 4.30).

In the breeding season, slightly less than half of all birds recorded were flying south-west (44.4%) in the general direction of the Bass Rock breeding colony, with 20.4% of birds flying north-east. In the non-breeding period, just over a quarter of birds (26.3%) were recorded flying south-west, with 25.7% flying north-east. An additional 2,113 birds were recorded as circling in Years 1 to 3 (not shown).

Foraging behaviour was recorded for 2,073 gannets in the offshore site and 8 km buffer area in Years 1 to 3, with eight types of foraging behaviour recorded, and unspecified feeding behaviour recorded for a further 122 birds (Table 4.24) (Figure 4.31). Deep plunging was the most frequently recorded foraging behaviour (56.8%).

Table 4.24 Gannet foraging behaviour in the offshore site and 8 km buffer area in Years 1 to 3

Behaviour	Number of birds
Actively searching	457
Deep plunging	1,245
Pursuit plunging	40
Shallow plunging	6
Surface seizing	38
Scavenging	1
Scavenging at fishing vessel	284
Holding fish	2
Feeding method unspecified	122
Total	**2,195**

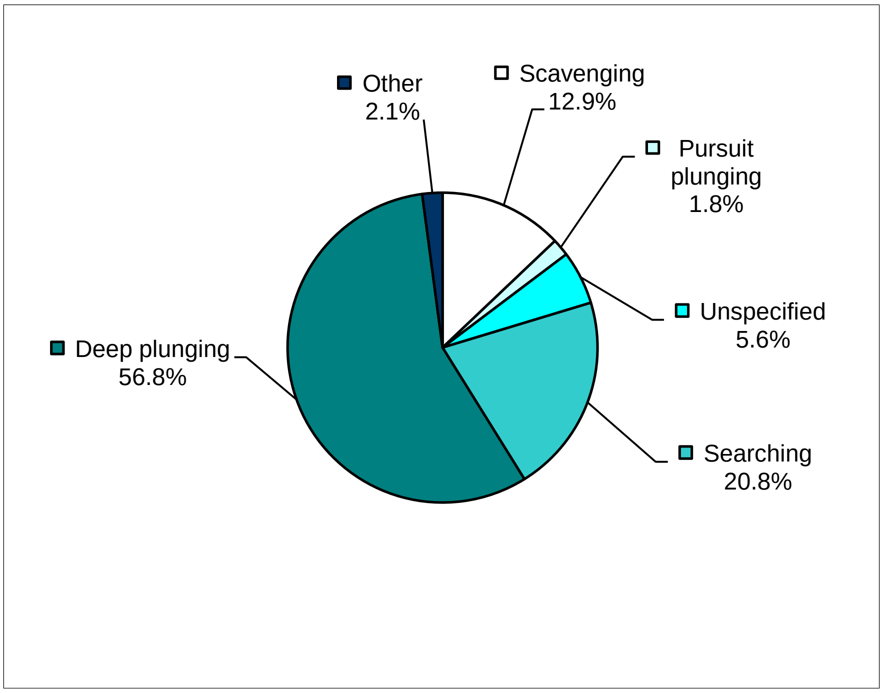


Figure 4.31 Foraging behaviours of gannets in the offshore site and 8 km buffer area in Years 1 to 3

A total of 40,500 gannets were aged on baseline surveys in the offshore site and 8 km buffer area. In the breeding season (April to September) age was recorded for 33,764 gannets, with 877 immature (non-breeding) birds (2.6%) and 32,887 adults (97.4%) aged on surveys (Table 4.25).

Table 4.25 Monthly breakdown of immature and adult gannets in the offshore site and 8 km buffer area in Years 1 to 3 combined

Month	No of immature birds	Number of adult birds	Number of aged birds	Percentage of immature birds
January	18	154	172	10.47
February	5	1,175	1,180	0.42
March	0	2,642	2,642	0
April	8	3,805	3,813	0.21
May	32	5,528	5,560	0.58
June	115	7,829	7,944	1.45
July	123	5,342	5,465	2.25
August	80	5,641	5,721	1.4
September	519	4,742	5,261	9.87
October	81	2,286	2,367	3.42
November	66	263	329	20.06
December	23	23	46	50.0

4.4.1.3 Species sensitivity

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

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 4.26). 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, 2013).

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 4.32.

Table 4.26 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*	*3,862*	*2012*
Forth Islands (Bass Rock)	**16**	**34,400**	**13.1**	**17.1**	**55,482**	**2009** [3]
Flamborough Head & Bempton Cliffs	*259*	*2,501*	*0.95*	*1.2*	*11,061*	*2012*
Hermaness, Saxa Vord & Valla Field	*510*	*12,000*	*4.6*	*6.0*	*24,353*	*2008*
Noss	*428*	*7,310*	*2.8*	*3.6*	*9,767*	*2008*
Total	**-**	**57,377**	**21.9**	**28.5**	**104,525**	**-**

Sources: 1 JNCC (2013) – SPA online species accounts. 2 SMP (2013) – 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).

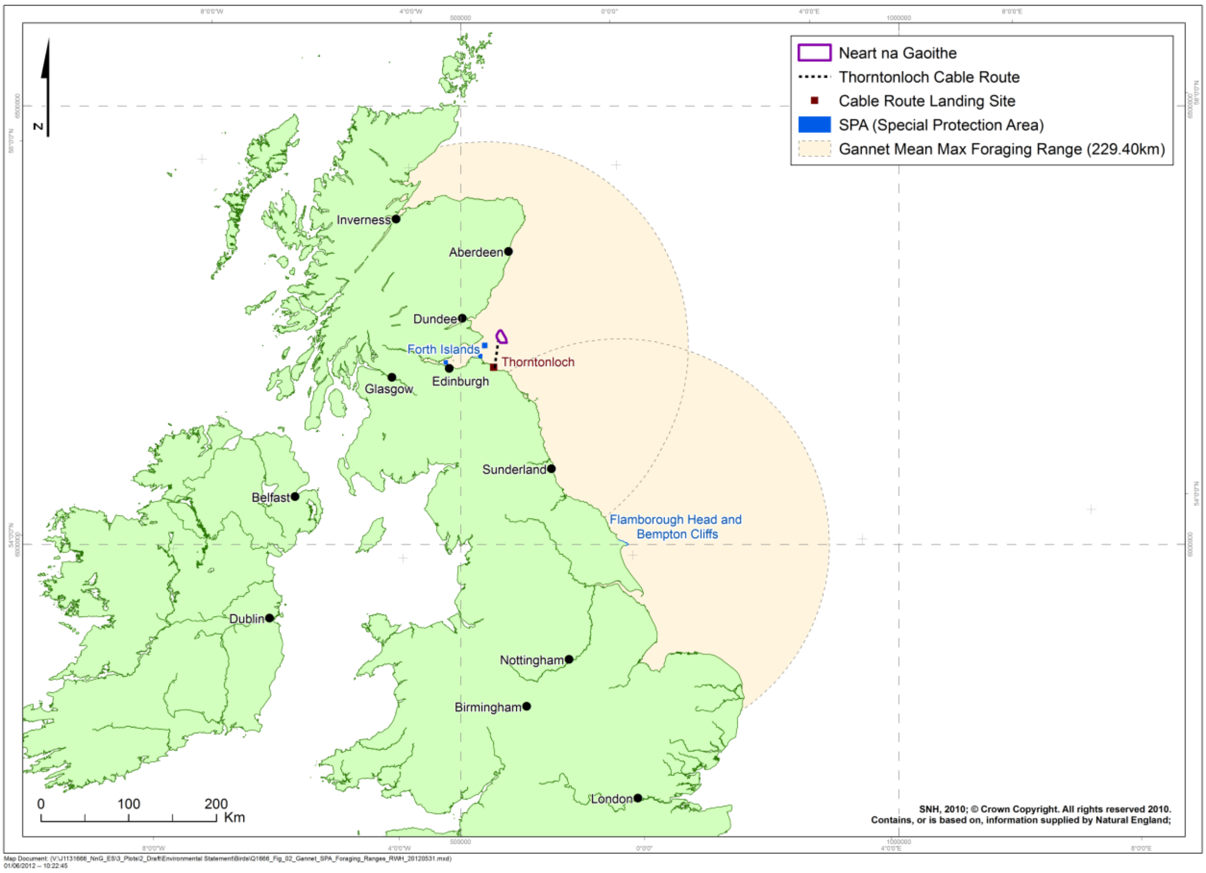


Figure 4.32 Gannet mean maximum foraging range from breeding SPAs in relation to the Development

4.4.1.4 Assessment

Definition of seasons

The annual cycle for gannet was 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, was defined as April 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 was defined as October to March 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).

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).

The size of the non-breeding-period regional gannet population is assumed to be 31,200 birds based on Skov *et al*. (1995). 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

The nature conservation importance (NCI) 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 internationally designated population. The three-year mean peak estimated number of gannets present in the breeding season in the offshore site (1,308 birds) is 1.1% of the regional breeding population (58,269 pairs). The majority of these birds are from the Bass Rock colony, 27 km south-west of the offshore site, which is internationally important for this species, and is a component of the Forth Islands SPA.

Offshore wind farm studies

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).

Raw data for post-construction monitoring at Robin Rigg Offshore Wind Farm in the Solway Firth suggested a displacement rate of 50% for gannets in the first year of operation, however more data is required to complete this analysis (Walls *et al.,* 2013).

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.

Construction Phase

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 some species. 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. However, gannet prey species (such as sandeels, mackerel, whiting, and herring) are typically highly mobile, with a high degree of variability in their seasonal and annual distribution each year.

Gannets are not considered susceptible to disturbance impacts. 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.

Operational Phase

Displacement

In order to assess the potential impacts from displacement, the approach presented in the interim guidance note on displacement issued by the JNCC and NE has been followed (JNCC & NE 2012).

Peak estimated numbers of gannets in the offshore site in the breeding season (April to September) and non-breeding season (October to March) for Years 1 to 3 were averaged to get the three-year mean peak per season. Where peak numbers occurred in different months within the same season across different years, the peak month was used. This was repeated for a 1 km and 2 km buffer around the offshore site (Table 4.27).

Table 4.27 Seasonal three-year mean peak estimated numbers of gannets in the offshore site (plus 1 & 2 km buffer)

Year	Offshore site		Offshore site + 1 km		Offshore site + 2 km	
	Breeding	Non-breeding	Breeding	Non-breeding	Breeding	Non-breeding
Year 1	734	477	1,159	504	1,648	538
Year 2	2,113	410	2,614	526	2,867	696
Year 3	611	327	907	503	1,200	850
3-year mean peak	1,153	405	1,560	511	1,905	695

Guidance recommends presenting a range of potential displacement and mortality rates and wherever possible selecting a suitable impact based on empirical evidence. Where, there is little evidence to support the assessment a precautionary approach should be taken.

Likely impacts of displacement

Studies have shown that the offshore site is within the core foraging areas of gannets from the Bass Rock colony (Hamer *et al.,* 2011). In addition, studies of average gannet foraging range from colonies indicates that all breeding adults in the Study Area in the breeding season are likely to be from the Bass Rock (Lewis *et al.,* 2001).

There is evidence from existing offshore wind farms indicating that gannets avoid flying through wind farms. 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).

Based on evidence from existing wind farms, it was assumed that there will be 90% displacement of gannets from the offshore site in the breeding and non-breeding seasons. Additional scenarios considering 90% displacement out to a 1 km and a 2 km buffer are also presented (Tables 4.30 to 4.35).

Assuming 90% of all gannets were displaced from the offshore site during the breeding season, this would affect an estimated 1,038 birds (Table 4.28), increasing to 1,715 birds including the 2 km buffer (Table 4.30). However, this estimate includes non-breeding immature birds, as well as breeding adults. During the breeding period (April to September), 2.6% of aged gannets were immature birds (Table 4.25). This percentage was applied to the estimated numbers of displaced gannets in the breeding season to estimate the number of adults potentially displaced, as recommended in the draft guidance note on displacement (JNCC & NE, 2012).

Table 4.28 Estimated number of gannets predicted to be at risk of mortality following displacement from offshore site in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	2	6	12	23	35	46	58	69	81	92	104	115
	20%	5	12	23	46	69	92	115	138	161	184	208	231
	30%	7	17	35	69	104	138	173	208	242	277	311	346
	40%	9	23	46	92	138	184	231	277	323	369	415	461
	50%	12	29	58	115	173	231	288	346	404	461	519	577
	60%	14	35	69	138	208	277	346	415	484	553	623	692
	70%	16	40	81	161	242	323	404	484	565	646	726	807
	80%	18	46	92	184	277	369	461	553	646	738	830	922
	90%	**21**	52	104	208	311	415	519	623	726	830	934	1,038
	100%	23	58	115	231	346	461	577	692	807	922	1,038	1,153
Three-year mean peak of 1,153 gannets in the offshore site in the breeding season (April to Sept)													
SPA Population within mean max foraging range (229.4 km) = 55,482 pairs (Murray 2011)													
Regional population within mean max foraging range = 58,269 pairs (SMP 2012)													

Table 4.29 Estimated number of gannets predicted to be at risk of mortality following displacement from offshore site plus 1 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	3	8	16	31	47	62	78	94	109	125	140	156
	20%	6	16	31	62	94	125	156	187	218	250	281	312
	30%	9	23	47	94	140	187	234	281	328	374	421	468
	40%	12	31	62	125	187	250	312	374	437	499	562	624
	50%	16	39	78	156	234	312	390	468	546	624	702	780
	60%	19	47	94	187	281	374	468	562	655	749	842	936
	70%	22	55	109	218	328	437	546	655	764	874	983	1,092
	80%	25	62	125	250	374	499	624	749	874	998	1,123	1,248
	90%	**28**	70	140	281	421	562	702	842	983	1,123	1,264	1,404
	100%	31	78	156	312	468	624	780	936	1,092	1,248	1,404	1,560
Three-year mean peak of 1,560 gannets in the offshore site & 1 km buffer in the breeding season (April to Sept)													
SPA Population within mean max foraging range (229.4 km) = 55,482 pairs (Murray 2011)													
Regional population within mean max foraging range = 58,269 pairs (SMP 2012)													

Table 4.30 Estimated number of gannets predicted to be at risk of mortality following displacement from offshore site plus 2 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	4	10	19	38	57	76	95	114	133	152	171	191
	20%	8	19	38	76	114	152	191	229	267	305	343	381
	30%	11	29	57	114	171	229	286	343	400	457	514	572
	40%	15	38	76	152	229	305	381	457	533	610	686	762
	50%	19	48	95	191	286	381	476	572	667	762	857	953
	60%	23	57	114	229	343	457	572	686	800	914	1,029	1,143
	70%	27	67	133	267	400	533	667	800	933	1,067	1,200	1,334
	80%	30	76	152	305	457	610	762	914	1,067	1,219	1,372	1,524
	90%	**34**	86	171	343	514	686	857	1,029	1,200	1,372	1,543	1,715
	100%	38	95	191	381	572	762	953	1,143	1,334	1,524	1,715	1,905
Three-year mean peak of 1,905 gannets in the offshore site & 2 km buffer in the breeding season (April to Sept)													
SPA Population within mean max foraging range (229.4 km) = 55,482 pairs (Murray 2011)													
Regional population within mean max foraging range = 58,269 pairs (SMP 2012)													

Assuming 2.6% of all birds were non-breeding immature birds, it was estimated that if 90% of all adult gannets were displaced from the offshore site during the breeding season, this would affect an estimated 1,010 birds, increasing to 1,670 birds including the 2 km buffer. For the purposes of this assessment, it was assumed that 2% of all adult gannets displaced from the offshore site during the breeding season (up to 33 birds) would die as a result. It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

For the remaining displaced adults that survived, there could potentially be a detrimental impact on their breeding success, as a result of having to travel further on each trip to forage elsewhere. If it was assumed that this affected up to 1,636 pairs out of the total population of 55,482 pairs, then an estimated 2.9% of the Bass Rock breeding population could potentially be affected.

However, it is likely that gannets can compensate for a moderate amount of displacement by choosing to forage elsewhere. Comparing the distribution of gannets in the study area from Years 1 to 3 at this time of year shows that there is considerable variation between years. Between April and September of Year 1, gannets were present at mostly low densities in the offshore site, with higher densities in the south. Densities in the buffer area at this time was also generally low (Figure 4.27). In the same period for Year 2, gannets were generally present in higher densities across the offshore site and the rest of the study area (Figure 4.28). In the Year 3 breeding period, densities were lower in the offshore site, and gannet distribution was similar to Year 1 (Figure 4.29). These changes in seasonal distribution between years are most likely influenced by changes in the distribution of prey, and show that gannets are not regularly relying on the offshore site exclusively at this time of year.

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. Chick-rearing adults from the Bass Rock forage over a very wide area of the North Sea (> 200,000 km^2), extending as far as Bergen/Viking Bank (SW Norway) in the north and the Frisian Islands (NW Netherlands) in the south. The extent and duration of foraging trips varies between breeding seasons, with prey availability being the key determining factor (Hamer *et al.,* 2011). Gannet prey species are typically highly mobile, with a high degree of variability in their seasonal and annual distribution each year.

The approximate area of foraging habitat that would be lost to gannets through displacement following construction of the offshore site is approximately 105 km^2, increasing to 199 km^2 when a 2 km buffer is applied. When this area was compared to estimated foraging ranges from three breeding seasons, the percentage of habitat lost ranged from 0.05% of the estimated foraging range for the offshore site to 0.2% for the offshore site plus a 2 km buffer (Table 4.31).

Table 4.31 Percentage of area potentially lost to foraging breeding gannets from the Bass Rock through displacement for offshore site and 1 & 2 km buffers

Offshore site plus buffer	Area km^2	Area covered by foraging gannets (95% FKD within this area) [1]	Available area remaining after construction	Area lost through displacement as a % of total estimated foraging area
Satellite telemetry data from 1998				
Offshore Site	105 km^2	96,290 km^2	96,185 km^2	0.10%
Site + 1 km buffer	149 km^2	96,290 km^2	96,141 km^2	0.15%
Site + 2 km buffer	199 km^2	96,290 km^2	96,091 km^2	0.20%
Satellite telemetry data from 2002				
Offshore Site	105 km^2	211,120 km^2	211,015 km^2	0.05%
Site + 1 km buffer	149 km^2	211,120 km^2	210,971 km^2	0.07%
Site + 2 km buffer	199 km^2	211,120 km^2	210,921 km^2	0.09%
Satellite telemetry data from 2003				
Offshore Site	105 km^2	45,890 km^2	45,785 km^2	0.23%
Site + 1 km buffer	149 km^2	45,890 km^2	45,741 km^2	0.32%
Site + 2 km buffer	199 km^2	45,890 km^2	45,691 km^2	0.43%

1 Data from Hamer *et al.,* (2011). FKD – Fixed Kernal Density estimate

For the non-breeding season, all birds (immature and adults) were considered. Assuming 90% of all gannets were displaced from the offshore site during the non-breeding season, this would affect an estimated 365 birds (Table 4.32), increasing to 626 birds including the 2 km buffer (Table 4.34). However, given that gannets are not tied to a colony in the non-breeding season, and are therefore free to forage further afield, any additional mortality arising from displacement from the offshore site is likely to be minimal. Tracking research has shown that more than 80% of tracked gannets from the Bass Rock (n=22 birds) overwintered away from the North Sea, mainly off West Africa and in the Mediterranean Sea (Hamer *et al.,* 2011). For the purposes of this assessment, it was assumed that 2% of all gannets displaced from the offshore site and 2 km buffer during the non-breeding season (up to 13 birds) would die as a result. Such a mortality rate would affect an estimated 0.04% of the regional population in the non-breeding period (31,200 birds) (Skov *et al.,* 1995). It is concluded that the remaining displaced gannets would move to alternative foraging areas over the winter months.

Based on evidence from foraging range studies summarised above (e.g. Hamer *et al.,* 2011), the regional gannet population in the breeding and non-breeding seasons 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 population throughout the year.

This impact is categorised as having 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 gannet population in the breeding and non-breeding seasons are ***not significant*** under the EIA Regulations.

Table 4.32 Estimated number of gannets predicted to be at risk of mortality following displacement from offshore site in the non-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	2	4	8	12	16	20	24	28	32	36	41
	20%	2	4	8	16	24	32	41	49	57	65	73	81
	30%	2	6	12	24	36	49	61	73	85	97	109	122
	40%	3	8	16	32	49	65	81	97	113	130	146	162
	50%	4	10	20	41	61	81	101	122	142	162	182	203
	60%	5	12	24	49	73	97	122	146	170	194	219	243
	70%	6	14	28	57	85	113	142	170	198	227	255	284
	80%	6	16	32	65	97	130	162	194	227	259	292	324
	90%	**7**	18	36	73	109	146	182	219	255	292	328	365
	100%	8	20	41	81	122	162	203	243	284	324	365	405
Three-year mean peak of 405 gannets in the offshore site in the non-breeding season (Oct to Mar)													
SPA breeding population within mean max foraging range (229.4 km) = 55,482 pairs (Murray 2011)													
Regional population in the non-breeding season = 31,200 birds (Skov *et al.*, 1995)													

Table 4.33 Estimated number of gannets predicted to be at risk of mortality following displacement from offshore site plus 1 km buffer in the non-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	3	5	10	15	20	26	31	36	41	46	51
	20%	2	5	10	20	31	41	51	61	72	82	92	102
	30%	3	8	15	31	46	61	77	92	107	123	138	153
	40%	4	10	20	41	61	82	102	123	143	164	184	204
	50%	5	13	26	51	77	102	128	153	179	204	230	256
	60%	6	15	31	61	92	123	153	184	215	245	276	307
	70%	7	18	36	72	107	143	179	215	250	286	322	358
	80%	8	20	41	82	123	164	204	245	286	327	368	409
	90%	**9**	23	46	92	138	184	230	276	322	368	414	460
	100%	10	26	51	102	153	204	256	307	358	409	460	511
Three-year mean peak of 511 gannets in the offshore site & 1 km buffer in the non-breeding season (Oct to Mar)													
SPA breeding population within mean max foraging range (229.4 km) = 55,482 pairs (Murray 2011)													
Regional population in the non-breeding season = 31,200 birds (Skov *et al.*, 1995)													

Table 4.34 Estimated number of gannets predicted to be at risk of mortality following displacement from offshore site plus 2 km buffer in the non-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	3	7	14	21	28	35	42	49	56	63	70
	20%	3	7	14	28	42	56	70	83	97	111	125	139
	30%	4	10	21	42	63	83	104	125	146	167	188	209
	40%	6	14	28	56	83	111	139	167	195	222	250	278
	50%	7	17	35	70	104	139	174	209	243	278	313	348
	60%	8	21	42	83	125	167	209	250	292	334	375	417
	70%	10	24	49	97	146	195	243	292	341	389	438	487
	80%	11	28	56	111	167	222	278	334	389	445	500	556
	90%	**13**	31	63	125	188	250	313	375	438	500	563	626
	100%	14	35	70	139	209	278	348	417	487	556	626	695
Three-year mean peak of 695 gannets in the offshore site & 2 km buffer in the non-breeding season (Oct to Mar)													
SPA breeding population within mean max foraging range (229.4 km) = 55,482 pairs (Murray 2011)													
Regional population in the non-breeding season = 31,200 birds (Skov *et al.*, 1995)													

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 likely 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 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 3.7).

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 on the 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) (Table 4.28). 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). Therefore almost all flights in the direction of the wind farm are likely to have destinations beyond the offshore site. For the purpose of assessment a precautionary 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 3.5% (Table 3.8).

Studies on foraging gannets have shown that they are capable of extending foraging distances in response to distribution of prey, suggesting that birds 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, gannets 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

CRM estimated the number of potential gannet collisions per season for four wind turbine designs scenarios (Table 3.6). The minimum height for the turbine blades above the sea surface for all four options is 27.5 m at mean sea level (MSL).

There is no specific SNH guidance on avoidance rates for seabirds, and therefore their default value of 98.0% is presented in this assessment for comparison purposes However, based on evidence from the Egmond aan Zee wind farm in the Netherlands, gannets that entered the wind farm 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). Such behaviour reduces the risk of potential collision and it is predicted that avoidance rates for gannet will be significantly greater than 98%. An avoidance rate of 99% is therefore considered more realistic, and is used in this assessment.

Data from several different gannet colonies indicate that gannets do not fly at night but rather rest on the surface of the sea (Hamer *et al.,* 2011). The level of nocturnal flight activity used in the model was therefore set as very low.

Highest estimated collisions in the breeding period were for Option 1 (90 x 5 MW turbines). Using a 98% avoidance rate, the number of predicted gannet collisions in the breeding period was 387 birds (Table 4.35). Using the more realistic 99% avoidance rate, the number of predicted gannet collisions in the breeding period was 193 birds (Table 4.36).

Lowest estimated collisions in the breeding period were for Options 3 and 4 (73 x 6.15 MW turbines). Using a 98% avoidance rate, the number of predicted gannet collisions in the breeding period was 292 birds (Table 4.35). Using the more realistic 99% avoidance rate, the number of predicted gannet collisions in the breeding period was 146 birds (Table 4.36).

Table 4.35 Number of estimated gannet collisions using avoidance rate of 98% for four wind farm options

	Option 1 (90x5MW)	Option 2 (75x6MW)	Option 3 (73x6.15MW)	Option 4 (73x6.15MW)
Collisions in breeding season (April to September), all ages	387	327	292	292
Collisions in breeding season (April to September), adults only	377	318	284	284
Collisions in non-breeding period (October to March), all ages	86	72	66	66
Total collisions per year, all ages	473	399	357	357

Table 4.36 Number of estimated gannet collisions using avoidance rate of 99% for four wind farm options

	Option 1 (90x5MW)	Option 2 (75x6MW)	Option 3 (73x6.15MW)	Option 4 (73x6.15MW)
Collisions in breeding season (April to September), all ages	193	163	146	146
Collisions in breeding season (April to September), adults only	188	159	142	142
Collisions in non-breeding period (October to March), all ages	43	36	32	32
Total collisions per year, all ages	237	200	178	179

Baseline surveys between April and September recorded the age for a total of 33,764 gannets, with 877 immature (non-breeding) birds (2.7%) and 32,887 adults (97.4%) (Table 4.25). The estimated number of collisions for the breeding period were reduced by 2.6% so that they represented the estimated numbers adult gannets involved with collisions.

Based on this ratio and using a 99% avoidance rate, the predicted number of adult gannet collisions in the breeding season for the worst case design option (Option 1) was 188 birds (Table 4.36). This equates to approximately 0.16% of the regional population (58,269 pairs), or 0.17% of the regional SPA population (55,482 pairs) (Murray 2011) in the breeding period.

In the non-breeding period (October to March) and using a 99% avoidance rate, the predicted number of gannet collisions for the worst case design option (Option 1) was 43 birds (Table 4.36). This equates to approximately 0.14% of the regional population in the non-breeding period (31,200 birds) (Skov *et al.,* 1995).

The breeding gannet population on the Bass Rock 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, (largely due to lack of space at the colony), it is clear that recruitment of immature birds into the breeding population still 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 additional adult mortality would have to be in the order of 2,000 birds per year to lead to a population decline in the Bass Rock gannet population (WWT, 2011).

Taking into consideration that 99% is likely to be much closer to the true avoidance rate than 98%, yet still being precautionary, it is concluded that for the most adverse design (Option 1: 90 x 5 MW turbines), collision mortality for gannet is an effect of negligible magnitude, that is temporally long-term and reversible.

It is further concluded that for the least adverse designs (Options 3 and 4: 73 x 6.15MW turbines), collision mortality for gannet is an effect of negligible magnitude, that is temporally long-term and reversible.

It is concluded that for the wind farm designs examined here, the effects of collision mortality on gannets from the regional population in the breeding and non-breeding periods are ***not significant*** under 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.

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 gannets to displacement it is concluded that the impacts of disturbance by vessels during the construction, operation and decommissioning phases on the regional population of gannets in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

Decommissioning Phase

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 gannets to displacement it is concluded that the impacts of displacement during decommissioning operations on the regional population of gannets in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

Summary of combined effects

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. However, because the population has low sensitivity to all effects except collision, it is judged that the overall impact on the population is not significant under the EIA regulations (Table 4.37 and Table 4.38).

Table 4.37 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	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	High	Not significant
All effects combined	Negligible	Long term	Moderate	Not significant

Table 4.38 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	Low	Not significant
Vessel disturbance	Negligible	Long term	Low	Not significant
Collision mortality	Negligible	Long term	High	Not significant
All effects combined	Negligible	Long term	Moderate	Not significant

4.4.1.5 Cumulative Impact Assessment

Displacement

The potential cumulative displacement risk to gannets from Neart na Gaoithe and other plans or projects is higher than for Neart na Gaoithe alone.

Projects identified during consultation and the undertaking of the EIA for which there is a potential for a cumulative displacement impact are:

- Inch Cape offshore wind farm;
- Seagreen Project Alpha offshore wind farm; and
- Seagreen Project Bravo offshore wind farm.

Evidence from existing offshore wind farms support the potential for a high, but not total, level of displacement behaviour (e.g. Leopold et al. 2011) and a predicted 90% displacement effect has been used in this assessment.

Data presented in the Seagreen HRA (Seagreen 2012) report peak numbers of gannets in the Seagreen Project Alpha site boundary during the breeding period. In 2010 the peak number within the site boundary was during June with 2,716 birds and in 2011 peak numbers occurred in May with 1,841 birds. This provides a peak mean across two years, of 2,279 gannets during the breeding period within the Seagreen Project Alpha boundary.

Assuming 90% of all gannets were displaced from the Seagreen project Alpha and a 2 km buffer during the breeding season, this would affect an estimated 2,051 birds (Table 4.39). Assuming 3.3% of all birds were non-breeding immature birds (Seagreen 2012), it was estimated that this would affect an estimated 1,983 adult gannets.

Assuming that there is the potential for up to 2% rate of mortality during the breeding period then up to 41 gannets may be impacted by displacement from Seagreen project Alpha and a 2 km buffer (Table 4.39). Assuming 3.3% of all birds were non-breeding immature birds (Seagreen 2012), it was estimated that this would affect an estimated 40 adult gannets.

Table 4.39 Estimated number of gannets predicted to be at risk of mortality following displacement from Seagreen Project Alpha plus 2 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	5	11	23	46	68	91	114	137	160	182	205	228
	20%	9	23	46	91	137	182	228	273	319	365	410	456
	30%	14	34	68	137	205	273	342	410	479	547	615	684
	40%	18	46	91	182	273	365	456	547	638	729	820	912
	50%	23	57	114	228	342	456	570	684	798	912	1,026	1,140
	60%	27	68	137	273	410	547	684	820	957	1,094	1,231	1,367
	70%	32	80	160	319	479	638	798	957	1,117	1,276	1,436	1,595
	80%	36	91	182	365	547	729	912	1,094	1,276	1,459	1,641	1,823
	90%	**41**	103	205	410	615	820	1,026	1,231	1,436	1,641	1,846	2,051
	100%	46	114	228	456	684	912	1,140	1,367	1,595	1,823	2,051	2,279
Two-year mean peak of 2,279 gannets in the Seagreen Project Alpha & 2 km buffer in the breeding season (April to Sept)													
SPA Population within mean max foraging range (229.4 km) = 55,482 pairs (Murray 2011)													
Regional population within mean max foraging range = 58,269 pairs (SMP 2012)													

Data presented in the Seagreen Environmental Statement (Seagreen 2012) report peak numbers of gannets in the Seagreen Project Bravo during the breeding period. In 2010 the peak number within the site boundary was during August with 1,141 birds and in 2011 peak numbers occurred in June with 854 birds. This provides a peak mean of 998 gannets during the breeding period within the Seagreen Project Bravo boundary.

Assuming 90% of all gannets were displaced from the Seagreen project Bravo and a 2 km buffer during the breeding season, this would affect an estimated 898 birds (Table 4.40). Assuming 2.2% of all birds were non-breeding immature birds (Seagreen 2012), it was estimated that this would affect an estimated 878 adult gannets.

Assuming that there is the potential for up to 2% rate of mortality during the breeding period then up to 18 gannets may be impacted by displacement from Seagreen project Bravo and a 2 km buffer (Table 4.40). Assuming 2.2% of all birds were non-breeding immature birds (Seagreen 2012), it was estimated that this would affect an estimated 18 adult gannets.

Table 4.40 Estimated number of gannets predicted to be at risk of mortality following displacement from Seagreen Project Bravo plus 2 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	2	5	10	20	30	40	50	60	70	80	90	100
	20%	4	10	20	40	60	80	100	120	140	160	180	200
	30%	6	15	30	60	90	120	150	180	210	240	269	299
	40%	8	20	40	80	120	160	200	240	279	319	359	399
	50%	10	25	50	100	150	200	250	299	349	399	449	499
	60%	12	30	60	120	180	240	299	359	419	479	539	599
	70%	14	35	70	140	210	279	349	419	489	559	629	699
	80%	16	40	80	160	240	319	399	479	559	639	719	798
	90%	**18**	45	90	180	269	359	449	539	629	719	808	898
	100%	20	50	100	200	299	399	499	599	699	798	898	998
Two-year mean peak of 998 gannets in the Seagreen Project Bravo & 2 km buffer in the breeding season (April to Sept)													
SPA Population within mean max foraging range (229.4 km) = 55,482 pairs (Murray 2011)													
Regional population within mean max foraging range = 58,269 pairs (SMP 2012)													

Based on the above, the predicted cumulative displacement impacts on adult gannets from Neart na Gaoithe and Seagreen projects Alpha and Bravo and a 2 km buffer in the breeding period involve a predicted 4,531 adult birds, with 2% mortality of 93 adults. The 2% mortality equates to approximately 0.08% of the regional population (58,269 pairs), or 0.08% of the regional SPA population (55,482 pairs) (Murray 2011) in the breeding period.

The predicted number of adult gannets from the Bass Rock (Forth Islands SPA) displaced from the Inch Cape site during the breeding season is 968 birds (ICOL, 2013).

Assuming 90% of all gannets were displaced from the Inch Cape site during the breeding season, this would affect an estimated 871 birds. Assuming that there is the potential for up to 2% rate of mortality during the breeding period then up to 17 adult gannets may be impacted by displacement from the Inch Cape site.

Based on the above, the predicted cumulative displacement impacts on adult gannets from Neart na Gaoithe and a 2 km buffer and the Inch Cape site in the breeding period involve a predicted 2,541 adult birds, with 2% mortality of 50 adults. The 2% mortality equates to approximately 0.04% of the regional population (58,269 pairs), or 0.05% of the regional SPA population (55,482 pairs) (Murray 2011) in the breeding period.

Based on the predicted numbers reported for the four proposed development areas a total of 5,402 adult gannets may be displaced from the development sites and a 2 km buffer during the breeding period, with 2% mortality of 110 adult birds. The 2% mortality equates to approximately 0.09% of the regional population (58,269 pairs), or 0.1% of the regional SPA population (55,482 pairs) (Murray 2011) in the breeding period.

Based on evidence from foraging range studies summarised above (e.g. Hamer *et al.,* 2011), the regional gannet population in the breeding season is considered to have low sensitivity to displacement effects and it is therefore unlikely that the predicted cumulative displacement will result in any discernible population effects on the regional population in the breeding season.

This impact is categorised as having negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative effects of displacement on the regional gannet population in the breeding season are ***not significant*** under the EIA Regulations.

A separate CIA for displacement has not been undertaken for the regional gannet population in the non-breeding-period. 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.

Collision mortality

The potential cumulative collision risk to gannets from Neart na Gaoithe and other plans or projects is higher than for Neart na Gaoithe alone.

Projects identified during consultation and the undertaking of the EIA for which there is a potential for a cumulative collision risk impact are:

- Inch Cape offshore wind farm;
- Seagreen Project Alpha offshore wind farm; and
- Seagreen Project Bravo offshore wind farm.

Data on the number of predicted collisions arising from the Seagreen Projects Alpha and Bravo offshore wind farms are presented within the applicant's Environmental Statement (Seagreen 2012) and HRA (Seagreen 2013). The proposed Seagreen Round 3 Zone development consists of a three phase programme of six separate developments. Projects Alpha and Bravo are the first developments and the only Seagreen projects for which applications have been made.

No application has been made for proposed Inch Cape offshore wind farm. Unpublished information on the predicted number of collisions arising from the Inch Cape offshore wind farm was obtained in communications with ICOL.

The predicted cumulative collision impacts on gannets from Neart na Gaoithe and Seagreen projects Alpha and Bravo in the breeding and non-breeding periods are presented in Table 4.41.

Table 4.41 Predicted number of cumulative gannet (adult & immature) mortality with Seagreen Projects Alpha and Bravo in the breeding and non-breeding periods

Wind Farm	Breeding	Non-breeding
Neart na Gaoithe [1]		
98% avoidance	387	86
99% avoidance	193	43
Seagreen (Project Alpha) [2]		
98% avoidance	904	100
99% avoidance	416[-3]	50
Seagreen (Project Bravo) [2]		
98% avoidance	552	109
99% avoidance	257[-3]	55
Total[-4]		
98% avoidance	1,843	295
99% avoidance	866	148

1 Based on 3 years data. 2 Based on 2 years data 3 Adults only (Seagreen 2013)

Using a 99% avoidance behavioural rate, a total of 866 gannets are predicted to collide with Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms during the breeding period.

Of those birds that were aged during the breeding period, 2.6% of gannets recorded at Neart na Gaoithe were immature birds.

Accounting for the proportion of non-breeding immature birds recorded during the breeding period the total number of adult gannets at risk of collision during the breeding period at Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms is 861 birds. This equates to approximately 0.7% of the regional population (58,269 pairs), or 0.8% of the regional SPA population (55,482 pairs) (Murray 2011) in the breeding period.

Using a 99% avoidance behavioural rate, a total of 148 gannets are predicted to collide with Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms in the non-breeding period. This equates to approximately 0.5% of the regional population in the non-breeding period (31,200 birds) (Skov *et al.,* 1995).

The predicted cumulative collision impacts on gannets from Neart na Gaoithe and Inch Cape in the breeding and non-breeding periods are presented in Table 4.42.

Table 4.42 Predicted number of cumulative gannet (adult & immature) mortality with Inch Cape in the breeding and non-breeding periods

Wind Farm	Breeding	Non-breeding
Neart na Gaoithe [1]		
98% avoidance	387	86
99% avoidance	193	43
Inch Cape [2]		
98% avoidance	630 [3]	Not available
99% avoidance	315 [3]	Not available
Total [4]		
98% avoidance	1,017	381
99% avoidance	508	186

1 Based on 3 years data. 2 Based on 1 years data 3 Adults only (ICOL, 2013)

Using a 99% avoidance behavioural rate, a total of 508 gannets are predicted to collide with Neart na Gaoithe and Inch Cape offshore wind farms during the breeding period.

The Inch Cape figures included above are for adult birds only. Of those that were aged during the breeding period at Neart na Gaoithe, 2.6% were immature birds. Accounting for the proportion of non-breeding immature birds recorded during the breeding period the total number of adult gannets at risk of collision during the breeding period at Neart na Gaoithe and Inch Cape offshore wind farms is 503 birds. This equates to approximately 0.4% of the regional population (58,269 pairs), or 0.5% of the regional SPA population (55,482 pairs) (Murray 2011) in the breeding period.

Figures for gannet collisions for Inch Cape in the non-breeding season were not available.

The predicted cumulative impacts on gannets from Neart na Gaoithe, Seagreen projects Alpha and Bravo and Inch Cape in the breeding and non-breeding periods are presented in Table 4.43.

Accounting for the proportion of non-breeding immature birds recorded during the breeding period the total number of adult gannets at risk of collision during the breeding period at Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms is 1,176 birds. This equates to approximately 1.0% of the regional population (58,269 pairs), or 1.1% of the regional SPA population (55,482 pairs) (Murray 2011) in the breeding period.

Taking into consideration that 99% is likely to be much closer to the true avoidance rate than 98%, yet still being precautionary, it is concluded that the cumulative collision mortality for the regional gannet population in the breeding period is an effect of low magnitude, that is temporally long-term and reversible.

Table 4.43 Predicted number of cumulative gannet (adult & immature) mortality with Seagreen Projects Alpha and Bravo and Inch Cape in the breeding and non-breeding periods

Wind Farm	Breeding	Non-breeding
Neart na Gaoithe [1]		
98% avoidance	387	86
99% avoidance	193	43
Seagreen (Project Alpha) [2]		
98% avoidance	904	100
99% avoidance	416 [3]	50
Seagreen (Project Bravo) [2]		
98% avoidance	552	109
99% avoidance	257 [3]	55
Inch Cape [2]		
98% avoidance	630 [3]	Not available
99% avoidance	315 [3]	Not available
Total [4]		
98% avoidance	2,473	295+
99% avoidance	1,181	148+

1 Based on 3 years data. 2 Based on 2 years data 3 Adults only

Figures for gannet collisions for Inch Cape in the non-breeding season were not available. However, assuming that predicted collisions for Inch Cape were similar to Neart na Gaoithe, a figure of 43 birds was used for Inch Cape. Using a 99% avoidance behavioural rate, an estimated total of 191 gannets are predicted to collide with Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms in the non-breeding period. This equates to approximately 0.6% of the regional population in the non-breeding period (31,200 birds) (Skov *et al.,* 1995).

It is further concluded that the cumulative collision mortality for the regional gannet population in the non-breeding period is an effect of negligible magnitude, that is temporally long-term and reversible.

It is concluded that for the four projects examined here, the effects of cumulative collision mortality on gannets from the regional population in the breeding and non-breeding periods are ***not significant*** under the Electricity Regulations.

4.4.1.6 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.

4.4.2 Cormorant *Phalacrocorax carbo*

4.4.2.1 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).

4.4.2.2 Offshore site and 8 km buffer area

The species was not recorded in the offshore site on baseline surveys. In the buffer area, one cormorant was recorded in March of Year 1, with three birds in September of Year 2, and three in February of Year 3 (Table 4.2) (Figure 4.33). All seven birds were flying below 27.5 m.

It is likely that water depth in the offshore site and 8 km buffer area is not optimal for foraging cormorants, hence the lack of records of this species.

4.4.2.3 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).

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 4.44). 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).

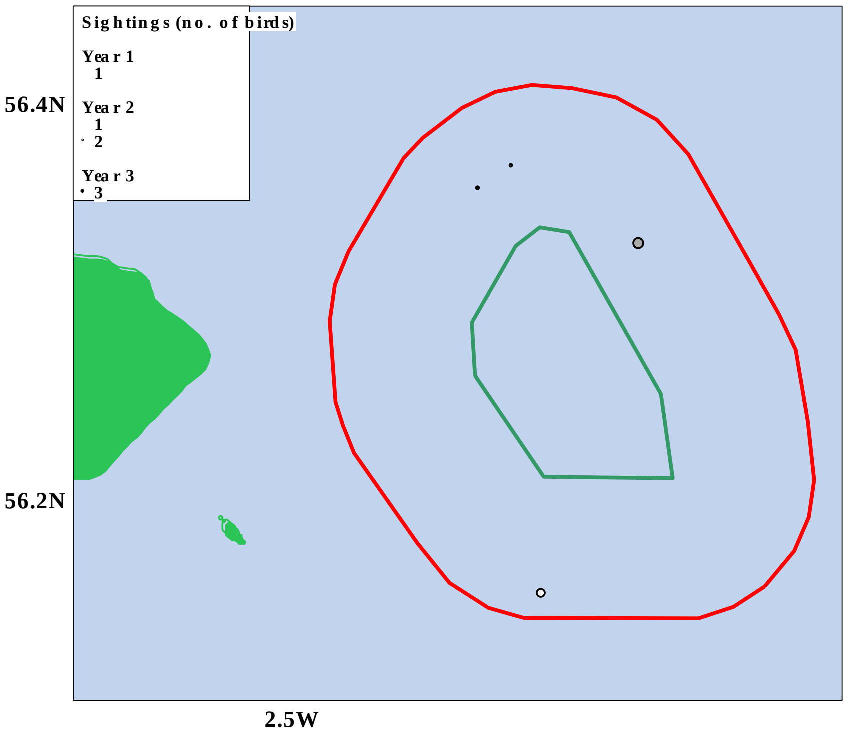


Figure 4.33 Cormorant sightings in Years 1 to 3

Table 4.44 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**	**132**	**2011**
Farne Islands	72	194	0.5	2.8	135	2012
Total	-	**434**	**1.1**	**6.2**	**267**	-

Sources: 1 JNCC (2013) – SPA online species accounts. 2 SMP (2013) – 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 4.45). These SPAs held 7.0% of the UK non-breeding population, and 0.8% of the biogeographic population at the time of designation (JNCC, 2013). Recent five-year means are also shown for comparison, where available.

Table 4.45 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

4.4.2.4 Assessment

The water depth in the offshore site is between 40 and 60 m (Harker & Buse 2009) and is therefore likely to be too deep to attract foraging cormorants from the Firth of Forth SPA population. The results from the baseline surveys, with no cormorants recorded within the offshore site in Years 1 to 3, together with the foraging ecology of this species indicate that cormorants are unlikely to be affected by the development.

4.4.3 European Shag *Phalacrocorax aristotelis*

4.4.3.1 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.

4.4.3.2 Offshore site and 8 km buffer area

There were no shags recorded in the offshore site on baseline surveys (Table 4.2). In the buffer area, 11 shags were recorded in Year 1, with four birds in March, three in April and four in October. In Year 2, two shags were seen in February and four in March. In Year 3, there were two shags in March, and one in May.

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 (Figure 4.34). Shags were more scattered across the buffer area in Year 2. Shag distribution in Year 3 was similar to Year 1, with all three sightings in the north-western sector of the buffer area.

A total of 17 shags were recorded in flight on baseline surveys, with all birds recorded flying below 27.5 m.

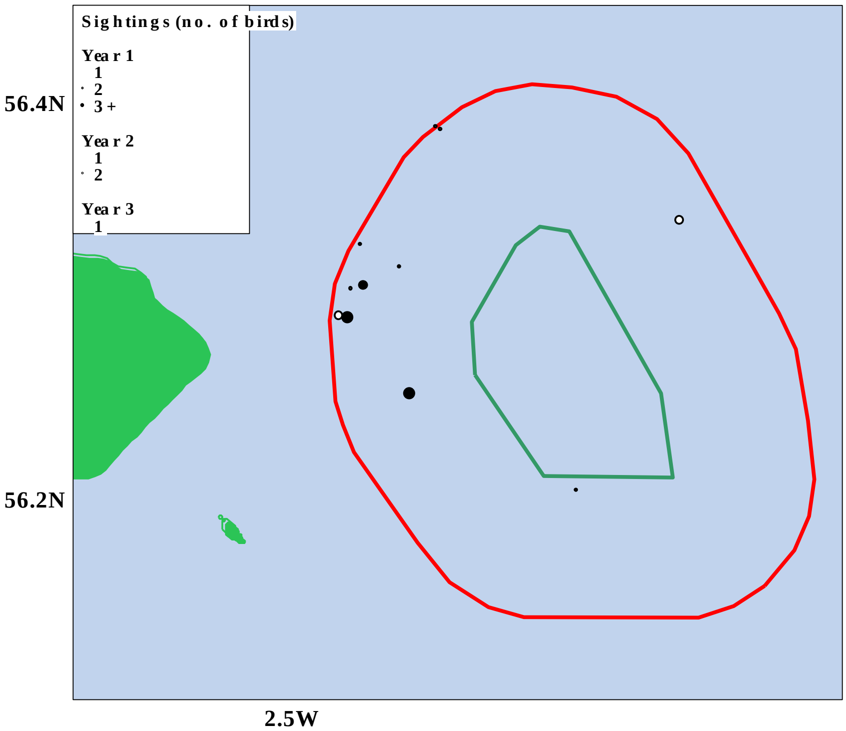


Figure 4.34 Shag sightings in Years 1 to 3

4.4.3.3 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).

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 4.46). These SPAs held 14.9% of the UK breeding population and 4.4% of the biogeographic population at the time of designation (JNCC, 2013). 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 4.46 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	965	2012
Firth of Forth Islands	*16*	*2,887*	*2.3*	*7.7*	*1,063*	*2012*
St Abb's Head to Fast Castle	31	651	0.5	1.7	329	2000
Total	-	**5,577**	**4.4**	**14.9**	**2,701**	-

Sources: 1 JNCC (2013) – SPA online species accounts. 2 SMP (2013) – 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).

4.4.3.4 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 the Year 1 and Year 3 baseline surveys corroborates this, with all shags recorded in the shallower western sector of the buffer area, although birds were more scattered across the buffer area in Year 2. The results from the baseline surveys, with no shags recorded within the offshore site in Years 1 to 3, together with the foraging ecology of this species indicate that cormorants are unlikely to be affected by the Development.

4.4.4 Common Eider *Somateria mollissima*

4.4.4.1 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).

4.4.4.2 Offshore site and 8 km buffer area

Low numbers of eider were recorded in the offshore site and 8 km buffer 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 4.35). Just two eider were recorded in Year 2, in the buffer area in January. In Year 3, seven eider were recorded in the buffer area, between November and February.

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 baseline surveys, as well as published information on the species distribution in the region (Barton and Pollock 2004, Söhle *et al*., 2007, Stone *et al*., 1995).

A total of 29 eider were recorded in flight on baseline surveys, all below 27.5 m in height (Table 4.3).

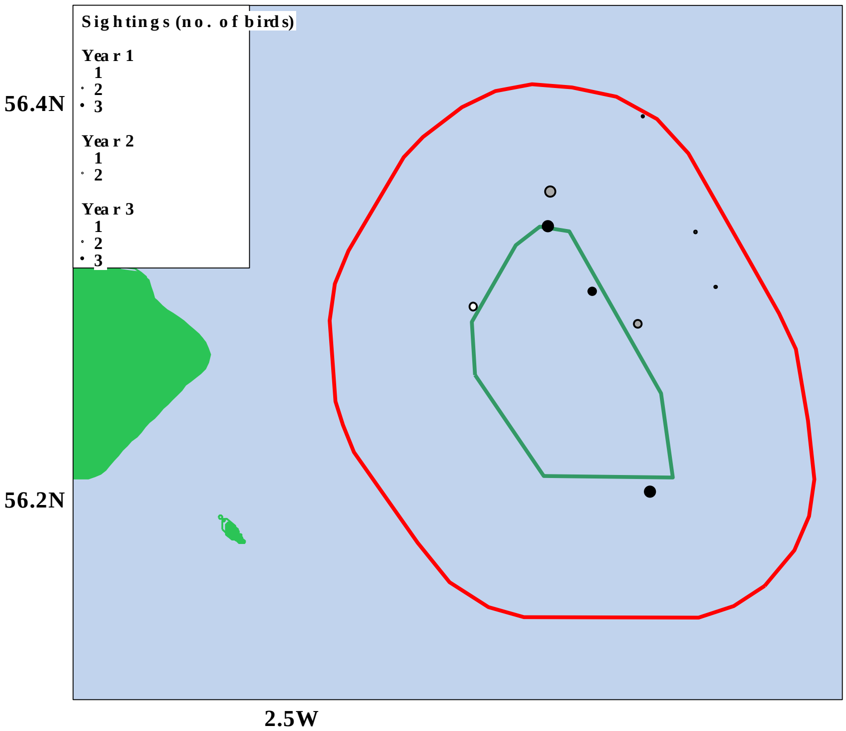


Figure 4.35 Eider sightings in Years 1 to 3

4.4.4.3 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).

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 4.47). 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 4.47 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 (2013) – SPA online species accounts. 2 Calbrade *et al.*, 2010
3 Holt *et al.*, 2011

4.4.4.4 Assessment

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

4.4.5 Common scoter *Melanitta nigra*

4.4.5.1 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).

4.4.5.2 Offshore site and 8 km buffer 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 4.36). Just two common scoter were recorded in Year 2, in the buffer area in September. In Year 3, one common scoter was recorded in the buffer area in October.

A total of eight birds were recorded in flight on baseline surveys, with two birds (25%) flying above 27.5 m in height, at an estimated height of 50 m.

4.4.5.3 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)

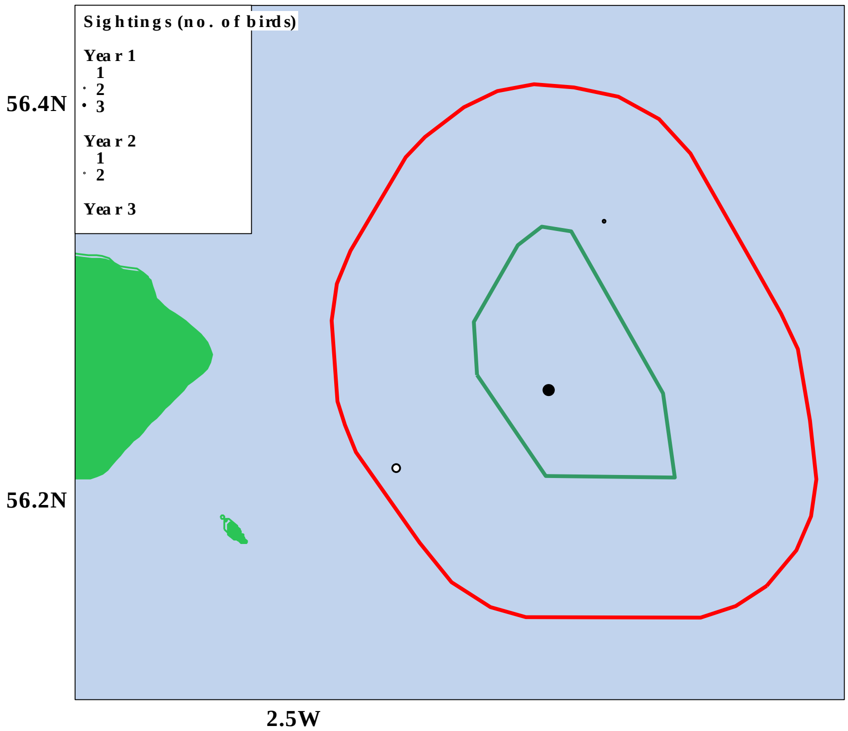


Figure 4.36 Common scoter sightings in Years 1 to 3

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 4.48). These SPAs held 17.3% of the UK non-breeding population, and > 0.3% of the biogeographic population at the time of designation (JNCC, 2013). Recent five-year means are also shown for comparison, where available.

Table 4.48 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 (2013) – SPA online species accounts. 2 Calbrade *et al*., 2010
3 Holt *et al*., 2011

4.4.5.4 Assessment

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

4.4.6 Grey phalarope *Phalaropus fulicarius*

4.4.6.1 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.

4.4.6.2 Offshore site and 8 km buffer area

One grey phalarope was recorded on surveys in Year 1, on the water in the offshore site in November (Table 4.2) (Figure 4.37). In Year 2, two grey phalaropes were recorded on surveys, in the buffer area in October. In Year 3, three grey phalaropes were recorded in the buffer area in November.

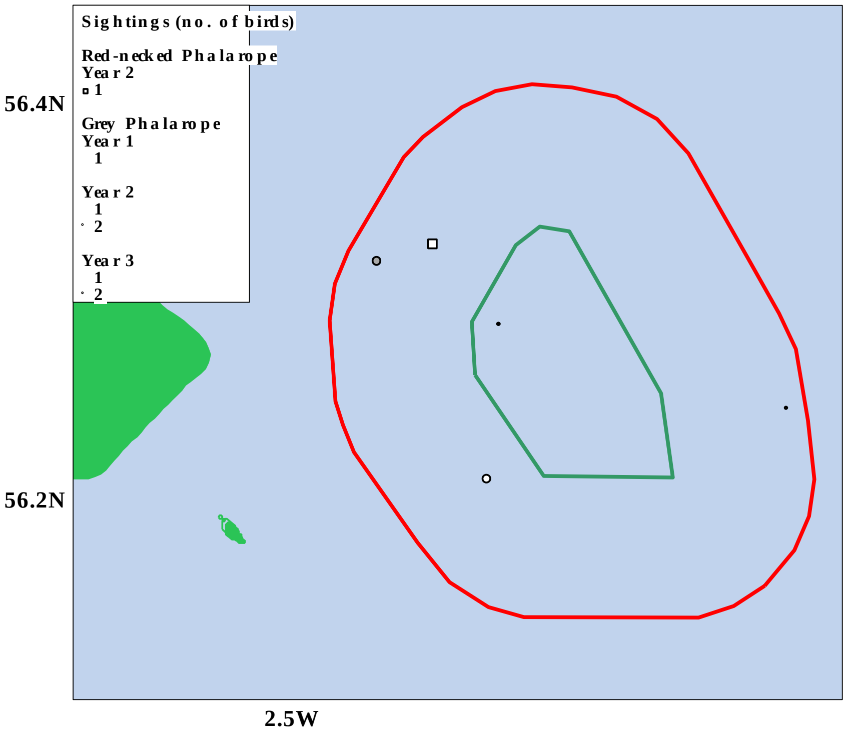


Figure 4.37 Red-necked and grey phalarope sightings in Years 1 to 3

4.4.6.3 Assessment

The results from the baseline surveys, with only one grey phalarope recorded within the offshore site in Year 1, and a further five birds in the buffer area in Years 2 and 3 indicate that grey phalaropes are unlikely to be affected by the Development.

4.4.7 Red-necked Phalarope *Phalaropus lobatus*

4.4.7.1 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).

4.4.7.2 Offshore site and 8 km buffer area

One red-necked phalarope was recorded on surveys in Year 2, in the buffer area in September (Table 4.2) (Figure 4.37). The species was not recorded in Years 1 and 3.

4.4.7.3 Assessment

The results from the baseline surveys, with no birds recorded within the offshore site, and only one red-necked phalarope recorded in the buffer area in Year 2 indicate that red-necked phalarope is unlikely to be affected by the Development.

4.4.8 Pomarine skua *Stercorarius pomarinus*

4.4.8.1 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.

4.4.8.2 Offshore site and 8 km buffer area

In Year 1, six pomarine skuas were recorded during October in the west and south-west of the buffer area (Table 4.2) (Figure 4.38). No pomarine skuas were recorded on surveys in Year 2. In Year 3, three pomarine skuas were recorded in the offshore site, with one in July and two in November. A further 19 were seen in the buffer area in November.

A total of 18 birds were recorded in flight on baseline surveys, with one bird (5.6%) flying above 27.5 m in height, at an estimated 40 m.

4.4.8.3 Assessment

As pomarine skuas were only occasionally recorded in the offshore site on baseline surveys, with three birds on Year 3 surveys, and a further 25 birds seen in the buffer area over the period, it is considered that pomarine skuas are unlikely to be affected by the Development.

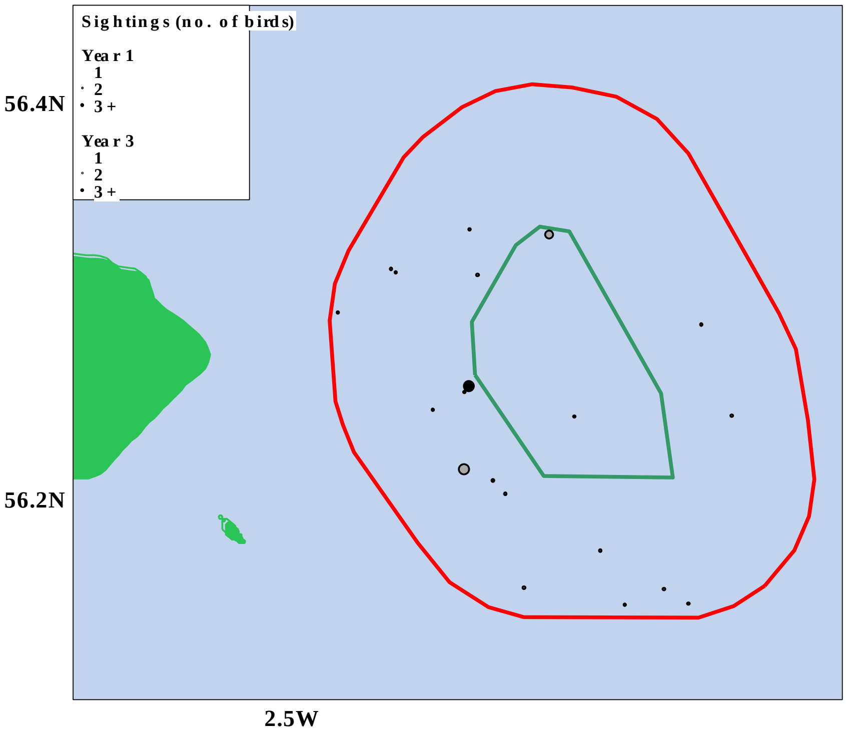


Figure 4.38 Pomarine skua sightings in Years 1 and 3

4.4.9 Arctic skua *Stercorarius parasiticus*

4.4.9.1 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).

4.4.9.2 Offshore site and 8 km buffer area

No Arctic skuas were recorded in the offshore site on baseline surveys. A total of six Arctic skuas were recorded in the buffer area in Year 1, between July and October, with a peak of four in September (Table 4.2) (Figure 4.39). 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. Numbers of Arctic skuas were lower in Year 3, with two birds in the buffer in June and three in November.

A total of 22 Arctic skuas were recorded in flight on baseline surveys, with the majority of birds (95.5%) flying below 27.5 m (Table 4.3). One bird was recorded flying above 27.5 m, i.e. within the rotor-swept zone, at an estimated height of 30 m.

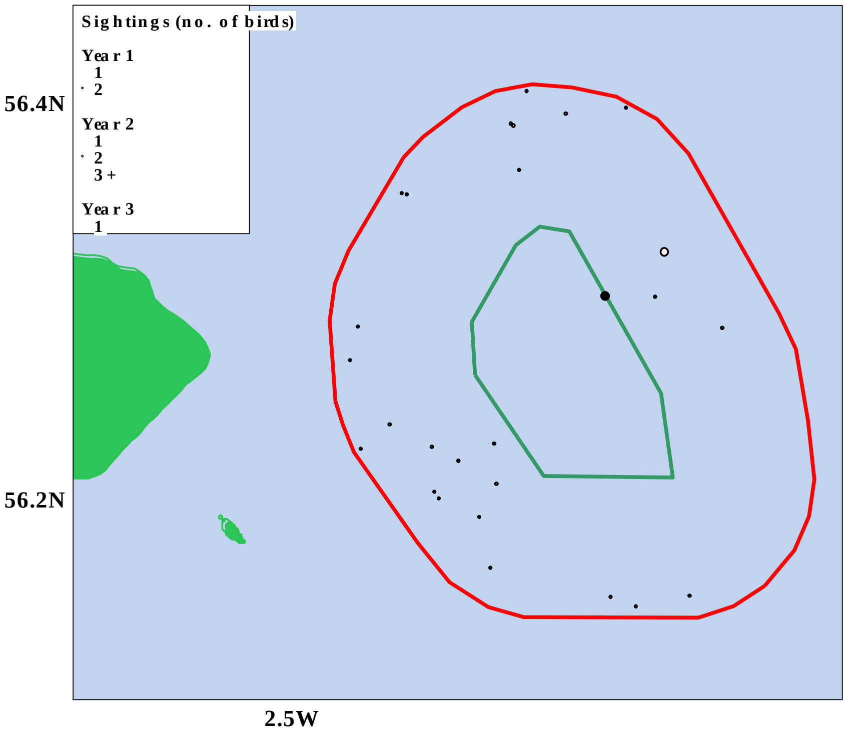


Figure 4.39 Arctic skua sightings in Years 1 to 3

4.4.9.3 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 4.49 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 (2013) – SPA online species accounts. 2 SMP (2013) – 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).

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 4.49). These SPAs held 24.3% of the UK breeding population and 2.7%

of the biogeographic population at the time of designation (JNCC, 2013). 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).

4.4.9.4 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 the Study Area in Years 1 to 3.

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%. Further details are presented in Ornithology Appendix 2: Collision Rate Estimates of Seabirds at Neart na Gaoithe.

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 collisions per annum for a 98.0% avoidance rates. This is approximately 0.2% of the Scottish Arctic skua population.

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 collision mortality of Arctic skuas on the baseline mortality rate is rated as being 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.

4.4.9.5 Cumulative Impact Assessment

There were no significant impacts predicted for the regional population of Arctic skuas in the autumn passage period from the proposed Neart na Gaoithe development. The predicted effects of the development on the regional population of Arctic skuas in the autumn passage period arising from construction, operation and decommissioning are very close to no effect.

Based on these findings, it was considered that it is not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for this population, and therefore no further cumulative impact assessment was undertaken for this species.

4.4.9.6 Mitigation measures

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

4.4.10 Great skua *Stercorarius skua*

4.4.10.1 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).

4.4.10.2 Offshore site and 8 km buffer area

In Year 1, 24 great skuas were recorded on surveys in the offshore site and 8 km buffer 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 4.40). In Year 3, single birds were recorded in the offshore site in November and January, with 18 birds in the buffer area between July and November, peaking in November (10 birds).

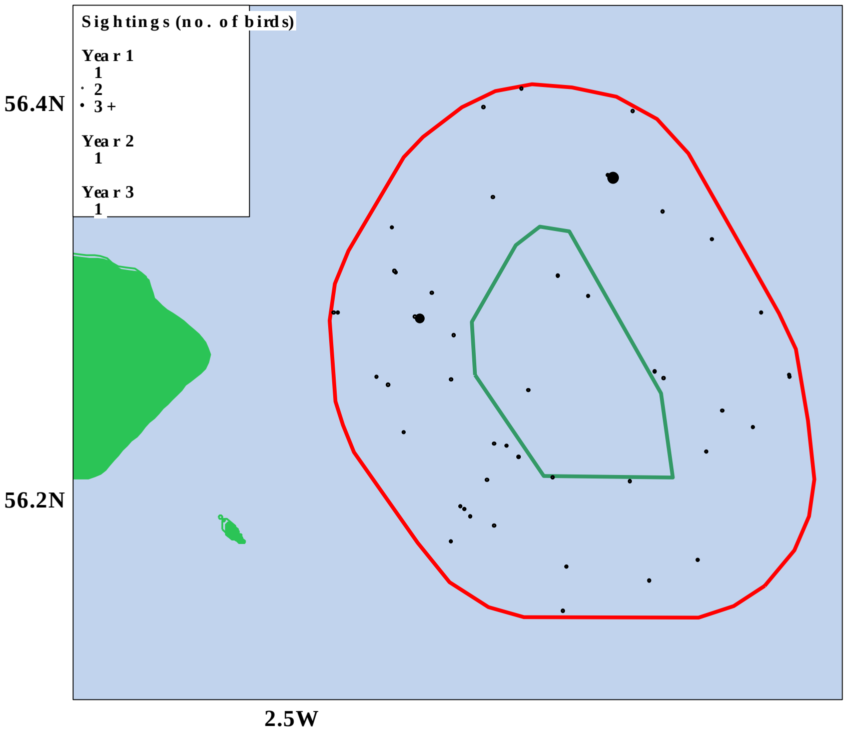


Figure 4.40 Great skua sightings in Years 1 to 3

A total of 49 great skuas were recorded in flight on baseline surveys, with the majority (95.9%) flying below 27.5 m in height (Table 4.3). Two birds were recorded flying above 27.5 m, i.e. within the rotor swept zone, at estimated heights of 30 m and 35 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).

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 4.50). These SPAs held 69.1% of the UK breeding population and 43.4% of the biogeographic population at the time of designation (JNCC, 2013). 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 4.50 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 (2013) – SPA online species accounts. 2 SMP (2013) – 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).

4.4.10.3 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 on the baseline surveys.

Populations

There is no published estimate of the size of the regional autumn passage population of great skua. The 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%. Further details are presented in Ornithology Appendix 2: Collision Rate Estimates of Seabirds at Neart na Gaoithe.

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 collisions per annum for a 98.0% avoidance rate. This is approximately 0.2% of the Scottish great skua population.

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. Firstly, 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. Secondly, 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.

4.4.10.4 Cumulative Impact Assessment

There were no significant impacts predicted for the regional population of great skuas in the autumn passage period from the proposed Neart na Gaoithe development. The predicted effects of the development on the regional population of great skuas in the autumn passage period arising from construction, operation and decommissioning are very close to no effect.

Based on these findings, it was considered that it is not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for this population, and therefore no further cumulative impact assessment was undertaken for this species.

4.4.10.5 Mitigation measures

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

4.4.11 Little gull *Larus minutus*

4.4.11.1 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.

4.4.11.2 Offshore site and 8 km buffer area

Little gulls were mainly recorded on baseline surveys in the offshore site and 8 km buffer area in autumn, with a total of 298 birds in Year 1, 220 birds in Year 2 and 422 birds in Year 3 (raw numbers, all sea states). The majority of birds were recorded in the buffer area (Table 4.2).

During the Year 1 autumn period (August to October), the mean estimated number of little gulls in the offshore site was 109 birds, with a peak of 309 birds in October (Table 4.51). In the same period of Year 2, the mean estimated number of little gulls in the offshore site was 14 birds, with a peak of 41 birds in September. In Year 3, the mean estimated number of little gulls during the autumn period was 152 birds, with a peak of 455 birds in September.

Overall, mean estimated numbers of little gulls in the offshore site were well below the 1% threshold for internationally important numbers of little gulls (1,230 birds) (Holt *et al*., 2011) in all months. In the buffer area, mean estimated numbers exceeded this 1% threshold in September, with a peak of 1,384 birds (three-year mean)(Figure 4.41).

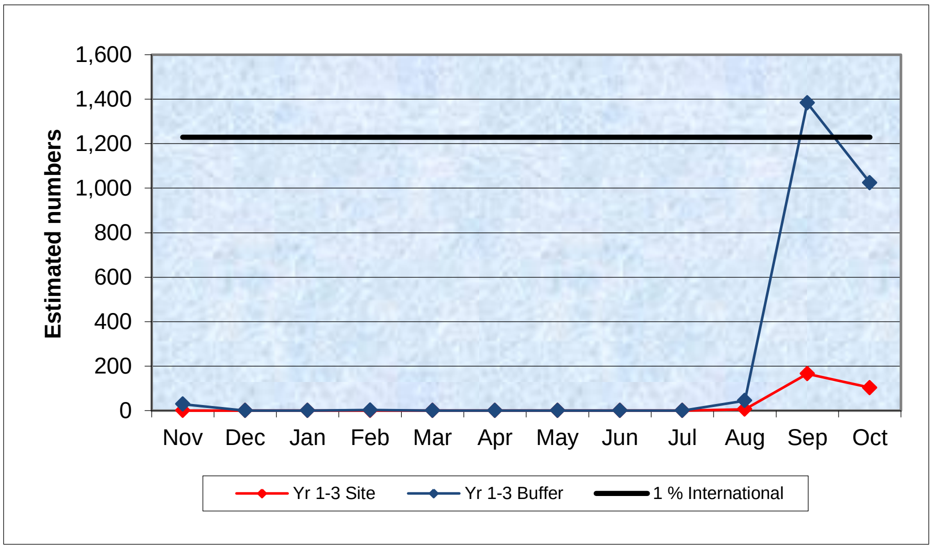


Figure 4.41 Monthly mean estimated numbers of little gulls in the offshore site & buffer area in Years 1 to 3 (Three-year mean)

Table 4.51 Estimated numbers of little gulls in the offshore site (and 1, 2 & 8 km buffer) in Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 8 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	0	0	0	0	0	0	19	57
Yr1 Dec	0	0	0	0	0	0	0	0
Yr1 Jan	0	0	0	0	0	0	0	0
Yr1 Feb	0	0	0	0	0	0	0	0
Yr1 Mar	0	0	0	0	0	0	0	0
Yr1 Apr	0	0	0	0	0	0	0	0
Yr1 May	0	0	0	0	0	0	0	0
Yr1 Jun	0	0	0	0	0	0	0	0
Yr1 Jul	0	0	0	0	0	0	0	0
Yr1 Aug	18	6	58	0	18	18	37	112
Yr1 Sep	0	0	0	0	0	0	0	149
Yr1 Oct	303	157	584	7	309	410	457	1,756
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	0	0	0	0	0	0	0	0
Yr2 Jan	0	0	0	0	0	0	0	0
Yr2 Feb	0	0	0	0	0	0	0	0
Yr2 Mar	0	0	0	0	0	0	0	0
Yr2 Apr	0	0	0	0	0	0	0	0
Yr2 May	0	0	0	0	0	0	0	0
Yr2 Jun	0	0	0	0	0	0	0	0
Yr2 Jul	0	0	0	0	0	0	0	0
Yr2 Aug	0	0	0	0	0	0	0	0
Yr2 Sep	0	0	0	41	41	41	41	657
Yr2 Oct	0	0	0	0	0	0	0	1,352
Yr3 Nov	0	0	0	0	0	0	0	0
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	0	0	0	0	0	0	0	0
Yr3 Feb	0	0	0	0	0	0	0	7
Yr3 Mar	0	0	0	0	0	0	0	0
Yr3 Apr	0	0	0	0	0	0	0	0
Yr3 May	0	0	0	0	0	0	0	0
Yr3 Jun	0	0	0	0	0	0	0	0
Yr3 Jul	0	0	0	0	0	0	0	0
Yr3 Aug	0	0	0	0	0	0	0	40
Yr3 Sep	455	114	1,817	0	455	455	986	3,841
Yr3 Oct	0	0	0	0	0	0	0	275

In the autumn period of Year 1, little gull distribution in the offshore site was patchy, with moderate to high densities in the north. Highest densities of little gulls were recorded in the north of the 8 km buffer area, with few birds recorded to the south of the offshore site (Figure 4.42).

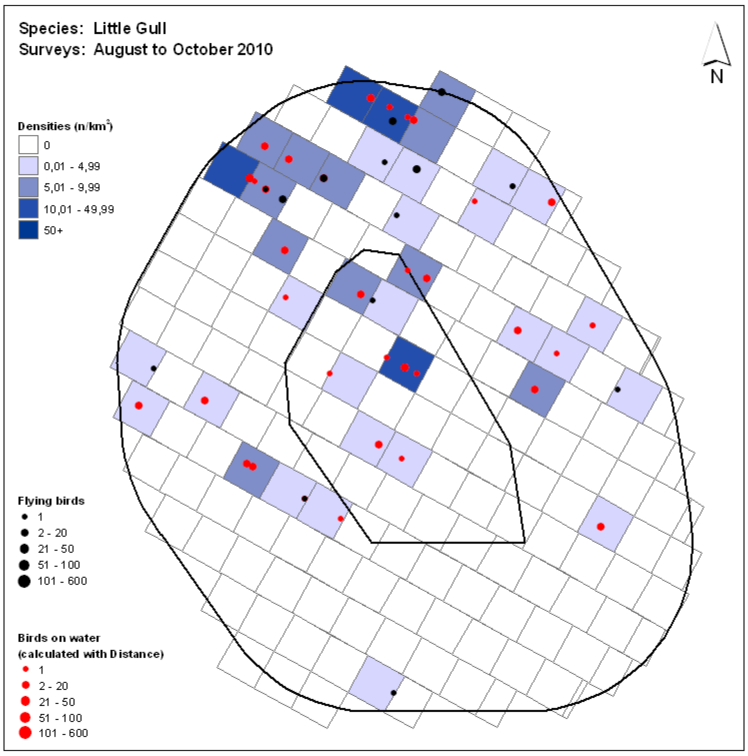


Figure 4.42 Little gull density between August and October, Year 1

A similar distribution pattern was recorded in autumn of Year 2, with fewer little gulls in the offshore site between August and October than in Year 1. Highest densities of little gulls were recorded in the north-west corner of the buffer area (Figure 4.43).

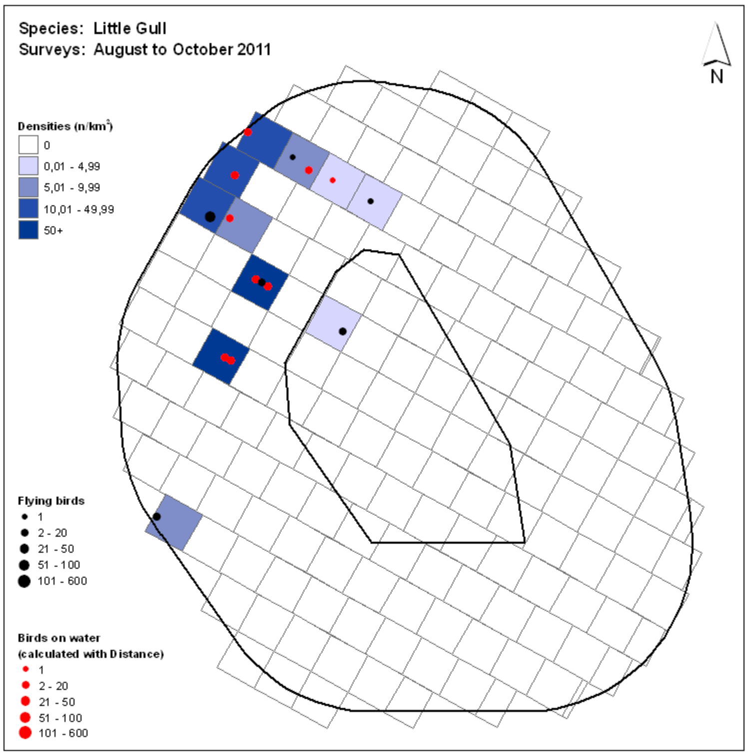


Figure 4.43 Little gull density between August and October, Year 2

A similar distribution pattern was again recorded in autumn of Year 3, with few little gulls in the offshore site (Figure 4.44). Again, highest densities of little gulls were recorded in the north of the buffer area at this time.

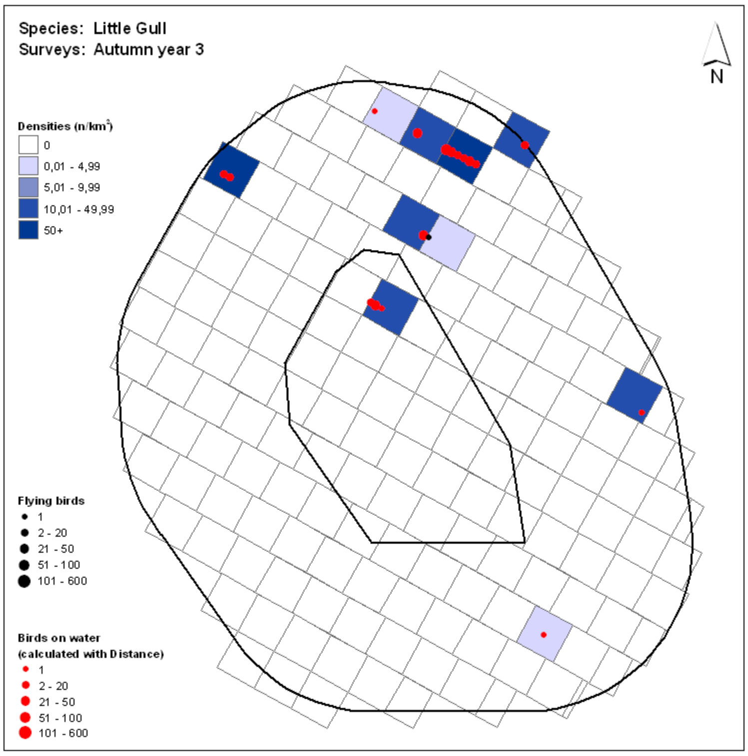


Figure 4.44 Little gull density between August and October, Year 3

A total of 327 little gulls were recorded in flight, with 74.9% of all birds flying below 27.5 m in height (Table 4.3). A total of 82 birds (25.1%) were recorded flying above 27.5 m, i.e. within the rotor swept zone, at an estimated height of 30 m.

Three types of foraging behaviour were recorded for 140 little gulls in Years 1 to 3 (Table 4.52). Actively searching was the most frequently recorded foraging behaviour (38.6%).

Table 4.52 Little gull foraging behaviour in the offshore site and 8 km buffer area in Years 1 to 3

Behaviour	Number of birds
Actively searching	54
Dipping	45
Surface pecking	41
Total	**140**

4.4.11.3 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).

4.4.11.4 Assessment

Definition of seasons

Little gulls were only recorded in the offshore site between August and October. Therefore, this was the period considered for assessment. This time of the year corresponds to the autumn passage. 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).

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 peak estimates for the offshore site and 8 km buffer area of 1,756 birds in October of Year 1, 1,352 birds in October of Year 2 and 3,841 birds in September of Year 3. For the purposes of assessment the regional autumn passage population is assumed to be 4,500 birds. This is based on the peak estimated number recorded on baseline surveys for the offshore site and 8 km buffer area (3,841 birds in September 2012) plus the maximum numbers recorded in the first year of baseline surveys in the Inch Cape survey area (approximately 370 birds, (ICOL, 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

The nature conservation importance of little gulls using the site is rated as high during the autumn passage period, because the species is listed on Annex 1 of the Birds Directive.

Offshore wind farms studies

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.

Construction Phase

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 population of little gulls in the autumn passage period is ***not significant*** under the EIA Regulations.

Operational phase

Displacement

In order to assess the potential impacts from displacement, the approach presented in the interim guidance note on displacement issued by the JNCC and NE has been followed (JNCC & NE 2012).

Peak estimated numbers of little gulls in the offshore site in the autumn passage period (August to October) for Years 1 to 3 were averaged to get the three-year mean peak for the period. Where peak numbers occurred in different months within the same season across different years, the peak month was used. This was repeated for a 1 km and 2 km buffer around the offshore site (Table 4.53).

Table 4.53 Seasonal three-year mean peak estimated numbers of little gulls in the offshore site (plus 1 & 2 km buffer)

Year	Offshore site	Offshore site + 1 km	Offshore site + 2 km
	Autumn passage	Autumn passage	Autumn passage
Year 1	309	410	457
Year 2	41	41	41
Year 3	455	455	986
3-year mean peak	268	302	495

Guidance recommends presenting a range of potential displacement and mortality rates and wherever possible selecting a suitable impact based on empirical evidence. Where, there is little evidence to support the assessment a precautionary approach should be taken.

Likely impacts of displacement

Based on evidence from post-construction studies at the Arklow Bank Offshore Wind Farm (Barton *et al*., 2009, Barton *et al*., 2010), indicating that little gulls are unlikely to be displaced by the presence of operating turbines, a precautionary level of 10% displacement was used for this assessment.

Assuming 10% of all little gulls were displaced from the offshore site during the autumn passage period, this would affect an estimated 27 birds, (Table 4.54) increasing to 50 birds for the offshore site plus 2 km buffer (Table 4.56). This is approximately 1.1% of the estimated regional population in the autumn passage period (4,500 birds). As little gulls are on migration at this time, they are moving through the area and it was considered that any displaced birds would be able to find other suitable foraging areas, and that no little gulls would die as a result of being displaced from the offshore site and 2 km buffer area in the autumn passage period.

Table 4.54 Estimated number of little gulls predicted to be at risk of mortality following displacement from offshore site in the autumn passage period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	1	3	5	8	11	13	16	19	21	24	27
	20%	1	3	5	11	16	21	27	32	38	43	48	54
	30%	2	4	8	16	24	32	40	48	56	64	72	80
	40%	2	5	11	21	32	43	54	64	75	86	96	107
	50%	3	7	13	27	40	54	67	80	94	107	121	134
	60%	3	8	16	32	48	64	80	96	113	129	145	161
	70%	4	9	19	38	56	75	94	113	131	150	169	188
	80%	4	11	21	43	64	86	107	129	150	172	193	214
	90%	5	12	24	48	72	96	121	145	169	193	217	241
	100%	5	13	27	54	80	107	134	161	188	214	241	268
Three-year mean peak of 268 little gulls in the offshore site in the autumn passage period (Aug to Oct)													
Estimated regional population in the autumn period = 4,500 birds													

Table 4.55 Estimated number of little gulls predicted to be at risk of mortality following displacement from offshore site plus 1 km buffer in the autumn passage period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	2	3	6	9	12	15	18	21	24	27	30
	20%	1	3	6	12	18	24	30	36	42	48	54	60
	30%	2	5	9	18	27	36	45	54	63	72	82	91
	40%	2	6	12	24	36	48	60	72	85	97	109	121
	50%	3	8	15	30	45	60	76	91	106	121	136	151
	60%	4	9	18	36	54	72	91	109	127	145	163	181
	70%	4	11	21	42	63	85	106	127	148	169	190	211
	80%	5	12	24	48	72	97	121	145	169	193	217	242
	90%	5	14	27	54	82	109	136	163	190	217	245	272
	100%	6	15	30	60	91	121	151	181	211	242	272	302
Three-year mean peak of 302 little gulls in the offshore site in the autumn passage period (Aug to Oct)													
Estimated regional population in the autumn period = 4,500 birds													

Table 4.56 Estimated number of little gulls predicted to be at risk of mortality following displacement from offshore site plus 2 km buffer in the autumn passage period

		Mortality (%)										
	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
Displacement level (%)	10%	1	2	5	10	15	20	25	30	35	40	45	50
	20%	2	5	10	20	30	40	50	59	69	79	89	99
	30%	3	7	15	30	45	59	74	89	104	119	134	149
	40%	4	10	20	40	59	79	99	119	139	158	178	198
	50%	5	12	25	50	74	99	124	149	173	198	223	248
	60%	6	15	30	59	89	119	149	178	208	238	267	297
	70%	7	17	35	69	104	139	173	208	243	277	312	347
	80%	8	20	40	79	119	158	198	238	277	317	356	396
	90%	9	22	45	89	134	178	223	267	312	356	401	446
	100%	10	25	50	99	149	198	248	297	347	396	446	495
Three-year mean peak of 495 little gulls in the offshore site in the autumn passage period (Aug to Oct)												
Estimated regional population in the autumn period = 4,500 birds												

Based on the distribution and densities of little gulls recorded in the offshore site from baseline studies, and evidence from other wind farm studies indicating that little gulls are not susceptible to displacement, it is likely that should displacement occur, it will not result in any discernible effects on the regional population of little gulls in the autumn passage period.

This impact is categorised as being of low magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the effects of displacement on the regional little gull population in the autumn passage 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.

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

CRM estimated the number of potential little gull collisions per season for four wind turbine designs scenarios (Table 3.6). The minimum height for the turbine blades above the sea surface for all four options is 27.5 m at mean sea level (MSL).

There is no specific SNH guidance on avoidance rates for seabirds, and therefore their default value of 98.0% is presented in this assessment.

Highest estimated collisions in the breeding period were for Option 1 (90 x 5 MW turbines). Using a 98% avoidance rate, the number of predicted little gull collisions in the autumn passage period was 14 birds (Table 4.57). This equates to approximately 0.3% of the estimated regional population (4,500 birds) in the autumn period.

Lowest estimated collisions in the breeding period were for Options 3 and 4 (73 x 6.15 MW turbines). Using a 98% avoidance rate, the number of predicted little gull collisions in the autumn passage period was 11 birds (Table 4.57). This equates to approximately 0.2% of the estimated regional population (4,500 birds) in the autumn period. Due to the low numbers of little gulls recorded in the offshore site in other months, the annual predicted collision levels were the same as the autumn levels.

Table 4.57 Number of estimated little gull collisions using avoidance rate of 98% for four wind farm options

	Option 1 (90x5MW)	Option 2 (75x6MW)	Option 3 (73x6.15MW)	Option 4 (73x6.15MW)
Collisions in Autumn period (August to October), all ages	14	12	11	11
Total collisions per year, all ages	14	12	11	11

It is concluded that for the most adverse design (Option 1: 90 x 5 MW turbines), collision mortality for little gull is an effect of negligible magnitude, that is temporally long-term and reversible. Similarly for the least adverse designs (Options 3 and 4: 73 x 6.15MW turbines), collision mortality for little gull is an effect of negligible magnitude, that is temporally long-term and reversible.

It is concluded that for the wind farm designs examined here, the effects of collision mortality on little gulls from the regional population in the autumn passage period are ***not significant*** under the Electricity Regulations.

Decommissioning Phase

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.

Summary of combined effects

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 as having high NCI.

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 ***no significance*** under the EIA regulations (Table 4.58).

Table 4.58 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	Low	Long term	Moderate	***Not significant***
Vessel disturbance	Negligible	Long term	Low	***Not significant***
Collision mortality for 98.0% A R	Negligible	Long term	Moderate	***Not significant***
All effects combined (collision 98.0% A R)	Negligible	Long term	Moderate	***Not significant***

4.4.11.5 Cumulative Impact Assessment

There were no significant impacts predicted for the regional population of little gulls in the autumn passage period from the proposed Neart na Gaoithe development. The predicted effects of the development on the regional population of little gulls in the autumn passage period arising from construction, operation and decommissioning are very close to no effect.

Based on these findings, it was considered that it is not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for this population, and therefore no further cumulative impact assessment was undertaken for this species.

4.4.11.6 Mitigation measures

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

4.4.12 Sabine's gull *Larus sabini*

4.4.12.1 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).

4.4.12.2 Offshore site and 8 km buffer 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 4.45). 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. There were no sightings of Sabine's gull in the study area in Year 3.

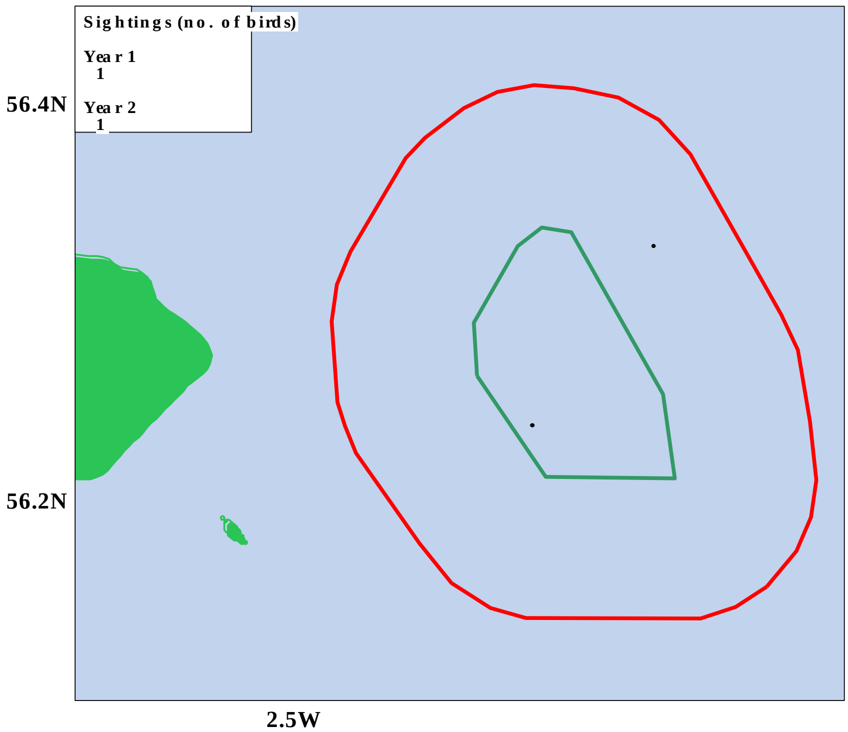


Figure 4.45 Sabine's gull sightings in Years 1 and 2

4.4.12.3 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.

4.4.13 Black-headed gull *Larus ridibundus*

4.4.13.1 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).

4.4.13.2 Offshore site and 8 km buffer area

No black-headed gulls were recorded in the offshore site on baseline surveys. A total of 27 black-headed gulls were recorded in the buffer area in Year 1, with a peak count of 25 in November (Table 4.2). Fewer black-headed gulls were seen in Year 2, with 11 birds recorded between March and October, and a peak of 6 birds in August. In Year 3, just one black-headed gull was recorded, in the buffer area in November. The majority of birds were in the west of the study area, over shallower water (Figure 4.46).

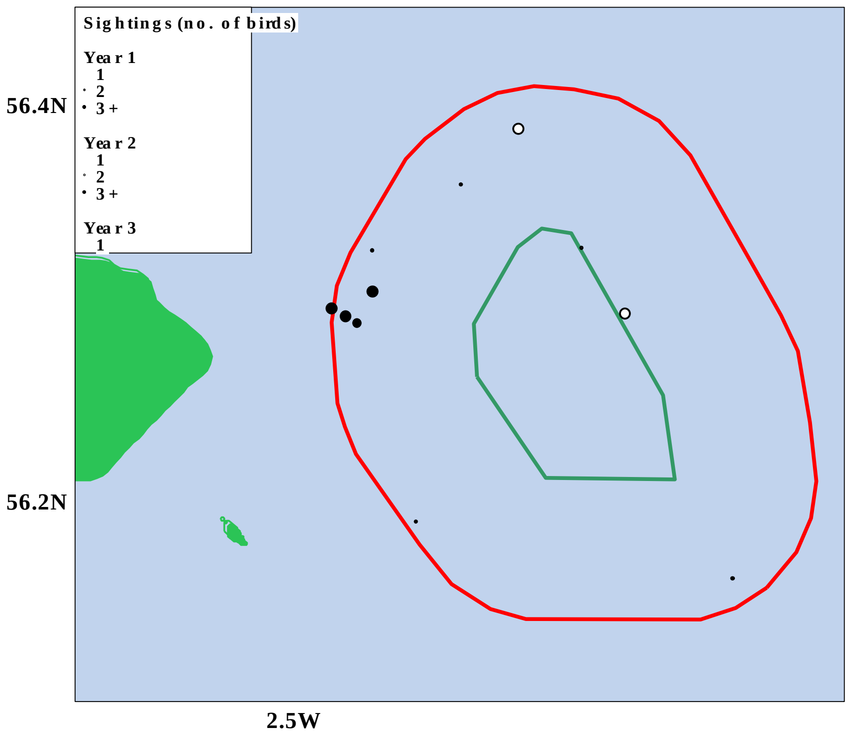


Figure 4.46 Black-headed gull sightings in Years 1 to3

A total of 38 black-headed gulls were recorded in flight, with 29 birds (76.3%) flying below 27.5 m and nine birds (23.7%) flying above 27.5 m, i.e. within the rotor swept zone, at an estimated height of 30 m. (Table 4.3).

10.3.1.1 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).

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, 2013). Since designation, the population at Coquet Island has increased to an estimated 3,720 pairs in 2012 (SMP, 2013). The distance between the offshore site and this SPA is greater than the maximum known foraging range of this species (25.5 km) (Thaxter *et al*., 2012).

4.4.13.3 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 should displacement occur, the effect 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 (40 km) (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 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.

4.4.13.4 Cumulative Impact Assessment

There were no significant impacts predicted for the regional population of black-headed gulls in the breeding or non-breeding periods from the proposed Neart na Gaoithe development. The predicted effects of the development on the regional population of black-headed gulls in the breeding or non-breeding periods arising from construction, operation and decommissioning are very close to no effect.

Based on these findings, it was considered that it is not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for this population, and therefore no further cumulative impact assessment was undertaken for this species.

4.4.13.5 Mitigation measures

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

4.4.14 Common gull *Larus canus*

4.4.14.1 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).

4.4.14.2 Offshore site and 8 km buffer area

In Year 1, 78 common gulls were recorded on surveys in the offshore site and 8 km buffer area, with a peak of 28 birds in October (Table 4.2). Fewer common gulls were seen in Year 2, with 52 birds recorded, with a peak of 15 birds in December. In Year 3, a total of 22 common gulls were recorded in the study area, with a peak of 9 birds in November (raw numbers, all sea states). In all three years, numbers recorded in the offshore site on baseline surveys were very low, with six birds seen in Year 1, 12 in Year 2 and two birds in Year 3.

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 very low in the offshore site and the buffer area in Years 1 to 3, with a three-year mean peak of 0.09 birds/km in the offshore site in December (Figure 4.47).

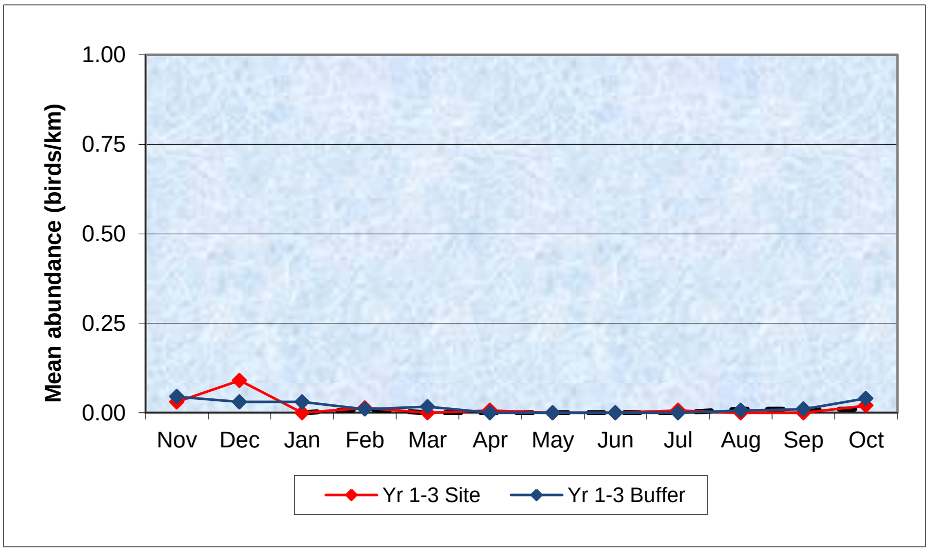


Figure 4.47 Comparison of common gull monthly mean abundance in the offshore site and buffer area in Years 1 to 3 (Three-year mean)

Common gulls were scattered sporadically in the western half of the offshore site and 8 km buffer area at low abundances in Year 1 (Figure 4.48). Few birds were recorded in the offshore site in Year 1.

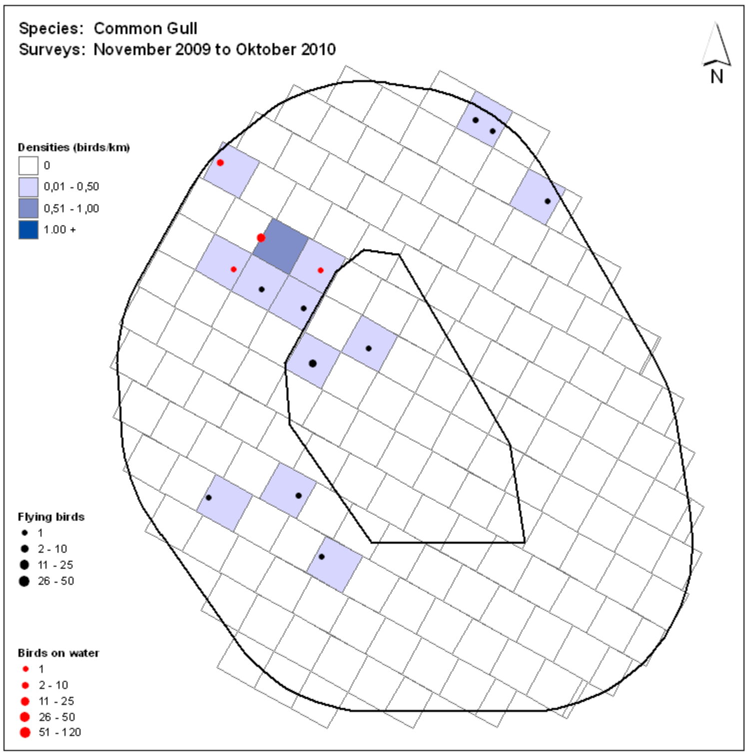


Figure 4.48 Common gull abundance all months combined, Year 1

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 4.49). As in Year 1, few common gulls were recorded in the offshore site over the period.

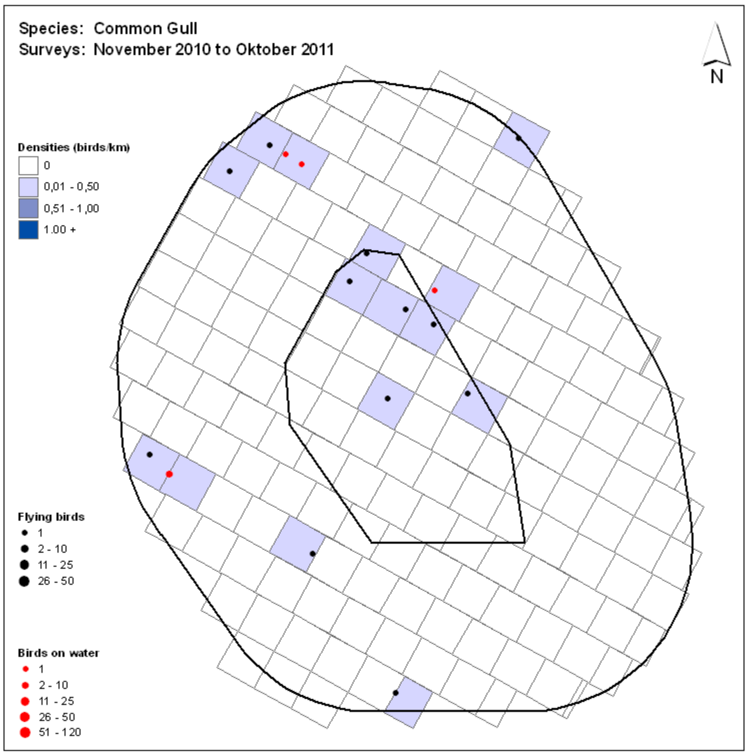


Figure 4.49 Common gull abundance all months combined, Year 2

Common gull numbers in Year 3 were lower than in the two previous years, with no birds recorded in the offshore site in Year 3 (Figure 4.50).

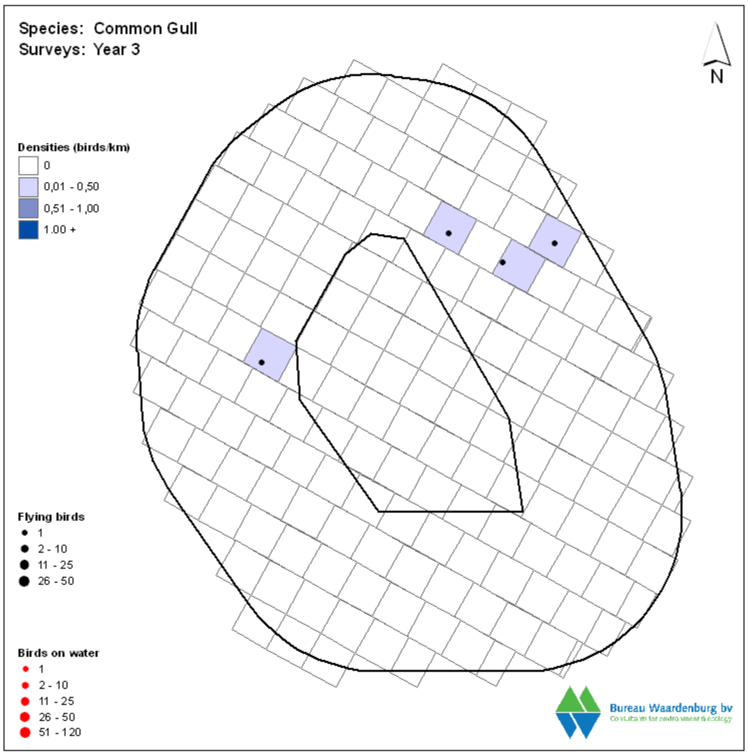


Figure 4.50 Common gull abundance all months combined, Year 3

A total of 123 common gulls were recorded in flight on baseline surveys, with 78.0% of birds flying below 27.5 m, and 27 birds (22.0%) flying above 27.5 m, i.e. within the rotor swept zone, at estimated heights of between 30 and 50 m. (Table 4.3).

4.4.14.3 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).

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). No recent count is available for this SPA, although numbers at the colony have declined (SMP 2013). 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).

4.4.14.4 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. Very 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 common gulls are predominantly found in coastal waters or inland (Forrester *et al*., 2007). Based on this, and the very 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 (123 birds). Although 22.0% of all flying birds were recorded flying above 27.5 m (27 birds), 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 collision mortality on common gulls is rated as being of negligible 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.

4.4.14.5 Cumulative Impact Assessment

There were no significant impacts predicted for the regional population of common gulls in the breeding or non-breeding periods from the proposed Neart na Gaoithe development. The predicted effects of the development on the regional population of common gulls in the breeding or non-breeding periods arising from construction, operation and decommissioning are very close to no effect.

Based on these findings, it was considered that it is not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for this population, and therefore no further cumulative impact assessment was undertaken for this species.

4.4.14.6 Mitigation measures

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

4.4.15 Lesser black-backed gull *Larus fuscus*

4.4.15.1 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).

4.4.15.2 Offshore site and 8 km buffer area

In Year 1, 66 lesser black-backed gulls were recorded on surveys in the offshore site and 8 km buffer area, with 10 birds recorded in the offshore site (raw numbers, all sea states) (Table 4.2) (Table 4.59). Numbers on surveys in Year 2 were higher, with 195 birds recorded, although only 11 birds were recorded in the offshore site over the year. In Year 3, a total of 171 lesser black-backed gulls were recorded in the study area, with 37 birds recorded in the offshore site. In all three years, the majority of sightings were between April and September.

Table 4.59 Raw numbers of lesser black-backed gulls recorded in the offshore site between April and September, Years 1 to 3

Year	April	May	June	July	Aug	Sep	Total
Year 1	1	1	4	2	1	0	9
Year 2	0	0	2	8	1	0	11
Year 3	0	20	3	8	6	0	37
Mean	1	7	3	6	3	0	19

Due to the low sample size of lesser black-backed 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 lesser black-backed gull abundance was low in the offshore site and the buffer area on baseline surveys, with a three-year mean peak of 0.14 birds/km in the offshore site in May, and a three-year mean peak of 0.15 birds/km in the buffer area in September (Figure 4.51).

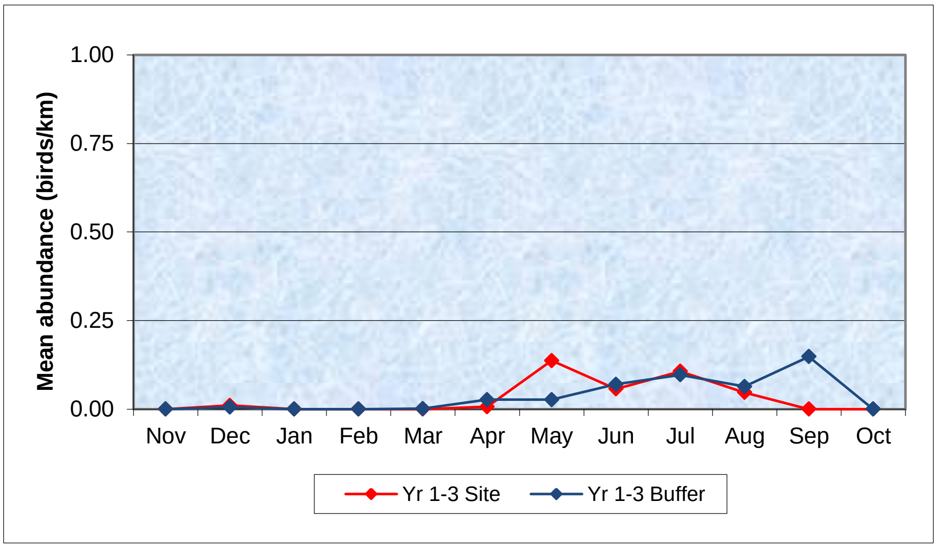


Figure 4.51 Comparison of lesser black-backed gull monthly mean abundance in the offshore site & buffer area in Years 1 to 3 (Three-year mean)

Lesser black-backed gulls were mainly scattered sporadically throughout the southern half of the offshore site and 8 km buffer area at low abundances between April and August of Year 1 (Figure 4.52). Few birds were recorded in the offshore site over the period.

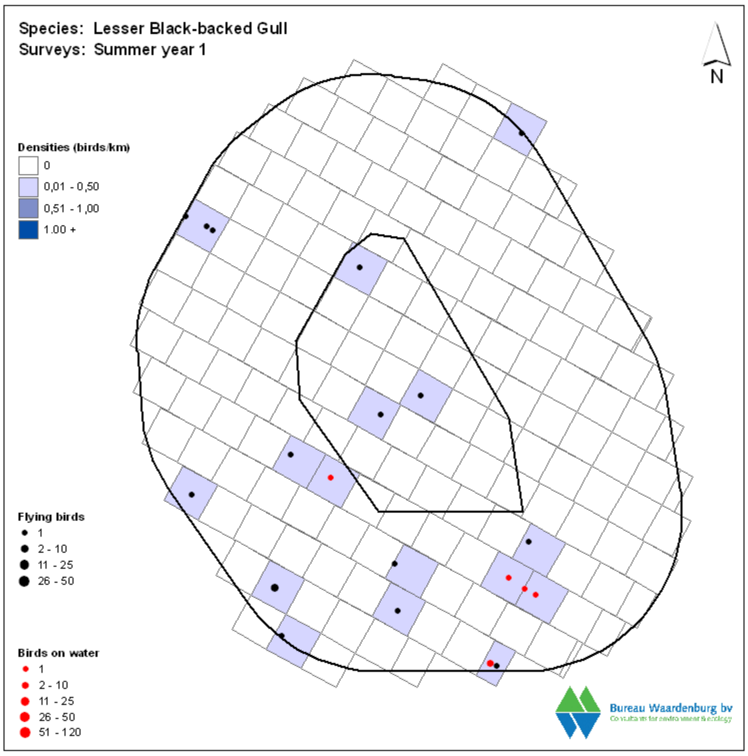


Figure 4.52 Lesser black-backed gull abundance between April and August, Year 1

A similar distribution pattern was recorded in Year 2 between April and August, with highest abundance of lesser black-backed gulls recorded in the south-west of the buffer area (Figure 4.53). Fewer lesser black-backed gulls were recorded in the offshore site over the period than in Year 1.

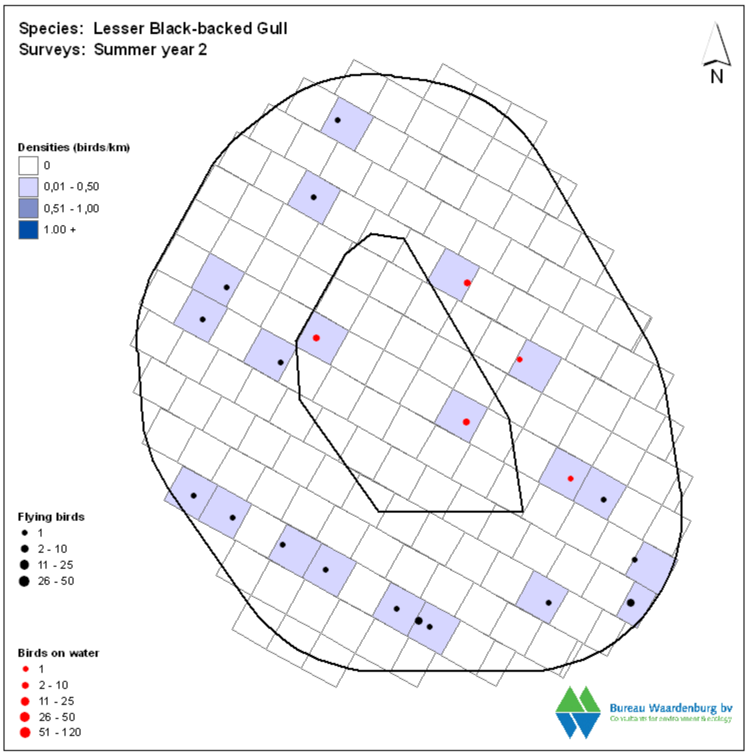


Figure 4.53 Lesser black-backed gull abundance between April and August, Year 2

Lesser black-backed gulls were more abundant between April and August of Year 3 compared to previous years, although numbers recorded in the offshore site were still low, (Figure 4.54). Most birds were recorded in the south of the buffer area at this time.

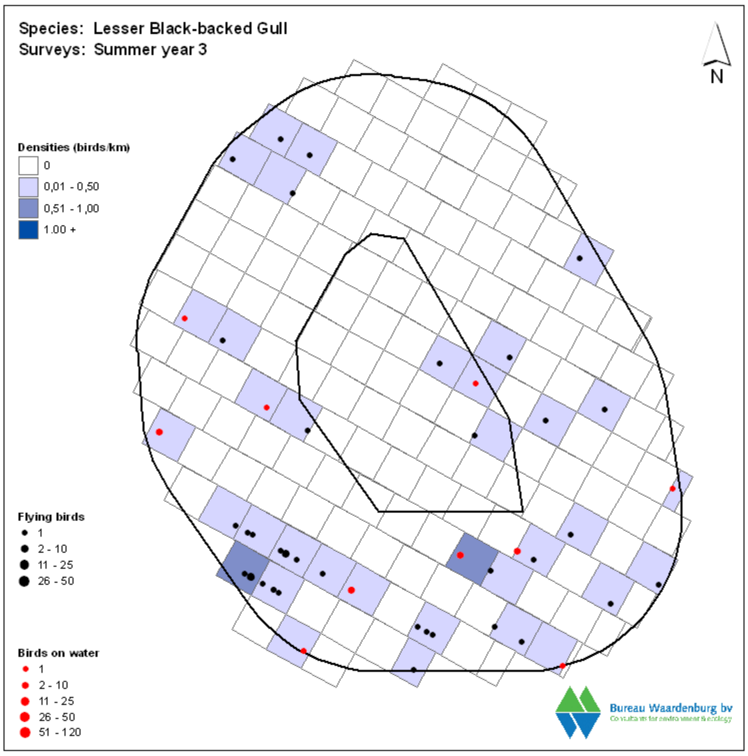


Figure 4.54 Lesser black-backed gull abundance between April and September, Year 3

A total of 358 lesser black-backed gulls were recorded in flight on baseline surveys, with 90.8% of birds flying below 27.5 m i.e. below the turbine swept zone (Table 4.3). A total of 33 birds (9.2%) were recorded flying above 27.5 m, i.e. within the rotor swept zone, at estimated heights of 30 to 50 m.

4.4.15.3 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).

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, 2013). Since designation, the breeding population at this SPA has decreased slightly from 2,920 pairs to approximately 2,854 pairs during Seabird 2000 (SMP, 2013). More recent data for some colonies in the SPA e.g. Craigleith was not available. The distance between the offshore site and the Forth Islands SPA is within the mean maximum foraging range of lesser black-backed gull (141 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 4.55.

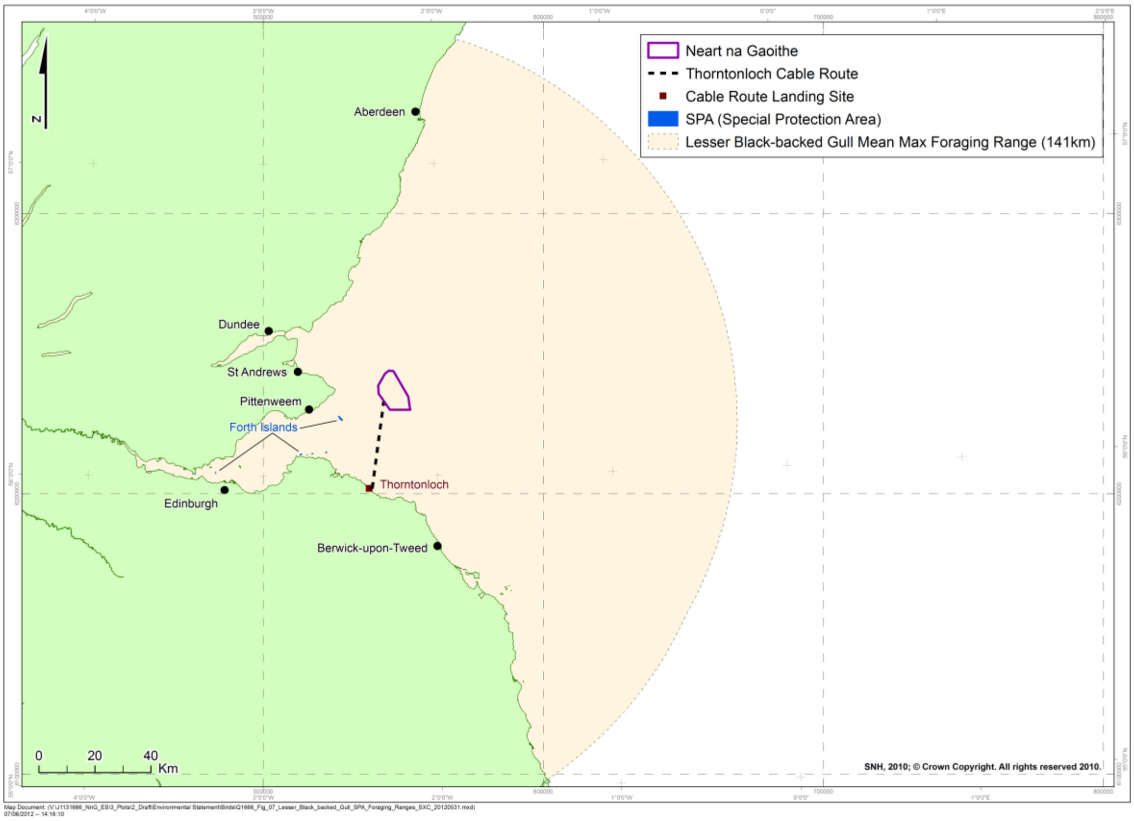


Figure 4.55 Lesser black-backed gull mean maximum foraging range from the Firth of Forth breeding SPA in relation to the Development

4.4.15.4 Assessment

Definition of seasons

Lesser black-backed gulls are predominantly a summer visitor and were only recorded in the offshore site during the breeding season (April to August), although numbers were low. This assessment therefore only considers the breeding period (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.

Populations

Recent counts on the Isle of May indicate the breeding population of lesser black-backed gulls has increased since Seabird 2000. However, as recent data was not available for other breeding colonies in the region, counts from Seabird 2000 have been used in this assessment. On this basis, the regional breeding population between Peterhead and Blyth was estimated as 8,752 breeding pairs (Mitchell *et al.,* 2004).

The regional breeding SPA population within mean maximum foraging range (141 km) (Thaxter *et al.,* 2012) is estimated to be 2,854 pairs, based on Seabird 2000 counts from Mitchell *et al.,* (2004) and SMP (2013).

The regional population size in the non-breeding period was estimated at approximately 500 birds, based on the "Western North Sea" region in Stone *et al.* (1995).

Nature conservation importance

The nature conservation importance of lesser black-backed gulls using the offshore site was 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 were well below 1% of this SPA population. The mean number present in the offshore site in the breeding season is also well below 1% of the regional population.

Offshore wind farm studies

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).

Analysis of changes in numbers of large gulls (including lesser black-backed gull) at Robin Rigg Offshore Wind Farm suggests a possible decline in numbers within the wind farm during the first year of operation, however this pattern was not clear, and further monitoring is required (Walls, *et al.,* 2013).

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).

Construction Phase

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.

Operational Phase

Displacement

It is not known whether lesser black-backed gulls will be displaced by the offshore site. However, evidence from other wind farms (e.g. Leopold *et al,* 2012) indicates that displacement levels are likely to be low. The review undertaken by Langston (2010) also supports this. Very low numbers of lesser black-backed gulls were recorded in the offshore site on baseline surveys between April and August, with no birds recorded outside these months. Overall, it is concluded that any effects of displacement on lesser black-backed gulls is ***not significant*** under the terms of 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 this assessment, it was 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. Therefore, the only birds considered in this 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 north east. 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 3.7). 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 3.8).

For birds breeding on Craigleith, the wind farm acting as a barrier would potentially block approximately 28% of the possible flight directions (Table 3.7). 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 3.8).

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, 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, 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

CRM estimated the number of potential lesser black-backed gull collisions per season for four wind turbine designs scenarios (Table 3.6). The minimum height for the turbine blades above the sea surface for all four options is 27.5 m at mean sea level (MSL).

There is no specific SNH guidance on avoidance rates for seabirds, and therefore their default value of 98.0% is presented in this assessment.

Highest estimated collisions in the breeding period were for Option 1 (90 x 5 MW turbines). Using a 98% avoidance rate, the number of predicted lesser black-backed gull collisions in the breeding period was 3 birds (Table 4.60). This equates to approximately 0.05% of the regional SPA population within mean maximum foraging range (2,854 pairs) (SMP 2013) in the breeding period.

Lowest estimated collisions in the breeding period were for Options 3 and 4 (73 x 6.15 MW turbines). Using a 98% avoidance rate, the number of predicted lesser black-backed gull collisions in the breeding period was 2 birds (Table 4.60). This equates to approximately 0.04% of the regional SPA population (2,854 pairs) (SMP 2013) in the breeding period.

Zero collisions were estimated for all four turbine options for the non-breeding period (September to March) (Table 4.60). Due to the low numbers of lesser black-backed gulls recorded in the offshore site in other months, the annual predicted collision levels were the same as for the breeding period.

Table 4.60 Number of estimated lesser black-backed gull collisions using avoidance rate of 98% for four wind farm options

	Option 1 (90x5MW)	Option 2 (75x6MW)	Option 3 (73x6.15MW)	Option 4 (73x6.15MW)
Collisions in breeding season (April to August), all ages	3	2	2	2
Collisions in non-breeding period (September to March), all ages	0	0	0	0
Total collisions per year, all ages	3	2	2	2

It is concluded that for the most adverse design (Option 1: 90 x 5 MW turbines), collision mortality for lesser black-backed gull is an effect of negligible magnitude, that is temporally long-term and reversible.

It is further concluded that for the least adverse designs (Options 3 and 4: 73 x 6.15MW turbines), collision mortality for lesser black-backed gull is an effect of negligible magnitude, that is temporally long-term and reversible.

It is concluded that for the wind farm designs examined here, the effects of collision mortality on lesser black-backed gulls from the regional population in the breeding and non-breeding periods are ***not significant*** under the Electricity Regulations.

Decommissioning Phase

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.

Summary of combined effects

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 4.61).

Table 4.61 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***

4.4.15.5 Cumulative Impact Assessment

Displacement

No significant impacts were predicted to arise from displacement caused by Neart na Gaoithe for lesser black-backed gulls in the breeding or non-breeding periods. This was based on the numbers of birds recorded within the offshore site, and evidence from other wind farms, indicating likely low levels of displacement for this species. It is therefore not plausible that Neart na Gaoithe could contribute to a significant cumulative displacement impact for the regional population of lesser black-backed gulls in the breeding or non-breeding period. As a result, no further cumulative impact assessment for displacement was undertaken for this species.

Collision mortality

The potential cumulative collision risk to lesser black-backed gulls from Neart na Gaoithe and other plans or projects is higher than for Neart na Gaoithe alone.

Projects identified during consultation and the undertaking of the EIA for which there is a potential for a cumulative collision risk impact are:

- Inch Cape offshore wind farm;
- Seagreen Project Alpha offshore wind farm; and
- Seagreen Project Bravo offshore wind farm.

Data on the number of predicted collisions arising from the Seagreen Projects Alpha and Bravo offshore wind farms are presented within the applicant's Environmental Statement

(Seagreen 2012) and HRA (Seagreen 2013). The proposed Seagreen Round 3 Zone development consists of a three phase programme of six separate developments. Projects Alpha and Bravo are the first developments and the only Seagreen projects for which applications have been made.

No application has been made for proposed Inch Cape offshore wind farm. Predicted number of collision arising from the Inch Cape offshore wind farm have been obtained from an unpublished annual ornithological report based on one years data (ICOL, 2012).

The predicted cumulative impacts on lesser black-backed gulls from Neart na Gaoithe and Seagreen projects Alpha and Bravo in the breeding and non-breeding periods are presented in Table 4.62.

Table 4.62 Predicted number of cumulative lesser black-backed gulls (adult & immature) mortality with Seagreen Projects Alpha and Bravo in the breeding and non-breeding periods

Wind Farm	Breeding	Non-breeding
Neart na Gaoithe [1]		
98% avoidance	3	0
99% avoidance	2	0
Seagreen (Project Alpha) [2]		
98% avoidance	2 [3]	6
99% avoidance	1 [3]	3
Seagreen (Project Bravo) [2]		
98% avoidance	5 [3]	2
99% avoidance	3 [3]	1
Total		
98% avoidance	10	8
99% avoidance	6	4

1 Based on 3 years data. 2 Based on 2 years data 3 Adults only (Seagreen 2013)

Using a 98% avoidance behavioural rate, a total of 10 lesser black-backed gulls are predicted to collide with Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms during the breeding period.

Not all birds recorded were adults and therefore a proportion of immature birds will not be from the SPA breeding population. Of those that were aged during the breeding period, 28.6% of lesser black-backed gulls recorded at Neart na Gaoithe were immature birds. Seagreen have assessed the number of adult lesser black-backed gulls predicted to be impacted by Projects Alpha and Bravo, taking into account the proportion of immature birds (Seagreen 2013). The number of collision mortalities predicted by Seagreen have been used in this assessment.

Using a 98% avoidance behavioural rate, and accounting for the proportion of non-breeding immature birds recorded during the breeding period, the total number of adult lesser black-backed gulls at risk of collision during the breeding period at Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms is nine birds. This equates to approximately 0.15% of the regional SPA population (2,854 pairs) (SMP 2013) in the breeding period.

The predicted cumulative impacts on lesser black-backed gulls from Neart na Gaoithe and Inch Cape in the breeding and non-breeding periods are presented in Table 4.63.

Table 4.63 Predicted number of cumulative lesser black-backed gulls (adult & immature) mortality with Inch Cape in the breeding and non-breeding periods

Wind Farm	Breeding	Non-breeding
Neart na Gaoithe [1]		
98% avoidance	3	0
99% avoidance	2	0
Inch Cape [2]		
98% avoidance	76 [3]	Not available
99% avoidance	38 [3]	Not available
Total [-4]		
98% avoidance	79	-
99% avoidance	40	-

1 Based on 3 years data. 2 Based on 1 years data 3 Adults only

Using a 98% avoidance behavioural rate, a total of 79 lesser black-backed gulls are predicted to collide with Neart na Gaoithe and Inch Cape offshore wind farms during the breeding period.

Of those that were aged during the breeding period, 28.6% of lesser black-backed gulls recorded at Neart na Gaoithe. Collision estimates from Inch Cape were of adult birds only (ICOL, 2012). Using a 98% avoidance behavioural rate, and accounting for the proportion of non-breeding immature birds recorded during the breeding period the total number of adult lesser black-backed gulls at risk of collision during the breeding period at Neart na Gaoithe and Inch Cape offshore wind farms is 78 birds. This equates to approximately 1.4% of the regional SPA population (2,854 pairs) (SMP 2013) in the breeding period.

The predicted cumulative impacts on lesser black-backed gulls from Neart na Gaoithe, Seagreen projects Alpha and Bravo and Inch Cape in the breeding and non-breeding periods are presented in Table 4.64.

Using a 98% avoidance behavioural rate, a total of 86 lesser black-backed gulls are predicted to collide with Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms during the breeding period.

Accounting for the proportion of non-breeding immature birds recorded during the breeding period the total number of adult lesser black-backed gulls at risk of collision during the breeding period at Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms is 85 birds. This equates to approximately 1.5% of the regional SPA population (2,854 pairs) (SMP 2013) in the breeding period.

Table 4.64 Predicted number of cumulative lesser black-backed gull (adult & immature) mortality with Seagreen Projects Alpha and Bravo and Inch Cape in the breeding and non-breeding periods

Wind Farm	Breeding	Non-breeding
Neart na Gaoithe [1]		
98% avoidance	3	0
99% avoidance	2	0
Seagreen (Project Alpha) [2]		
98% avoidance	2 [4]	6
99% avoidance	1 [4]	3
Seagreen (Project Bravo) [2]		
98% avoidance	5 [4]	10
99% avoidance	3 [4]	5
Inch Cape [3]		
98% avoidance	76 [4]	Not available
99% avoidance	38 [4]	Not available
Total [-4]		
98% avoidance	86	16+
99% avoidance	44	8+

1 Based on 3 years data. 2 Based on 2 years data 3 Based on 1 Years data 4 Adults only

It is concluded that the cumulative collision mortality for the regional lesser black-backed gulls population in the breeding period is an effect of low magnitude, that is temporally long-term and reversible.

No lesser black-backed gulls were predicted to collide with turbines at Neart na Gaoithe in the non-breeding period. It is therefore not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for the regional population in the non-breeding period. As a result, no further cumulative impact assessment for collision risk was undertaken for this species.

It is concluded that for the four projects examined here, the effects of cumulative collision mortality on lesser black-backed gulls from the regional population in the breeding and non-breeding periods are ***not significant*** under the Electricity Regulations.

4.4.15.6 Mitigation measures

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.

4.4.16 Herring gull *Larus argentatus*

4.4.16.1 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).

4.4.16.2 Offshore site and 8 km buffer area

A total of 1,723 herring gulls were recorded on surveys in the offshore site and 8 km buffer area in Year 1, however only 50 birds were recorded in the offshore site (Table 4.2) (raw numbers, all sea states). In Year 2, a total of 1,433 herring gulls were recorded on surveys, however only 58 birds were seen in the offshore site over the year. In Year 3, a total of 800 were recorded in the study area, with 54 birds recorded in the offshore site.

During the Year 1 breeding season (April to August), the mean estimated number of herring gulls in the offshore site was five birds, with a peak of 17 birds in June (Table 4.65). In the same period of Year 2, the mean estimated number of herring gulls in the offshore site was 14 birds, with a peak of 50 birds in July. In Year 3, the mean estimated number of herring gulls during the breeding season was 20 birds, with a peak of 53 birds in June.

In the Year 1 non-breeding season (September to March), the mean estimated number of herring gulls in the offshore site was 18 birds, with a peak of 39 birds in March (Table 4.65). In the same period of Year 2, the mean estimated number of herring gulls in the offshore site was 16 birds, with a peak of 41 birds in January. In Year 3, the mean estimated number of herring gulls during the non-breeding season was two birds, with a peak of 14 birds in January.

Mean estimated numbers of herring gulls in the offshore site were very low throughout Years 1 to 3 (Figure 4.56). In the buffer area, mean estimated numbers peaked at 2,319 birds in November, with a lower mean peak of 1,754 birds in January (three-year mean).

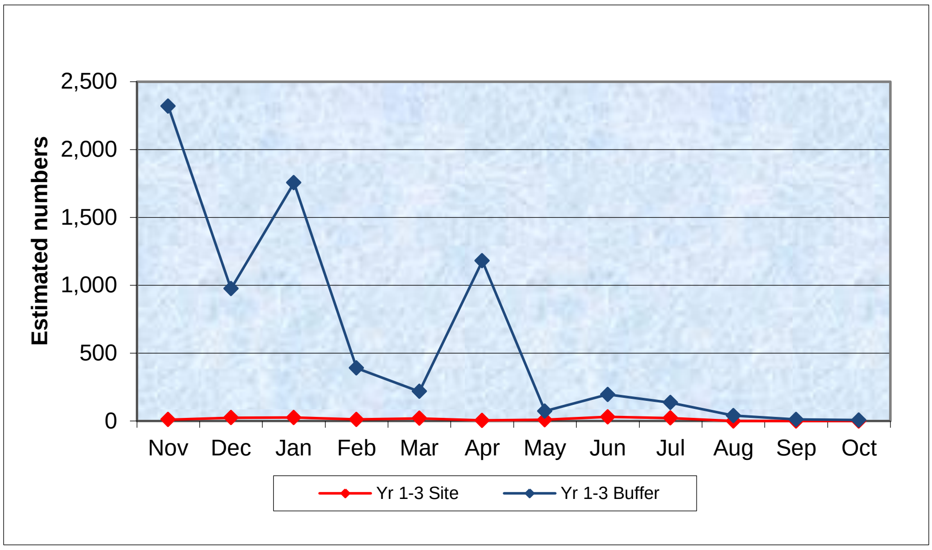


Figure 4.56 Mean monthly estimated numbers of herring gulls in the offshore site and buffer area in Years 1 to 3 (Three-year mean)

However, these estimates were probably inflated by the presence of fishing vessels in the buffer area with large numbers of herring gulls associating with them, and should therefore be treated with caution as they may not reflect typical conditions.

Table 4.65 Estimated numbers of herring gulls in the offshore site (plus 1, 2 & 8 km buffer) in Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 8 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	0	0	0	20	20	20	61	3,710
Yr1 Dec	0	0	0	27	27	41	41	972
Yr1 Jan	0	0	0	20	20	736	784	3,836
Yr1 Feb	11	3	41	7	18	38	82	118
Yr1 Mar	12	3	44	27	39	46	46	98
Yr1 Apr	0	0	0	7	7	7	7	205
Yr1 May	0	0	0	0	0	0	0	14
Yr1 Jun	11	4	31	7	17	17	28	377
Yr1 Jul	0	0	0	0	0	0	0	7
Yr1 Aug	0	0	0	0	0	0	0	28
Yr1 Sep	0	0	0	0	0	7	7	7
Yr1 Oct	0	0	0	0	0	0	0	19
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	15	3	81	7	22	22	29	1,026
Yr2 Jan	0	0	0	41	41	88	116	625
Yr2 Feb	0	0	0	14	14	20	41	133
Yr2 Mar	8	1	42	13	21	34	40	307
Yr2 Apr	0	0	0	0	0	0	146	3,314
Yr2 May	0	0	0	0	0	0	0	13
Yr2 Jun	0	0	0	20	20	20	20	135
Yr2 Jul	50	9	290	0	50	50	50	226
Yr2 Aug	0	0	0	0	0	0	0	7
Yr2 Sep	0	0	0	0	0	0	0	27
Yr2 Oct	0	0	0	0	0	0	0	0
Yr3 Nov	0	0	0	0	0	0	0	947
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	0	0	0	14	14	14	14	877
Yr3 Feb	0	0	0	0	0	0	209	949
Yr3 Mar	0	0	0	0	0	0	252	309
Yr3 Apr	0	0	0	7	7	20	20	33
Yr3 May	0	0	0	27	27	27	34	219
Yr3 Jun	53	19	147	0	53	53	82	162
Yr3 Jul	0	0	0	14	14	21	21	237
Yr3 Aug	0	0	0	0	0	0	0	85
Yr3 Sep	0	0	0	0	0	0	0	0
Yr3 Oct	0	0	0	0	0	0	0	0

In the Year 1 non-breeding period (September to March), highest densities of herring gulls were recorded in the south-west of the buffer area (Figure 4.57). Densities in the offshore site were low at this time of year. Few birds were recorded in the east of the offshore site and buffer area at this time.

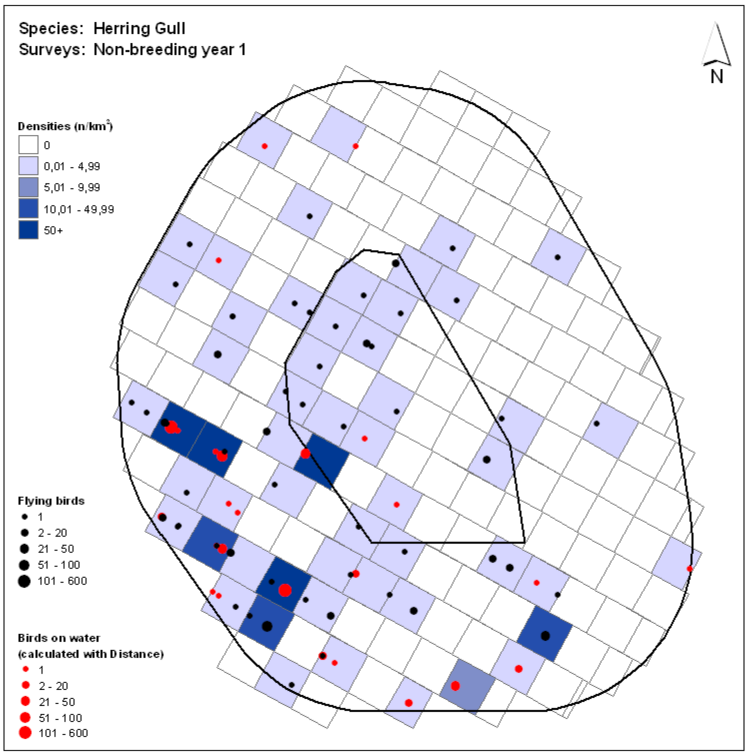


Figure 4.57 Herring gull density in the non-breeding season, Year 1

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 offshore site and buffer area and fewer birds in the north (Figure 4.58). As in Year 1, herring gull densities in the offshore site at this time were low.

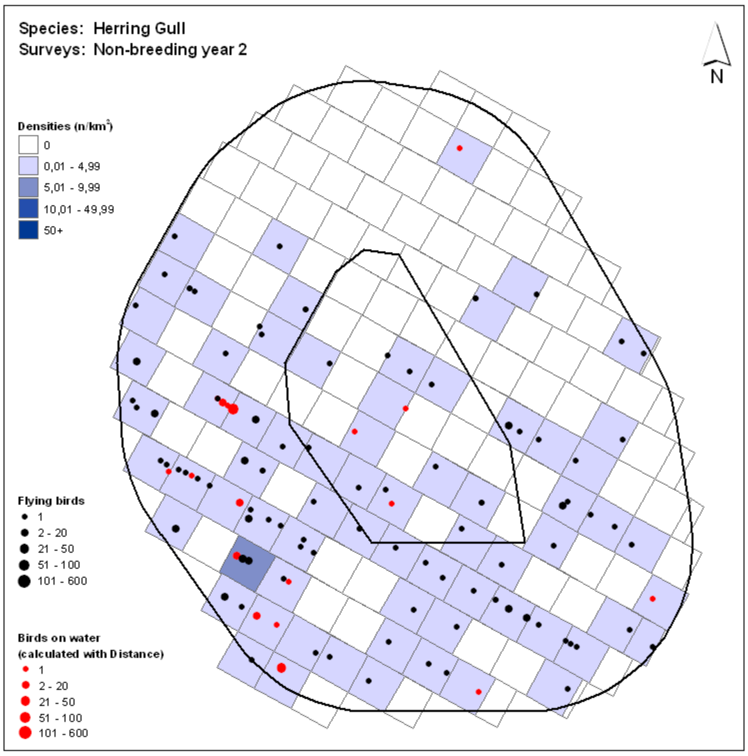


Figure 4.58 Herring gull density in the non-breeding season, Year 2

In the Year 3 non-breeding period, very few herring gulls were recorded in the offshore site (Figure 4.59). Birds were almost entirely restricted to the south-west of the buffer area where high densities were recorded.

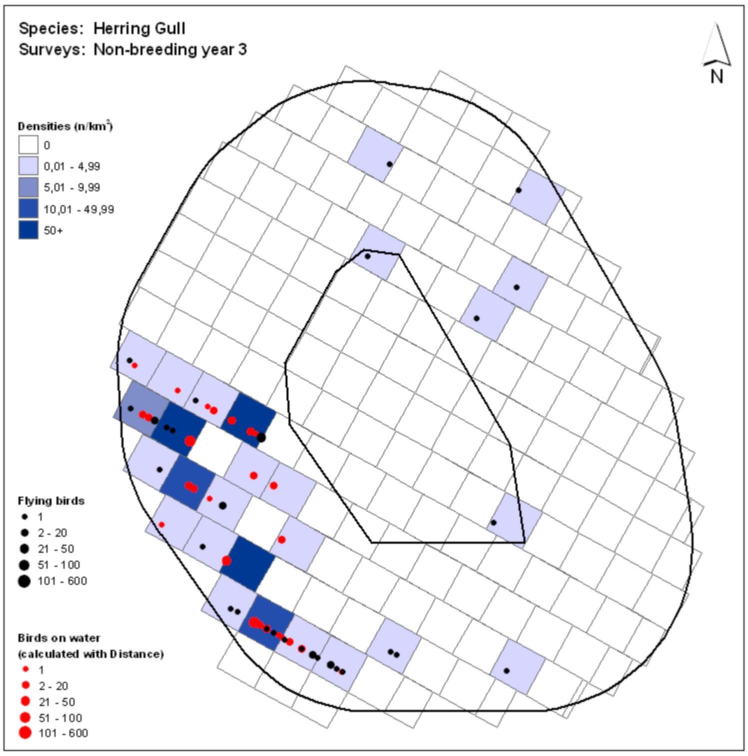


Figure 4.59 Herring gull density in the non-breeding season, Year 3

Herring gull density in the Year 1 breeding season (April to August) was very low in the offshore site, with few birds recorded over the period (Figure 4.60). Herring gulls were slightly more widespread in the buffer area at this time, with highest density recorded in the south of the buffer area.

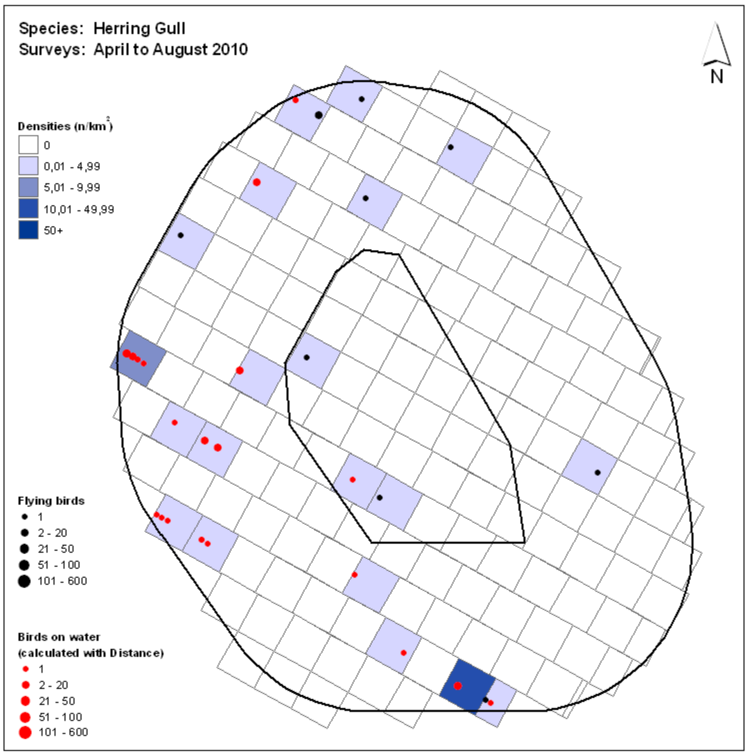


Figure 4.60 Herring gull density in the breeding season, Year 1

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 4.61).

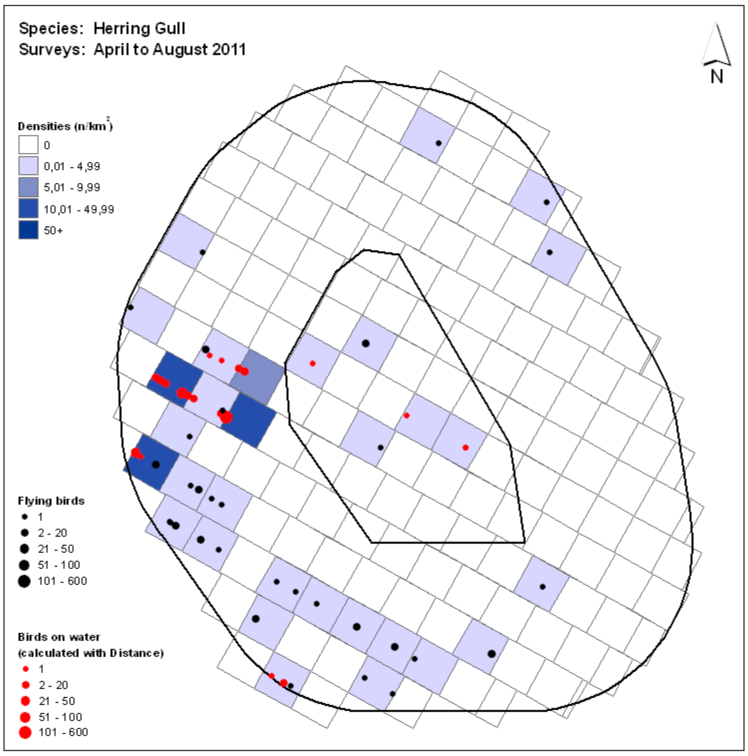


Figure 4.61 Herring gull density in the breeding season, Year 2

Herring gull distribution in the Year 3 breeding season was similar to the same period in Years 1 and 2, with low densities and few birds recorded within the offshore site at this time. Higher densities were recorded in the west and south of the buffer area at this time (Figure 4.62).

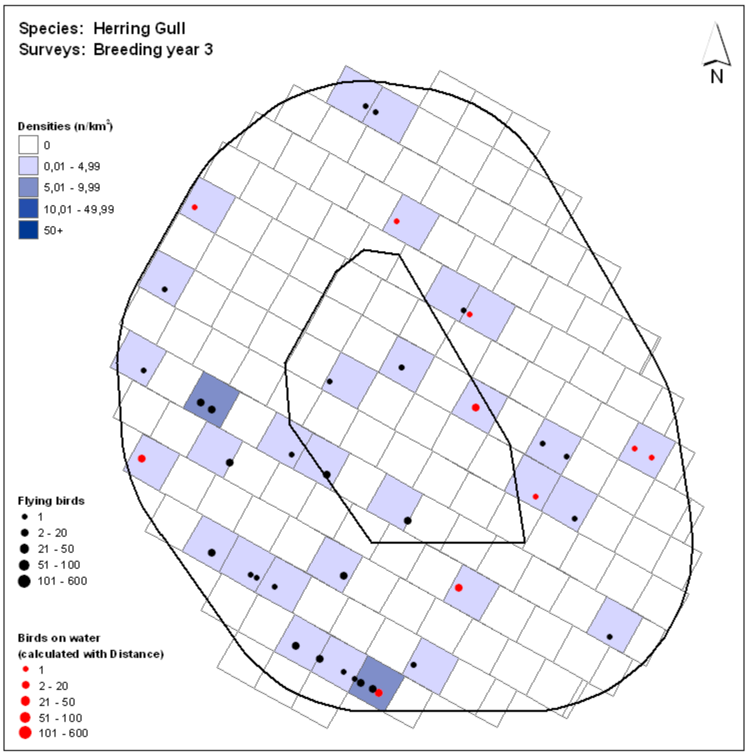


Figure 4.62 Herring gull density in the breeding season, Year 3

A total of 1,646 herring gulls were recorded in flight, with 78.3% of birds flying below 27.5 m (Table 4.3). A total of 357 birds (21.7%) were recorded flying above 27.5 m, i.e. within the rotor swept zone, at estimated heights of between 30 m and 60 m.

Foraging behaviour was recorded for 1,106 herring gulls in the offshore site and 8 km buffer area on baseline surveys, with four types of foraging behaviour recorded, and unspecified feeding behaviour recorded for a further 21 birds (Table 4.66). Scavenging at fishing vessels was the most frequently recorded foraging behaviour (86.7%).

Table 4.66 Herring gull foraging behaviour in the offshore site and 8 km buffer area in Years 1 to 3

Behaviour	Number of birds
Actively searching	53
Dipping	3
Scavenging at fishing vessels	959
Surface pecking	70
Feeding method unspecified	21
Total	**1,106**

A total of 1,700 herring gulls were aged on baseline surveys in the offshore site and 8 km buffer area. In the breeding season (April to August) age was recorded for 610 herring gulls, with 138 immature (non-breeding) birds (22.6%) and 472 adults (77.4%) aged on surveys (Table 4.67).

Table 4.67 Monthly breakdown of immature and adult herring gulls in the offshore site and 8 km buffer area in Years 1 to 3 combined

Month	No of immature birds	Number of adult birds	Number of aged birds	Percentage of immature birds
January	105	247	352	29.8
February	73	194	267	27.3
March	44	94	138	31.9
April	57	115	172	33.1
May	33	7	40	82.5
June	43	150	193	22.3
July	5	150	155	3.2
August	0	50	50	0.0
September	14	8	22	63.6
October	21	2	23	91.3
November	62	56	118	52.5
December	48	122	170	28.2
Total	**505**	**1,195**	**1,700**	**29.7**

4.4.16.3 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).

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 4.68). These SPAs held 9.5% of the UK breeding population and 1.6% of the biogeographic population at the time of designation (JNCC, 2013). Since designation, the populations at these SPAs have decreased (SMP, 2013). 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) (Thaxter *et al.*, 2012). For this assessment, Fowlsheugh SPA was also considered, as the distance between the offshore site and the SPA (62 km) lies just outside this range (62 km). Herring gull mean maximum foraging range from breeding SPAs in relation to the offshore site are shown in Figure 4.63.

Table 4.68 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**	**5,764**	**'99-'01**
Fowlsheugh	*62*	*3,190*	*0.3*	*2.0*	*259*	*2012*
St Abb's Head to Fast Castle	**31**	**1,160**	**0.1**	**0.7**	**605**	**2000**
Total	**-**	**15,242**	**1.6**	**9.5**	**9,742**	**-**

Sources: 1 JNCC (2013) – SPA online species accounts. 2 SMP (2013) – 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).

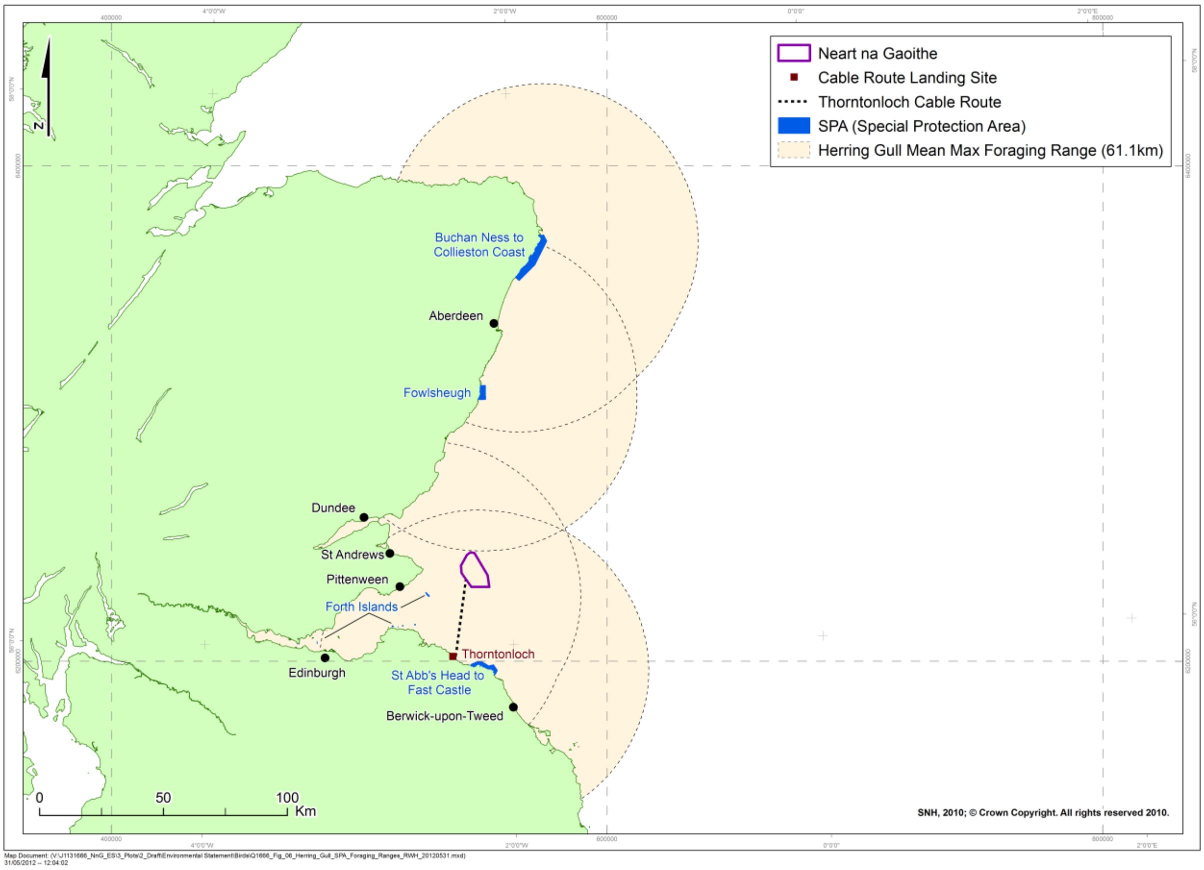


Figure 4.63 Herring gull mean maximum foraging range from breeding SPAs in relation to the Development

4.4.16.4 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 areas. 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 some birds from the regional breeding population will also be present.

Populations

Recent counts on the Isle of May indicate the breeding population of herring gulls has increased slightly since Seabird 2000. However, as recent data was not available for some other breeding colonies in the region, counts from Seabird 2000 have been used in this assessment. On this basis, the regional breeding population between Peterhead and Blyth was estimated as 26,159 breeding pairs (Mitchell *et al.,* 2004).

The regional breeding SPA population within mean maximum foraging range (61.1 km) (Thaxter *et al.,* 2012) is estimated to be 6,628 pairs. This figure is based on most recent available counts from these SPAs (Table 4.68).

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 large numbers of herring gulls also use terrestrial habitats such as agricultural fields and refuse tips at this time of year.

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).

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 are likely to be from SPA breeding populations. The three-year mean peak estimated number of herring gulls present in the breeding season in the offshore site (40 birds) is 0.3% of the regional breeding SPA population (6,628 pairs) within mean maximum foraging range (61.1 km) (Thaxter *et al.,* 2012).

Offshore wind farm studies

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).

Analysis of changes in numbers of large gulls (including herring gull) at Robin Rigg Offshore Wind Farm suggests a possible decline in numbers within the wind farm during the first year of operation, however this pattern was not clear, and further monitoring is required (Walls, *et al.,* 2013).

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.

Construction Phase

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.

Operational Phase

Displacement

In order to assess the potential impacts from displacement, the approach presented in the interim guidance note on displacement issued by the JNCC and NE has been followed (JNCC & NE 2012).

Peak estimated numbers of herring gulls in the offshore site in the breeding season (April to August) and non-breeding season (September to March) for Years 1 to 3 were averaged to get the three-year mean peak per season. Where peak numbers occurred in different months within the same season across different years, the peak month was used. This was repeated for a 1 km and 2 km buffer around the offshore site (Table 4.69).

Table 4.69 Seasonal three-year mean peak estimated numbers of herring gulls in the offshore site (plus 1 & 2 km buffer)

Year	Offshore site		Offshore site + 1 km		Offshore site + 2 km	
	Breeding	Non-breeding	Breeding	Non-breeding	Breeding	Non-breeding
Year 1	17	39	17	736	28	784
Year 2	50	41	50	88	146	116
Year 3	53	14	53	14	82	252
3-year mean peak	40	31	40	279	85	384

Guidance recommends presenting a range of potential displacement and mortality rates and wherever possible selecting a suitable impact based on empirical evidence. Where, there is little evidence to support the assessment a precautionary approach should be taken.

Likely impacts of displacement

As evidence from operational wind farms suggests that only a small proportion of herring gulls are likely be displaced from offshore wind farms, a precautionary level of 10% displacement was used for this assessment.

Breeding period

Baseline surveys recorded very low numbers of herring gulls in the offshore site during the breeding season. Assuming 10% of all herring gulls were to be displaced from the offshore site during the breeding season, this would affect an estimated four birds, (Table 4.70), increasing to nine birds for the offshore site plus 2 km buffer (Table 4.72).

During the breeding period (April to August), 22.6% of aged herring gulls were immature birds (Table 4.67). This percentage was applied to the estimated numbers of displaced herring gulls in the breeding season to estimate the number of adults potentially displaced, as recommended in the draft guidance note on displacement (JNCC & NE, 2012).

Assuming 22.6% of all herring gulls were non-breeding immature birds, it was estimated that if 90% of all adult herring gulls were displaced from the offshore site during the breeding season, this would affect an estimated three birds, increasing to seven birds including the 2 km buffer.

For the purposes of this assessment, it was assumed that 10% of all herring gulls displaced from the offshore site and 2 km buffer during the breeding season (up to one bird) would die as a result. It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

Table 4.70 Estimated number of herring gulls predicted to be at risk of mortality following displacement from offshore site in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	0	0	1	1	2	2	2	3	3	4	4
	20%	0	0	1	2	2	3	4	5	6	6	7	8
	30%	0	1	1	2	4	5	6	7	8	10	11	12
	40%	0	1	2	3	5	6	8	10	11	13	14	16
	50%	0	1	2	4	6	8	10	12	14	16	18	20
	60%	0	1	2	5	7	10	12	14	17	19	22	24
	70%	1	1	3	6	8	11	14	17	20	22	25	28
	80%	1	2	3	6	10	13	16	19	22	26	29	32
	90%	1	2	4	7	11	14	18	22	25	29	32	36
	100%	1	2	4	8	12	16	20	24	28	32	36	40
Three-year mean peak of 40 herring gulls in the offshore site in the breeding season (April to Aug)													
SPA Population within mean max foraging range (61.1 km) = 6,628 pairs (SMP 2013)													
Regional population in breeding season = 26,159 pairs (Mitchell *et al.,* 2004)													

Table 4.71 Estimated number of herring gulls predicted to be at risk of mortality following displacement from offshore site plus 1 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	0	0	1	1	2	2	2	3	3	4	4
	20%	0	0	1	2	2	3	4	5	6	6	7	8
	30%	0	1	1	2	4	5	6	7	8	10	11	12
	40%	0	1	2	3	5	6	8	10	11	13	14	16
	50%	0	1	2	4	6	8	10	12	14	16	18	20
	60%	0	1	2	5	7	10	12	14	17	19	22	24
	70%	1	1	3	6	8	11	14	17	20	22	25	28
	80%	1	2	3	6	10	13	16	19	22	26	29	32
	90%	1	2	4	7	11	14	18	22	25	29	32	36
	100%	1	2	4	8	12	16	20	24	28	32	36	40
Three-year mean peak of 40 herring gulls in the offshore site in the breeding season (April to Aug)													
SPA Population within mean max foraging range (61.1 km) = 6,628 pairs (SMP 2013)													
Regional population in breeding season = 26,159 pairs (Mitchell *et al.,* 2004)													

Table 4.72 Estimated number of herring gulls predicted to be at risk of mortality following displacement from offshore site plus 2 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	0	1	2	3	3	4	5	6	7	8	9
	20%	0	1	2	3	5	7	9	10	12	14	15	17
	30%	1	1	3	5	8	10	13	15	18	20	23	26
	40%	1	2	3	7	10	14	17	20	24	27	31	34
	50%	1	2	4	9	13	17	21	26	30	34	38	43
	60%	1	3	5	10	15	20	26	31	36	41	46	51
	70%	1	3	6	12	18	24	30	36	42	48	54	60
	80%	1	3	7	14	20	27	34	41	48	54	61	68
	90%	2	4	8	15	23	31	38	46	54	61	69	77
	100%	2	4	9	17	26	34	43	51	60	68	77	85
Three-year mean peak of 85 herring gulls in the offshore site in the breeding season (April to Aug)													
SPA Population within mean max foraging range (61.1 km) = 6,628 pairs (SMP 2013)													
Regional population in breeding season = 26,159 pairs (Mitchell *et al.,* 2004)													

For the remaining displaced birds that survived, there could potentially be a detrimental impact on their breeding success, as a result of having to travel further on each trip to forage elsewhere. Assuming this affected up to seven pairs out of the total SPA population of 6,628 pairs, then an estimated 0.1% of the Regional SPA breeding population within mean maximum foraging distance could potentially be affected.

Comparing the distribution of herring gulls in the study area from baseline surveys at this time of year shows that densities in the offshore site were very low, with few birds recorded over the period (Figure 4.60, Figure 4.61 and Figure 4.62). This indicates that breeding herring gulls from colonies in the vicinity are not regularly foraging in the offshore site in the breeding season, and therefore, should displacement occur, it will not affect a significant number of breeding herring gulls.

Non-breeding period

Assuming 10% of all herring gulls were displaced from the offshore site during the non-breeding season, this would affect an estimated three birds (Table 4.73), increasing to 38 birds for the offshore site plus 2 km buffer (Table 4.75). However, given that herring gulls are not tied to a colony in the non-breeding season, and are therefore free to forage further afield, any additional mortality arising from displacement from the offshore site is likely to be minimal. It is concluded that any displaced herring gulls would move to alternative foraging areas over the winter months.

Based on the very low estimated numbers of herring gulls in the offshore site in the breeding and non-breeding seasons, and evidence from wind farm studies indicating a low level of displacement, the regional herring gull population in the breeding and non-breeding seasons 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 population throughout the year.

This impact is categorised as having 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 breeding and non-breeding seasons are ***not significant*** under the EIA Regulations.

Table 4.73 Estimated number of herring gulls predicted to be at risk of mortality following displacement from offshore site in the non-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	0	0	1	1	1	2	2	2	2	3	3
	20%	0	0	1	1	2	2	3	4	4	5	6	6
	30%	0	0	1	2	3	4	5	6	7	7	8	9
	40%	0	1	1	2	4	5	6	7	9	10	11	12
	50%	0	1	2	3	5	6	8	9	11	12	14	16
	60%	0	1	2	4	6	7	9	11	13	15	17	19
	70%	0	1	2	4	7	9	11	13	15	17	20	22
	80%	0	1	2	5	7	10	12	15	17	20	22	25
	90%	1	1	3	6	8	11	14	17	20	22	25	28
	100%	1	2	3	6	9	12	16	19	22	25	28	31
Three-year mean peak of 31 herring gulls in the offshore site in the non-breeding season (Sept to Mar)													
Regional population in the non-breeding season = 200,000 birds (Skov *et al*., 1995)													

Table 4.74 Estimated number of herring gulls predicted to be at risk of mortality following displacement from offshore site plus 1 km buffer in the non-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	1	3	6	8	11	14	17	20	22	25	28
	20%	1	3	6	11	17	22	28	33	39	45	50	56
	30%	2	4	8	17	25	33	42	50	59	67	75	84
	40%	2	6	11	22	33	45	56	67	78	89	100	112
	50%	3	7	14	28	42	56	70	84	98	112	126	140
	60%	3	8	17	33	50	67	84	100	117	134	151	167
	70%	4	10	20	39	59	78	98	117	137	156	176	195
	80%	4	11	22	45	67	89	112	134	156	179	201	223
	90%	5	13	25	50	75	100	126	151	176	201	226	251
	100%	6	14	28	56	84	112	140	167	195	223	251	279
Three-year mean peak of 279 herring gulls in the offshore site in the non-breeding season (Sept to Mar)													
Regional population in the non-breeding season = 200,000 birds (Skov *et al*., 1995)													

Table 4.75 Estimated number of herring gulls predicted to be at risk of mortality following displacement from offshore site plus 2 km buffer in the non-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	2	4	8	12	15	19	23	27	31	35	38
	20%	2	4	8	15	23	31	38	46	54	61	69	77
	30%	2	6	12	23	35	46	58	69	81	92	104	115
	40%	3	8	15	31	46	61	77	92	108	123	138	154
	50%	4	10	19	38	58	77	96	115	134	154	173	192
	60%	5	12	23	46	69	92	115	138	161	184	207	230
	70%	5	13	27	54	81	108	134	161	188	215	242	269
	80%	6	15	31	61	92	123	154	184	215	246	276	307
	90%	7	17	35	69	104	138	173	207	242	276	311	346
	100%	8	19	38	77	115	154	192	230	269	307	346	384
Three-year mean peak of 384 herring gulls in the offshore site in the non-breeding season (Sept to Mar)													
Regional population in the non-breeding season = 200,000 birds (Skov *et al*., 1995)													

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 3.7). 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 3.8). For birds breeding on Craigleith, the wind farm acting as a barrier would potentially block approximately 28% of the possible flight directions (Table 3.7). 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 3.8). 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 3.7). 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 3.8).

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 of low magnitude (<1 %), 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 herring gull population in the breeding season is ***not significant*** under the terms of the Electricity Act.

Collision mortality

CRM estimated the number of potential herring gull collisions per season for four wind turbine designs scenarios (Table 3.6). The minimum height for the turbine blades above the sea surface for all four options is 27.5 m at mean sea level (MSL).

There is no specific SNH guidance on avoidance rates for seabirds, and therefore their default value of 98.0% is presented in this assessment.

Highest estimated collisions in the breeding period were for Option 1 (90 x 5 MW turbines). Using a 98% avoidance rate, the number of predicted herring gull collisions in the breeding period was 26 birds (Table 4.76). This equates to approximately 0.05% of the regional population (26,159 pairs)(Mitchell *et al,* 2004), or 0.2% of the regional SPA population (6,628 pairs) (SMP 2013) in the breeding period.

Lowest estimated collisions in the breeding period were for Options 3 and 4 (73 x 6.15 MW turbines). Using a 98% avoidance rate, the number of predicted herring gull collisions in the breeding period was 20 birds (Table 4.76). This equates to approximately 0.04% of the regional population (26,159 pairs)(Mitchell *et al,* 2004), or 0.15% of the regional SPA population (6,628 pairs) (SMP 2013) in the breeding period.

Baseline surveys between April and August recorded 22.6% of aged herring gulls as immature birds (Table 4.76). The estimated number of collisions for the breeding period was reduced by 22.6% to represent the estimated numbers of adult herring gulls involved in collisions.

Table 4.76 Number of estimated herring gull collisions using avoidance rate of 98% for four wind farm options

	Option 1 (90x5MW)	Option 2 (75x6MW)	Option 3 (73x6.15MW)	Option 4 (73x6.15MW)
Collisions in breeding season (April to August), all ages	26	22	20	20
Collisions in breeding season (April to August), adults only	20	17	17	16
Collisions in non-breeding period (September to March), all ages	58	49	44	44
Total collisions per year, all ages	85	71	64	64

Based on this ratio and using a 98% avoidance rate, the predicted number of adult herring gull collisions in the breeding season for the worst case design option (Option 1) was 20 birds (Table 4.76). This equates to approximately 0.04% of the regional population (26,159 pairs) (Mitchell *et al,* 2004), or 0.15% of the regional SPA population (6,628 pairs) (SMP 2013) in the breeding period.

In the non-breeding period (October to March) and using a 98% avoidance rate, the predicted number of herring gull collisions for the worst case design option (Option 1) was 58 birds (Table 4.76). This equates to approximately 0.03% of the regional population in the non-breeding period (200,000 birds) (Skov *et al.,* 1995).

Lowest estimated collisions in the non-breeding period were for Options 3 and 4 (73 x 6.15 MW turbines). Using a 98% avoidance rate, the number of predicted herring gull collisions in the non-breeding period was 44 birds (Table 4.76). This equates to approximately 0.02% of the regional population in the non-breeding period (200,000 birds) (Skov *et al.,* 1995).

It is concluded that for the most adverse design (Option 1: 90 x 5 MW turbines), collision mortality for herring gull in both the breeding and non-breeding period is an effect of negligible magnitude, that is temporally long-term and reversible.

It is further concluded that for the least adverse designs (Options 3 and 4: 73 x 6.15MW turbines), collision mortality for herring gull is an effect of negligible magnitude, that is temporally long-term and reversible.

It is concluded that for the wind farm designs examined here, the effects of collision mortality on herring gulls from the regional population in the breeding and non-breeding periods are ***not significant*** under the Electricity Regulations.

Decommissioning Phase

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.

Summary of combined effects

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, using a collision avoidance rate of 98.0%, it is judged that the combined magnitude of the three effects is negligible (Table 4.77 and Table 4.78). 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 and non-breeding seasons is ***not significant*** under the EIA Regulations.

Table 4.77 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 (98.0% A R)	Negligible	Long term	Moderate	***Not significant***
All effects combined	Negligible	Long term	Low - Moderate	***Not significant***

Table 4.78 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 (98.0% A R)	Negligible	Long term	Moderate	***Not significant***
All effects combined	Negligible	Long term	Low - Moderate	***Not significant***

4.4.16.5 Cumulative Impact Assessment

Displacement

No significant impacts were predicted to arise from displacement caused by Neart na Gaoithe for herring gulls in the breeding or non-breeding periods. This was based on the numbers of birds recorded within the offshore site, and evidence from other wind farms, indicating likely low levels of displacement for this species. It is therefore not plausible that Neart na Gaoithe could contribute to a significant cumulative displacement impact for the regional population of herring gulls in the breeding or non-breeding period. As a result, no further cumulative impact assessment for displacement was undertaken for this species.

Collision mortality

The potential cumulative collision risk to herring gulls from Neart na Gaoithe and other plans or projects is higher than for Neart na Gaoithe alone.

Projects identified during consultation and the undertaking of the EIA for which there is a potential for a cumulative collision risk impact are:

- Inch Cape offshore wind farm;
- Seagreen Project Alpha offshore wind farm; and
- Seagreen Project Bravo offshore wind farm.

Data on the number of predicted collisions arising from the Seagreen Projects Alpha and Bravo offshore wind farms are presented within the applicant's Environmental Statement (Seagreen 2012) and HRA (Seagreen 2013). The proposed Seagreen Round 3 Zone development consists of a three phase programme of six separate developments. Projects Alpha and Bravo are the first developments and the only Seagreen projects for which applications have been made.

No application has been made for proposed Inch Cape offshore wind farm. Predicted number of collisions arising from the Inch Cape offshore wind farm have been obtained from an unpublished annual ornithological report based on one years data (ICOL, 2012).

The predicted cumulative impacts on herring gulls from Neart na Gaoithe and Seagreen projects Alpha and Bravo in the breeding and non-breeding periods are presented in Table 4.79.

Using a 98% avoidance behavioural rate, a total of 67 herring gulls are predicted to collide with Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms during the breeding period.

Table 4.79 Predicted number of cumulative herring gulls (adult & immature) mortality with Seagreen Projects Alpha and Bravo in the breeding and non-breeding periods

Wind Farm	Breeding	Non-breeding
Neart na Gaoithe [1]		
98% avoidance	26	59
99% avoidance	13	29
Seagreen (Project Alpha) [2]		
98% avoidance	25 [3]	51
99% avoidance	13 [3]	25
Seagreen (Project Bravo) [2]		
98% avoidance	16 [3]	32
99% avoidance	8 [3]	16
Total		
98% avoidance	67	142
99% avoidance	34	70

1 Based on 3 years data. 2 Based on 2 years data

Not all birds recorded were adults and therefore a proportion of immature birds will not be from the SPA breeding population. Of those that were aged during the breeding period, 22.6% of herring gulls recorded at Neart na Gaoithe were immature birds (20 adults). Collision estimates provided by Seagreen were of adult birds only (Seagreen 2012).

Using a 98% avoidance behavioural rate, and accounting for the proportion of non-breeding immature birds recorded during the breeding period, the total number of adult herring gulls at risk of collision during the breeding period at Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms is 61 birds. This equates to approximately 0.1% of the regional population (26,159 pairs)(Mitchell *et al,* 2004), or 0.5% of the regional SPA population (6,628 pairs) (SMP 2013) in the breeding period.

Using a 98% avoidance behavioural rate, the total number of herring gulls at risk of collision during the non-breeding period at Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms is 142 birds. This equates to approximately 0.07% of the regional population in the non-breeding period (200,000 birds) (Skov *et al.,* 1995).

The predicted cumulative impacts on herring gulls from Neart na Gaoithe and Inch Cape in the breeding and non-breeding periods are presented in Table 4.80.

Using a 98% avoidance behavioural rate, a total of 110 herring gulls are predicted to collide with Neart na Gaoithe and Inch Cape offshore wind farms during the breeding period.

Table 4.80 Predicted number of cumulative herring gulls (adult & immature) mortality with Inch Cape in the breeding and non-breeding periods

Wind Farm	Breeding	Non-breeding
Neart na Gaoithe [1]		
98% avoidance	26	59
99% avoidance	13	29
Inch Cape [2]		
98% avoidance	84 [3]	Not available
99% avoidance	42 [3]	Not available
Total		
98% avoidance	110	59+
99% avoidance	55	29+

1 Based on 3 years data. 2 Based on 1 years data 3 Adults only (ICOL, 2012)

Of those that were aged during the breeding period, 22.6% of herring gulls recorded at Neart na Gaoithe were immature birds. Collision estimates from Inch Cape were of adult birds only (ICOL, 2012). Using a 98% avoidance behavioural rate, and accounting for the proportion of non-breeding immature birds recorded during the breeding period, the total number of adult herring gulls at risk of collision during the breeding period at Neart na Gaoithe and Inch Cape offshore wind farms is 104 adult birds. This equates to approximately 0.2% of the regional population (26,159 pairs) (Mitchell *et al,* 2004), or 0.8% of the regional SPA population (6,628 pairs) (SMP 2013) in the breeding period.

Predicted number of herring gull collisions in the non-breeding period for Inch Cape were not available. Assuming the number of collisions was the same as for Neart na Gaoithe (59 birds), and using a 98% avoidance behavioural rate, the total number of herring gulls at risk of collision during the non-breeding period at Neart na Gaoithe and Inch Cape offshore wind farms is estimated to be 118 birds. This equates to approximately 0.06% of the regional population in the non-breeding period (200,000 birds) (Skov *et al.,* 1995).

The predicted cumulative impacts on herring gulls from Neart na Gaoithe, Seagreen projects Alpha and Bravo and Inch Cape in the breeding and non-breeding periods are presented in Table 4.81.

Using a 98% avoidance behavioural rate, a total of 151 herring gulls are predicted to collide with Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms during the breeding period (Table 4.81).

Accounting for the proportion of non-breeding immature birds recorded during the breeding period the total number of adult herring gulls at risk of collision during the breeding period at Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms is 145 adult birds. This equates to approximately 0.3% of the regional population (26,159 pairs) (Mitchell *et al,* 2004), or 1.1% of the regional SPA population (6,628 pairs) (SMP 2013) in the breeding period.

It is concluded that the cumulative collision mortality for the regional herring gulls population in the breeding period is an effect of low magnitude, that is temporally long-term and reversible.

Table 4.81 Predicted number of cumulative herring gull (adult & immature) mortality with Seagreen Projects Alpha and Bravo and Inch Cape in the breeding and non-breeding periods

Wind Farm	Breeding	Non-breeding
Neart na Gaoithe [1]		
98% avoidance	26	59
99% avoidance	13	29
Seagreen (Project Alpha) [2]		
98% avoidance	25 [4]	51
99% avoidance	13 [4]	25
Seagreen (Project Bravo) [2]		
98% avoidance	16 [4]	32
99% avoidance	8 [4]	16
Inch Cape [3]		
98% avoidance	84 [4]	Not available
99% avoidance	42 [4]	Not available
Total [4]		
98% avoidance	151	142+
99% avoidance	76	70+

1 Based on 3 years data. 2 Based on 2 years data 3 Based on 1 Years data 4 Adults only

Predicted number of herring gull collisions in the non-breeding period for Inch Cape were not available. Using the number predicted for Neart na Gaoithe (59 birds), and a 98% avoidance behavioural rate, the total number of herring gulls at risk of collision during the non-breeding period at Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms is estimated to be 201 birds. This equates to approximately 0.1% of the regional population in the non-breeding period (200,000 birds) (Skov *et al.,* 1995).

Based on this, it is concluded that the cumulative collision mortality for the regional herring gulls population in the non-breeding period is an effect of negligible magnitude, that is temporally long-term and reversible.

It is concluded that for the four projects examined here, the effects of cumulative collision mortality on herring gulls from the regional population in the breeding and non-breeding periods are ***not significant*** under the Electricity Regulations.

4.4.16.6 Mitigation measures

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.

4.4.17 Great black-backed gull *Larus marinus*

4.4.17.1 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 40 breeding pairs in 2012 (SMP, 2013). Great black-backed gulls are omnivorous, foraging at sea, estuaries and beaches, and less commonly at rubbish dumps (Forrester *et al*., 2007).

4.4.17.2 Offshore site and 8 km buffer area

A total of 528 great black-backed gulls were recorded on surveys in the offshore site and 8 km buffer area in Year 1, however only 25 birds were seen in the offshore site (Table 4.2) (raw numbers, all sea states). 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. In Year 3, a total of 225 were recorded on surveys in the study area, with 17 birds in the offshore site.

During the Year 1 breeding season (April to August), no great black-backed gulls were recorded in the offshore site (Table 4.82). In the same period of Year 2, the mean estimated number of great black-backed gulls in the offshore site was one bird, with a peak of seven birds in June. In Year 3, the mean estimated number of great black-backed gulls during the breeding season was two birds, with a peak of nine birds in June.

In the non-breeding season (September to March), the mean estimated number of great black-backed gulls in the offshore site was nine birds, with a peak of 21 birds in December (Table 4.82). In the same period of Year 2, the mean estimated number of great black-backed gulls in the offshore site was five birds, with a peak of 13 birds in December. In Year 3, the mean estimated number of great black-backed gulls during the non-breeding season was five birds, with a peak of 20 birds in January.

Table 4.82 Estimated numbers of great black-backed gulls in the offshore site (plus 1, 2 & 8 km buffer) in Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 2 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	0	0	0	7	7	7	8	512
Yr1 Dec	0	0	0	21	21	21	27	76
Yr1 Jan	0	0	0	14	14	355	355	1,737
Yr1 Feb	2	0	9	0	2	2	8	17
Yr1 Mar	10	3	40	0	10	10	10	21
Yr1 Apr	0	0	0	0	0	0	0	21
Yr1 May	0	0	0	0	0	0	0	0
Yr1 Jun	0	0	0	0	0	0	0	0
Yr1 Jul	0	0	0	0	0	0	0	0
Yr1 Aug	0	0	0	0	0	0	0	0
Yr1 Sep	0	0	0	7	7	7	14	69
Yr1 Oct	0	0	0	0	0	0	27	208
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	6	1	29	7	13	20	20	404
Yr2 Jan	0	0	0	7	7	14	27	212
Yr2 Feb	0	0	0	0	0	0	0	7
Yr2 Mar	0	0	2	0	0	7	7	37
Yr2 Apr	0	0	0	0	0	0	21	411
Yr2 May	0	0	0	0	0	0	0	0
Yr2 Jun	0	0	0	7	7	7	7	7
Yr2 Jul	0	0	0	0	0	0	0	0
Yr2 Aug	0	0	0	0	0	0	0	0
Yr2 Sep	0	0	0	7	7	14	284	512
Yr2 Oct	0	0	0	0	0	14	14	51
Yr3 Nov	0	0	0	7	7	21	21	641
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	0	0	0	20	20	20	20	156
Yr3 Feb	0	0	0	0	0	0	21	109
Yr3 Mar	0	0	0	0	0	0	28	49
Yr3 Apr	0	0	0	0	0	0	0	0
Yr3 May	0	0	0	0	0	0	0	48
Yr3 Jun	9	2	56	0	9	9	13	19
Yr3 Jul	0	0	0	0	0	0	0	26
Yr3 Aug	0	0	0	0	0	0	0	0
Yr3 Sep	0	0	0	0	0	0	7	7
Yr3 Oct	0	0	0	0	0	0	0	14

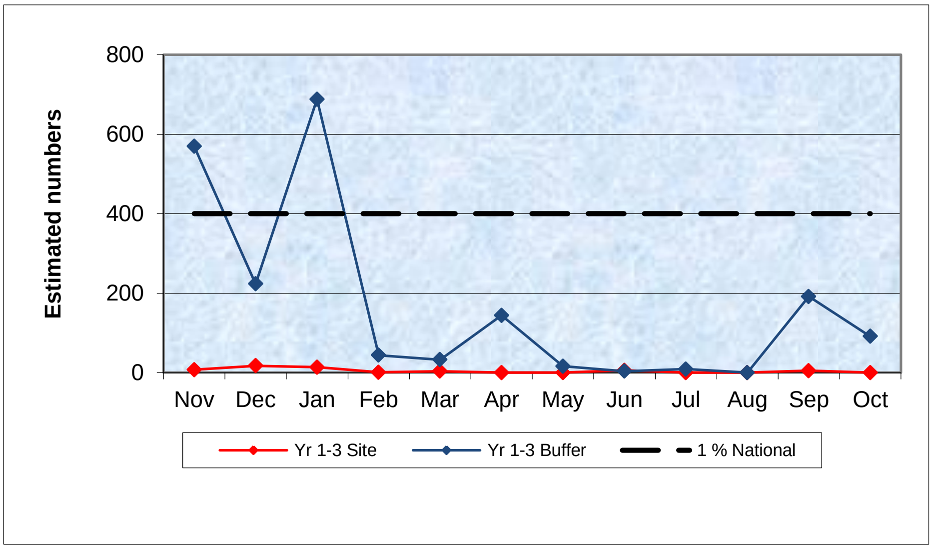


Figure 4.64 Mean monthly estimated numbers of great black-backed gulls in the offshore site & buffer areas in Years 1 to 3 (Three-year mean)

Overall, mean monthly estimated numbers in the offshore site were very low in all three baseline years (Figure 4.64). Estimated numbers in the buffer area were higher outside the breeding season, and exceeded the 1% threshold of national importance (400 birds) (Holt *et al*., 2011) in November (570 birds) and January (688 birds). However, these estimates were probably inflated by the presence of fishing vessels in the buffer area with large numbers of great black backed gulls associating with them, and should therefore be treated with caution as they may not reflect typical conditions.

In the Year 1 non-breeding period (September to March), generally low densities of great black-backed gulls were recorded sporadically in the offshore site (Figure 4.65). Highest densities were recorded in the south-west of the buffer area, with fewer birds elsewhere.

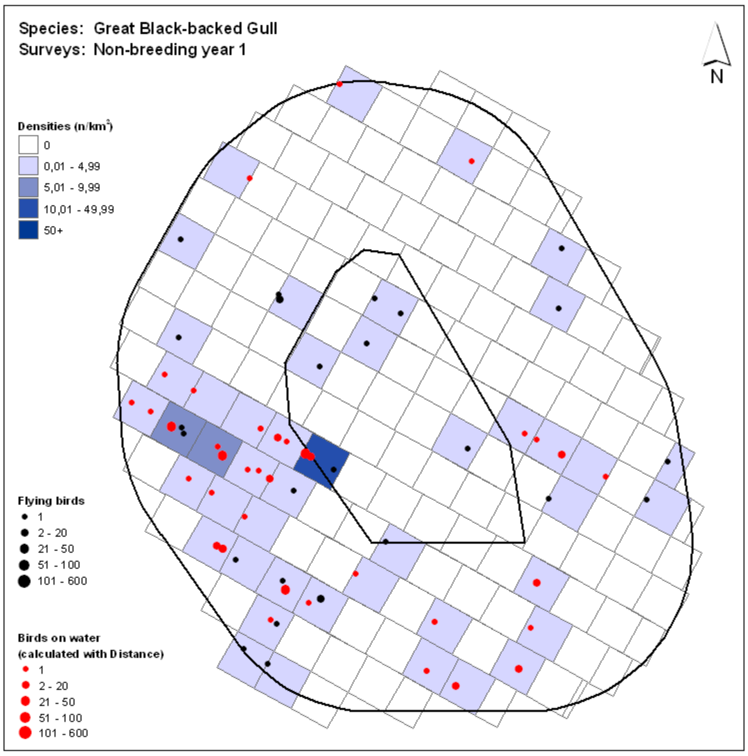


Figure 4.65 Great black-backed gull density in the non-breeding season, Year 1

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 4.66). Low densities were recorded occasionally in the offshore site at this time.

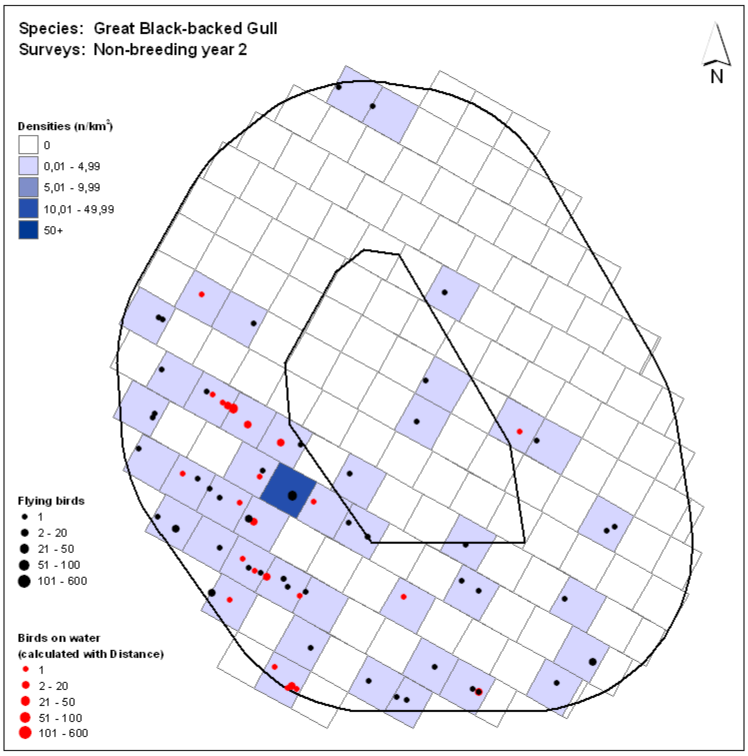


Figure 4.66 Great black-backed gull density in the non-breeding season, Year 2

The Year 3 distribution pattern of great black-backed gulls was similar to the same period of Years 1 and 2, with the majority of great black-backed gulls recorded at low densities in the south-west of the buffer area (Figure 4.67). Low densities were recorded occasionally in the offshore site at this time.

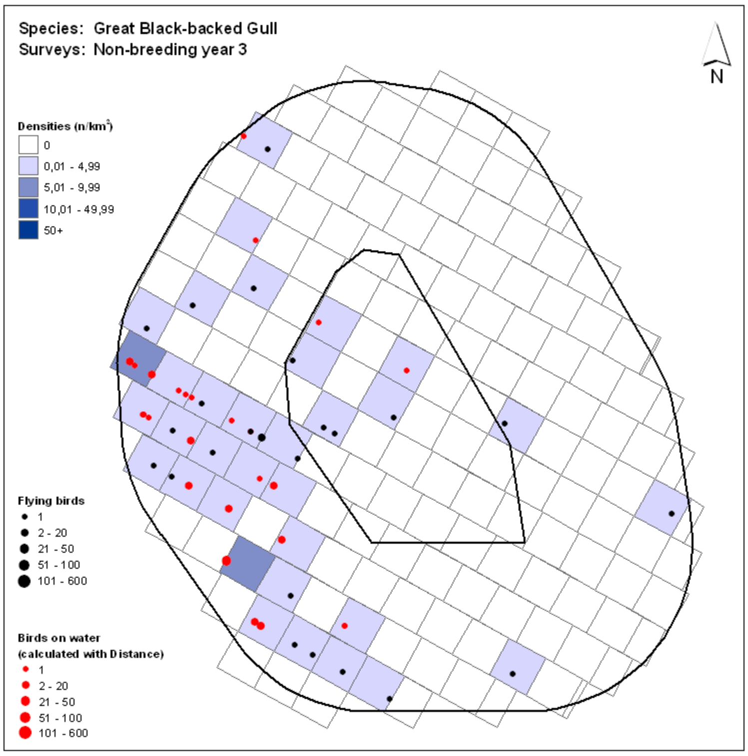


Figure 4.67 Great black-backed gull density in the non-breeding season, Year 3

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 4.68).

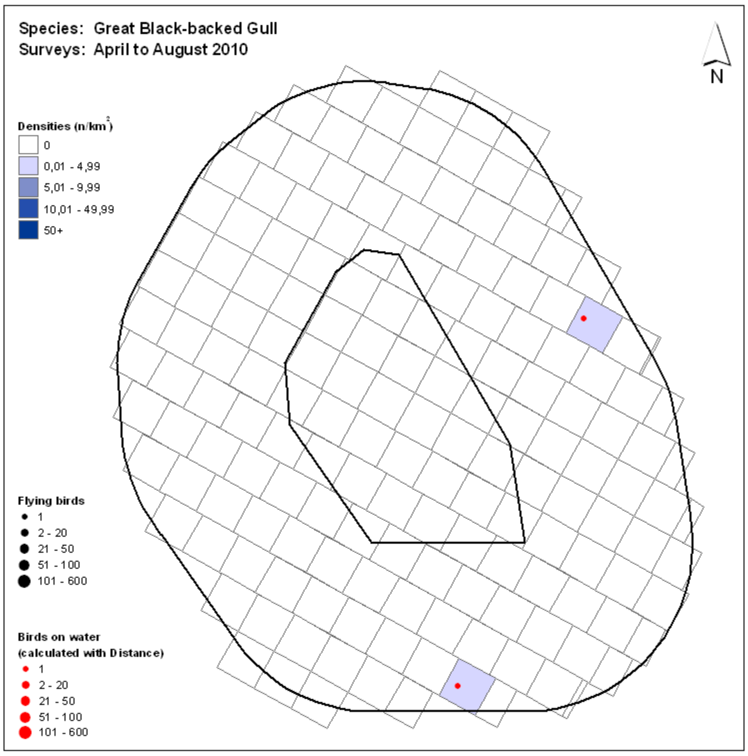


Figure 4.68 Great black-backed gull density in the breeding season, Year 1

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 4.69).

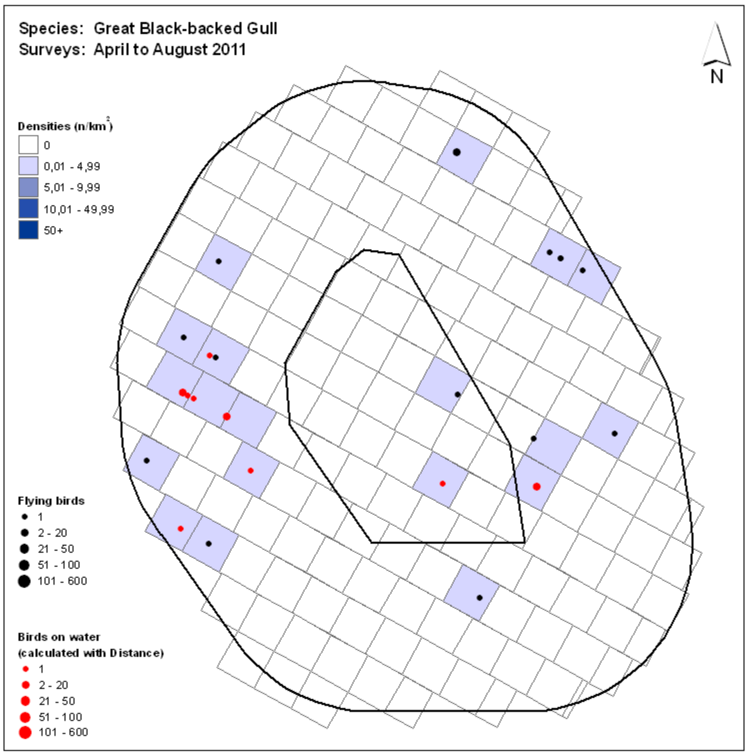


Figure 4.69 Great black-backed gull density in the breeding season, Year 2

The distribution of great black-backed gulls in the Year 3 breeding season was similar to the previous two years, with low densities and few birds recorded in the offshore site at this time. In the buffer area, birds were scattered at low densities (Figure 4.70).

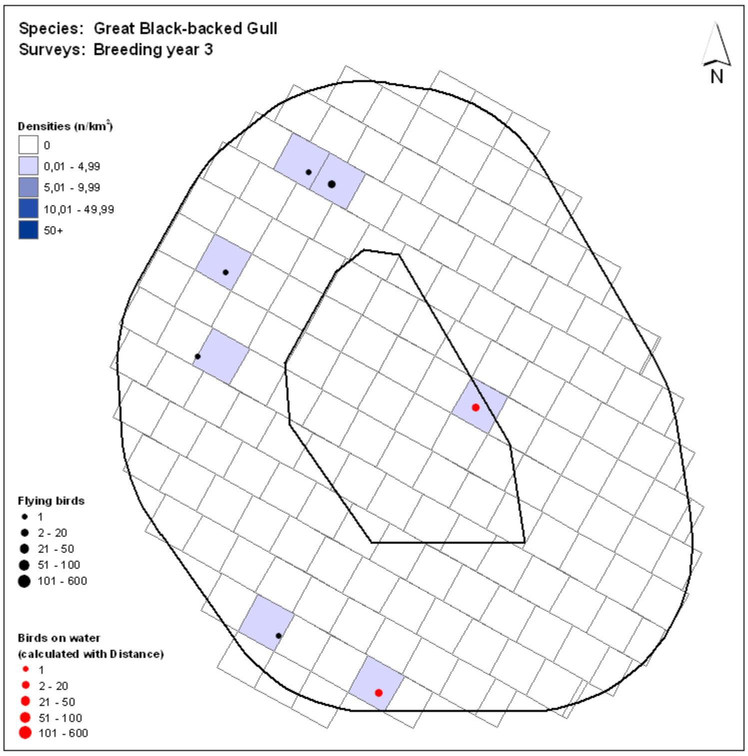


Figure 4.70 Great black-backed gull density in the breeding season, Year 3

A total of 553 great black-backed gulls were recorded in flight on baseline surveys, with 80.7% of birds flying below 27.5 m (Table 4.3). A total of 107 birds (19.3%) were recorded flying above 27.5 m, i.e. within the rotor swept zone, at estimated heights of between 30 m and 60 m.

Foraging behaviour was recorded for 215 great black-backed gulls in the offshore site and 8 km buffer area on baseline surveys, with five types of foraging behaviour recorded, and unspecified feeding behaviour recorded for a further 20 birds (Table 4.83). The majority of all foraging birds were recorded scavenging at fishing vessels (87.2%).

Table 4.83 Great black-backed gull foraging behaviour in the offshore site and 8 km buffer area in Years 1 to 3

Behaviour	Number of birds
Actively searching	8
Holding fish	2
Scavenging at fishing vessels	164
Surface pecking	20
Kleptoparasitising (chasing other birds for food)	1
Feeding method unspecified	20
Total	**188**

A total of 564 great black-backed gulls were aged during baseline surveys in the offshore site and 8 km buffer area. In the breeding season (April to August) age was recorded for 70 great black-backed gulls, with 40 immature (non-breeding) birds (57.1%) and 30 adults (42.9%) aged on surveys (Table 4.84).

Table 4.84 Monthly breakdown of immature and adult great black-backed gulls in the offshore site and 8 km buffer area in Years 1 to 3 combined

Month	No of immature birds	Number of adult birds	Number of aged birds	Percentage of immature birds
January	58	54	112	51.8%
February	35	13	48	72.%
March	21	11	32	65.6%
April	26	12	38	68.4%
May	11	3	14	78.6%
June	2	7	9	22.2%
July	1	2	3	33.3%
August	0	6	6	0.0%
September	51	27	78	65.4%
October	31	30	61	50.8%
November	37	50	87	42.5%
December	33	43	76	43.4%

4.4.17.3 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).

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).

4.4.17.4 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 areas. 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).

Populations

Recent counts on the Isle of May indicate the breeding population of great black-backed gulls has increased since Seabird 2000. However, as recent data was not available for some other breeding colonies in the region, counts from Seabird 2000 have been used in this assessment. On this basis, the regional breeding population between Peterhead and Blyth was estimated as 96 breeding pairs (Mitchell *et al.,* 2004).

As the maximum known foraging range of great black-backed gulls in the breeding season is very restricted (less than 10 km) (Roos *et al*., 2010), and the nearest SPA for the species is approximately 297 km away, the regional SPA population within mean maximum foraging range was not used for the assessment of this species.

The size of the non-breeding-period regional great black-backed gull population is assumed to be 21,600 birds. This is derived by summing the November to February period estimates for localities 4, 5 and 6 given in Skov *et al*. (1995).

Nature conservation importance

Great black-backed gull is rated as low NCI during the breeding season, due to the very low numbers of birds recorded in the offshore site in the breeding season.

Offshore wind farm studies

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).

Analysis of changes in numbers of large gulls (including great black-backed gull) at Robin Rigg Offshore Wind Farm suggests a possible decline in numbers within the wind farm during the first year of operation, however this pattern was not clear, and further monitoring is required (Walls, *et al.,* 2013).

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.

Construction Phase

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 population of great black-backed gulls in the breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

Operational Phase

Displacement

In order to assess the potential impacts from displacement, the approach presented in the interim guidance note on displacement issued by the JNCC and NE has been followed (JNCC & NE 2012).

Peak estimated numbers of great black-backed gulls in the offshore site in the breeding season (April to August) and non-breeding season (September to March) for Years 1 to 3 were averaged to get the three-year mean peak per season. Where peak numbers occurred in different months within the same season across different years, the peak month was used. This was repeated for a 1 km and 2 km buffer around the offshore site (Table 4.85).

Table 4.85 Seasonal three-year mean peak estimated numbers of great black-backed gulls in the offshore site (plus 1 & 2 km buffer)

Year	Offshore site		Offshore site + 1 km		Offshore site + 2 km	
	Breeding	Non-breeding	Breeding	Non-breeding	Breeding	Non-breeding
Year 1	0	21	0	355	0	355
Year 2	7	13	7	20	21	284
Year 3	9	20	9	21	13	28
3-year mean peak	5	18	5	132	11	222

Guidance recommends presenting a range of potential displacement and mortality rates and wherever possible selecting a suitable impact based on empirical evidence. Where, there is little evidence to support the assessment a precautionary approach should be taken.

Likely impacts of displacement

As evidence from operational wind farms suggests that only a small proportion of great black-backed gulls are likely be displaced from offshore wind farms, a precautionary level of 10% displacement was used for this assessment.

Breeding period

Baseline surveys recorded very low numbers of great black-backed gulls in the offshore site during the breeding season. Assuming 10% of all great black-backed gulls were to be displaced from the offshore site plus 2 km buffer during the breeding season, this would affect one bird (Table 4.88). For the purposes of this assessment, it was assumed that 10% of all great black-backed gulls displaced from the offshore site and 2 km buffer during the breeding season (up to 0.1 birds) would die as a result. It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

Table 4.86 Estimated number of great black-backed gulls predicted to be at risk of mortality following displacement from offshore site in the breeding season

	Mortality (%)												
	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
Displacement level (%)	10%	0	0	**0**	0	0	0	0	0	0	0	0	1
	20%	0	0	0	0	0	0	1	1	1	1	1	1
	30%	0	0	0	0	0	1	1	1	1	1	1	2
	40%	0	0	0	0	1	1	1	1	1	2	2	2
	50%	0	0	0	1	1	1	1	2	2	2	2	3
	60%	0	0	0	1	1	1	2	2	2	2	3	3
	70%	0	0	0	1	1	1	2	2	2	3	3	4
	80%	0	0	0	1	1	2	2	2	3	3	4	4
	90%	0	0	0	1	1	2	2	3	3	4	4	5
	100%	0	0	1	1	2	2	3	3	4	4	5	5
Three-year mean peak of five great black-backed gulls in the offshore site in the breeding season (April to Aug)													
Regional population in breeding season = 96 pairs (Mitchell *et al.*, 2004)													

Table 4.87 Estimated number of great black-backed gulls predicted to be at risk of mortality following displacement from offshore site plus 1 km buffer in the breeding season

	Mortality (%)												
	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
Displacement level (%)	10%	0	0	**0**	0	0	0	0	0	0	0	0	1
	20%	0	0	0	0	0	0	1	1	1	1	1	1
	30%	0	0	0	0	0	1	1	1	1	1	1	2
	40%	0	0	0	0	1	1	1	1	1	2	2	2
	50%	0	0	0	1	1	1	1	2	2	2	2	3
	60%	0	0	0	1	1	1	2	2	2	2	3	3
	70%	0	0	0	1	1	1	2	2	2	3	3	4
	80%	0	0	0	1	1	2	2	2	3	3	4	4
	90%	0	0	0	1	1	2	2	3	3	4	4	5
	100%	0	0	1	1	2	2	3	3	4	4	5	5
Three-year mean peak of five great black-backed gulls in the offshore site in the breeding season (April to Aug)													
Regional population in breeding season = 96 pairs (Mitchell *et al.*, 2004)													

Table 4.88 Estimated number of great black-backed gulls predicted to be at risk of mortality following displacement from offshore site plus 2 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	0	**0**	0	0	0	1	1	1	1	1	1
	20%	0	0	0	0	1	1	1	1	2	2	2	2
	30%	0	0	0	1	1	1	2	2	2	3	3	3
	40%	0	0	0	1	1	2	2	3	3	4	4	4
	50%	0	0	1	1	2	2	3	3	4	4	5	6
	60%	0	0	1	1	2	3	3	4	5	5	6	7
	70%	0	0	1	2	2	3	4	5	5	6	7	8
	80%	0	0	1	2	3	4	4	5	6	7	8	9
	90%	0	0	1	2	3	4	5	6	7	8	9	10
	100%	0	1	1	2	3	4	6	7	8	9	10	11
Three-year mean peak of 11 great black-backed gulls in the offshore site in the breeding season (April to Aug)													
Regional population in breeding season = 96 pairs (Mitchell *et al.,* 2004)													

For the remaining displaced birds that survived, there could potentially be a detrimental impact on their breeding success, as a result of having to travel further on each trip to forage elsewhere. However, given the very low numbers of great black-backed gulls recorded in the offshore site in the breeding season (Figure 4.68, Figure 4.69 and Figure 4.70), the very restricted foraging range for this species (<10 km) (Roos *et al.,* 2010), and evidence from other wind farms indicating a lack of displacement effect, there will be no significant displacement impact on breeding birds.

Non-breeding period

Assuming 10% of all great black-backed gulls were displaced from the offshore site during the non-breeding season, this would affect an estimated two birds (Table 4.89), increasing to 22 birds for the offshore site plus 2 km buffer (Table 4.91). However, given that great black-backed gulls are not tied to a colony in the non-breeding season, and are therefore free to forage further afield, any additional mortality arising from displacement from the offshore site is likely to be minimal. It is concluded that any displaced great black-backed gulls would move to alternative foraging areas over the winter months.

Table 4.89 Estimated number of great black-backed gulls predicted to be at risk of mortality following displacement from offshore site in the non-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	0	0	0	1	1	1	1	1	1	2	2
	20%	0	0	0	1	1	1	2	2	3	3	3	4
	30%	0	0	1	1	2	2	3	3	4	4	5	5
	40%	0	0	1	1	2	3	4	4	5	6	6	7
	50%	0	0	1	2	3	4	5	5	6	7	8	9
	60%	0	1	1	2	3	4	5	6	8	9	10	11
	70%	0	1	1	3	4	5	6	8	9	10	11	13
	80%	0	1	1	3	4	6	7	9	10	12	13	14
	90%	0	1	2	3	5	6	8	10	11	13	15	16
	100%	0	1	2	4	5	7	9	11	13	14	16	18
Three-year mean peak of 18 great black-backed gulls in the offshore site in the non-breeding season (Sept to Mar)													
Regional population in the non-breeding season = 21,600 birds (Skov *et al*., 1995)													

Table 4.90 Estimated number of great black-backed gulls predicted to be at risk of mortality following displacement from offshore site plus 1 km buffer in the non-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	1	1	3	4	5	7	8	9	11	12	13
	20%	1	1	3	5	8	11	13	16	18	21	24	26
	30%	1	2	4	8	12	16	20	24	28	32	36	40
	40%	1	3	5	11	16	21	26	32	37	42	48	53
	50%	1	3	7	13	20	26	33	40	46	53	59	66
	60%	2	4	8	16	24	32	40	48	55	63	71	79
	70%	2	5	9	18	28	37	46	55	65	74	83	92
	80%	2	5	11	21	32	42	53	63	74	84	95	106
	90%	2	6	12	24	36	48	59	71	83	95	107	119
	100%	3	7	13	26	40	53	66	79	92	106	119	132
Three-year mean peak of 132 great black-backed gulls in the offshore site in the non-breeding season (Sept to Mar)													
Regional population in the non-breeding season = 21,600 birds (Skov *et al*., 1995)													

Table 4.91 Estimated number of great black-backed gulls predicted to be at risk of mortality following displacement from offshore site plus 2 km buffer in the non-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	1	2	4	7	9	11	13	16	18	20	22
	20%	1	2	4	9	13	18	22	27	31	36	40	44
	30%	1	3	7	13	20	27	33	40	47	53	60	67
	40%	2	4	9	18	27	36	44	53	62	71	80	89
	50%	2	6	11	22	33	44	56	67	78	89	100	111
	60%	3	7	13	27	40	53	67	80	93	107	120	133
	70%	3	8	16	31	47	62	78	93	109	124	140	155
	80%	4	9	18	36	53	71	89	107	124	142	160	178
	90%	4	10	20	40	60	80	100	120	140	160	180	200
	100%	4	11	22	44	67	89	111	133	155	178	200	222
Three-year mean peak of 222 great black-backed gulls in the offshore site in the non-breeding season (Sept to Mar)													
Regional population in the non-breeding season = 21,600 birds (Skov *et al*., 1995)													

Based on the very low estimated numbers of great black-backed gulls in the offshore site in the breeding and non-breeding seasons, and evidence from wind farm studies indicating a low level of displacement, the regional great black-backed gull population in the breeding and non-breeding seasons 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 population throughout the year.

This impact is categorised as having 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 great black-backed gull population in the breeding and non-breeding seasons 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.

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 disturbance by vessels during the construction, operation and decommissioning phases on the regional populations of h great black-backed gulls in the breeding and non-breeding periods is ***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

CRM estimated the number of potential great black-backed gull collisions per season for four wind turbine designs scenarios (Table 3.6). The minimum height for the turbine blades above the sea surface for all four options is 27.5 m at mean sea level (MSL).

There is no specific SNH guidance on avoidance rates for seabirds, and therefore their default value of 98.0% is presented in this assessment.

Using a 98% avoidance rate, the number of predicted great black-backed gull collisions in the breeding period for all four design options was one bird (Table 4.92). This equates to approximately 1.0% of the regional population (96 pairs) (Mitchell *et al,* 2004) in the breeding period.

Table 4.92 Number of estimated great black-backed gull collisions using avoidance rate of 98% for four wind farm options

	Option 1 (90x5MW)	Option 2 (75x6MW)	Option 3 (73x6.15MW)	Option 4 (73x6.15MW)
Collisions in breeding season (April to August), all ages	1	1	1	1
Collisions in non-breeding period (September to March), all ages	20	17	14	14
Total collisions per year, all ages	21	18	16	16

In the non-breeding period (September to March) and using a 98% avoidance rate, the predicted number of great black-backed gull collisions for the worst case design option (Option 1) was 20 birds (Table 4.92). This equates to approximately 0.09% of the regional population in the non-breeding period (21,600 birds) (Skov *et al.,* 1995).

Lowest estimated collisions in the non-breeding period were for Options 3 and 4 (73 x 6.15 MW turbines). Using a 98% avoidance rate, the number of predicted great black-backed gull collisions in the non-breeding period was 14 birds (Table 4.92). This equates to approximately 0.06% of the regional population in the non-breeding period (21,600 birds) (Skov *et al.,* 1995).

It is concluded that for the most adverse design (Option 1: 90 x 5 MW turbines), collision mortality for great black-backed gull in the breeding period is an effect of low magnitude, that is temporally long-term and reversible. In the non-breeding period collision mortality is an effect of negligible magnitude, that is temporally long-term and reversible.

It is further concluded that for the least adverse designs (Options 3 and 4: 73 x 6.15MW turbines), collision mortality for great black-backed gull in the breeding period is an effect of low magnitude, that is temporally long-term and reversible. In the non-breeding period collision mortality is an effect of negligible magnitude, that is temporally long-term and reversible.

It is concluded that for the wind farm designs examined here, the effects of collision mortality on great black-backed gulls from the regional population in the breeding and non-breeding periods are ***not significant*** under the Electricity Regulations.

Decommissioning Phase

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.

Summary of combined effects

The adverse impacts on the great black-backed 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 98.0%, it is judged that the combined magnitude of the three effects is negligible (Table 4.93 and Table 4.94). 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 great black-backed gull in the breeding and non-breeding seasons is ***not significant*** under the EIA Regulations.

Table 4.93 Summary of effects on the regional population of great 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 (98.0% A R)	Low	Long term	Moderate	***Not significant***
All effects combined	Negligible	Long term	Low - Moderate	***Not significant***

Table 4.94 Summary of effects on the regional population of great black-backed 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 (98.0% A R)	Negligible	Long term	Moderate	***Not significant***
All effects combined	Negligible	Long term	Low - Moderate	***Not significant***

4.4.17.5 Cumulative Impact Assessment

Displacement

No significant impacts were predicted to arise from displacement caused by Neart na Gaoithe for great black-backed gulls in the breeding or non-breeding periods. This was based on the numbers of birds recorded within the offshore site, and evidence from other wind farms, indicating likely low levels of displacement for this species. It is therefore not plausible that Neart na Gaoithe could contribute to a significant cumulative impact on the regional population of great black-backed gulls in the breeding or non-breeding periods. As a result, no further cumulative impact assessment for displacement was undertaken for this species.

Collision mortality

The potential cumulative collision risk to great black-backed gulls from Neart na Gaoithe and other plans or projects is higher than for Neart na Gaoithe alone.

Projects identified during consultation and the undertaking of the EIA for which there is a potential for a cumulative collision risk impact are:

- Inch Cape offshore wind farm;
- Seagreen Project Alpha offshore wind farm; and
- Seagreen Project Bravo offshore wind farm.

Data on the number of predicted collisions arising from the Seagreen Projects Alpha and Bravo offshore wind farms are presented within the applicant's Environmental Statement (Seagreen 2012) and HRA (Seagreen 2013). The proposed Seagreen Round 3 Zone development consists of a three phase programme of six separate developments. Projects Alpha and Bravo are the first developments and the only Seagreen projects for which applications have been made.

No application has been made for proposed Inch Cape offshore wind farm. Predicted number of collisions arising from the Inch Cape offshore wind farm have been obtained from an unpublished annual ornithological report based on one year's data (ICOL, 2012).

The predicted cumulative impacts on herring gulls from Neart na Gaoithe and Seagreen projects Alpha and Bravo in the breeding and non-breeding periods are presented in Table 4.95.

Using a 98% avoidance behavioural rate, a total of 23 great black-backed gulls are predicted to collide with Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms during the breeding period.

Table 4.95 Predicted number of cumulative great black-backed gulls (adult & immature) mortality with Seagreen Projects Alpha and Bravo in the breeding and non-breeding periods

Wind Farm	Breeding	Non-breeding
Neart na Gaoithe [1]		
98% avoidance	1	20
99% avoidance	1	10
Seagreen (Project Alpha) [2]		
98% avoidance	5	141
99% avoidance	3	70
Seagreen (Project Bravo) [2]		
98% avoidance	17	104
99% avoidance	9	52
Total		
98% avoidance	23	265
99% avoidance	13	132

1 Based on 3 years data. 2 Based on 2 years data

Not all birds recorded were adults and therefore a proportion of immature birds will be present during the breeding season. Of those that were aged during the breeding period, 57.1% of great black-backed gulls recorded at Neart na Gaoithe were immature birds. This was also applied to the numbers at Project Alpha and Project Bravo, as there was no details of age given in the ES (Seagreen 2012).

Using a 98% avoidance behavioural rate, and accounting for the proportion of non-breeding immature birds recorded during the breeding period, the estimated total number of adult great black-backed gulls at risk of collision during the breeding period at Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms is 10 birds. This equates to approximately 5.2% of the regional population (96 pairs) (Mitchell *et al,* 2004) in the breeding period.

Using a 98% avoidance behavioural rate, the total number of great black-backed gulls at risk of collision during the non-breeding period at Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms is 142 birds. This equates to approximately 0.7% of the regional population in the non-breeding period (21,600 birds) (Skov *et al.,* 1995).

Using a 98% avoidance behavioural rate, a total of one great black-backed gull is predicted to collide with Neart na Gaoithe and Inch Cape offshore wind farms during the breeding period (Table 4.96). This equates to 0.5% of the regional population (96 pairs) (Mitchell *et al,* 2004) in the breeding period.

The number of great black-backed gull collisions in the non-breeding period for Inch Cape was 328 collisions,using 98% avoidance, and worst case turbine design (Seagreen 2012).

Table 4.96 Predicted number of cumulative great black-backed gulls (adult & immature) mortality with Inch Cape in the breeding and non-breeding periods

Wind Farm	Breeding	Non-breeding
Neart na Gaoithe [1]		
98% avoidance	1	20
99% avoidance	1	10
Inch Cape [2]		
98% avoidance	0	328
99% avoidance	0	164
Total [4]		
98% avoidance	1	348
99% avoidance	1	174

1 Based on 3 years data. 2 Based on 1 years data

Assuming a 98% avoidance behavioural rate, the total number of great black-backed gulls at risk of collision during the non-breeding period at Neart na Gaoithe and Inch Cape offshore wind farms is estimated to be 348 birds. This equates to approximately 1.6% of the regional population in the non-breeding period (21,600 birds) (Skov *et al.,* 1995).

However, there is uncertainty in the results relating to the proposed Inch Cape offshore wind farm which are based on one year's data and a very much worse-case turbine design giving, potentially a maximum level of predicted impacts.

Using a 98% avoidance behavioural rate, a total of 23 great black-backed gulls are predicted to collide with Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms during the breeding period (Table 4.97).

Table 4.97 Predicted number of cumulative great black-backed gull (adult & immature) mortality with Seagreen Projects Alpha and Bravo and Inch Cape in the breeding and non-breeding periods

Wind Farm	Breeding	Non-breeding
Neart na Gaoithe [1]		
98% avoidance	1	20
99% avoidance	1	10
Seagreen (Project Alpha) [2]		
98% avoidance	5	141
99% avoidance	3	70
Seagreen (Project Bravo) [2]		
98% avoidance	17	104
99% avoidance	9	52
Inch Cape [3]		
98% avoidance	0	328
99% avoidance	0	164
Total		
98% avoidance	23	593
99% avoidance	13	296

1 Based on 3 years data. 2 Based on 2 years data 3 Based on 1 years data

Accounting for the proportion of non-breeding immature birds recorded during the breeding period the total number of adult great black-backed gulls at risk of collision during the breeding period at Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms is 10 birds. This equates to approximately 5.2% of the regional population (96 pairs) (Mitchell *et al,* 2004) in the breeding period.

It is concluded that the cumulative collision mortality for the regional great black-backed gull population in the breeding period is an effect of low magnitude, that is temporally long-term and reversible.

Using a 98% avoidance behavioural rate, the total number of great black-backed gulls at risk of collision during the non-breeding period at Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms is estimated to be 593 birds. This equates to approximately 2.7% of the regional population in the non-breeding period (21,600 birds) (Skov *et al.,* 1995).

However, there is uncertainty in the results relating to the proposed Inch Cape offshore wind farm which are based on one year's data and a very much worse-case turbine design giving, potentially a maximum level of predicted impacts. It is therefore considered likely that the cumulative number of collisions would be lower than this estimate.

Based on this, it is concluded that the cumulative collision mortality for the regional great black-backed gull population in the non-breeding period is an effect of low magnitude, that is temporally long-term and reversible.

It is concluded that for the four projects examined here, the effects of cumulative collision mortality on great black-backed gulls from the regional population in the breeding and non-breeding periods are ***not significant*** under the Electricity Regulations.

4.4.17.6 Mitigation measures

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

4.4.18 Black-legged kittiwake *Rissa tridactyla*

4.4.18.1 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).

4.4.18.2 Offshore site and 8 km buffer area

Kittiwake was the fourth most frequently encountered seabird on surveys in the Neart na Gaoithe study area in Year 1, with 3,955 birds recorded, 4,123 birds recorded in Year 2 and

4,300 birds in Year 3 (raw numbers, all sea states). The majority of birds were recorded in the buffer area (Table 4.2).

During the Year 1 breeding period (April to August), the mean estimated number of kittiwakes in the offshore site was 45 birds, with a peak of 83 birds in August (Table 4.98). In the same period of Year 2, the mean estimated number of kittiwakes in the offshore site was 408 birds, with a peak of 1,451 birds in July. In Year 3, the mean estimated number of kittiwakes during the breeding period was 913 birds, with a peak of 3,783 birds in August.

In the Year 1 post-breeding season (September and October), the mean estimated number of kittiwakes in the offshore site was 2,018 birds, with a peak of 2,211 birds in September (Table 4.98). In the same period of Year 2, the mean estimated number of kittiwakes in the offshore site was 54 birds, with a peak of 88 birds in October. In Year 3, the mean estimated number of kittiwakes during the post-breeding period was 28, with a peak of 34 birds in October.

In the Year 1 non-breeding season (November to March), the mean estimated number of kittiwakes in the offshore site was 15 birds, with a peak of 38 birds in November (Table 4.98). In the same period of Year 2, the mean estimated number of kittiwakes in the offshore site was 216 birds, with a peak of 837 birds in December, although there was no November survey in Year 2. In Year 3, the mean estimated number of kittiwakes during the non-breeding period was 80 birds, with a peak of 146 birds in November, although there was no December survey in Year 3.

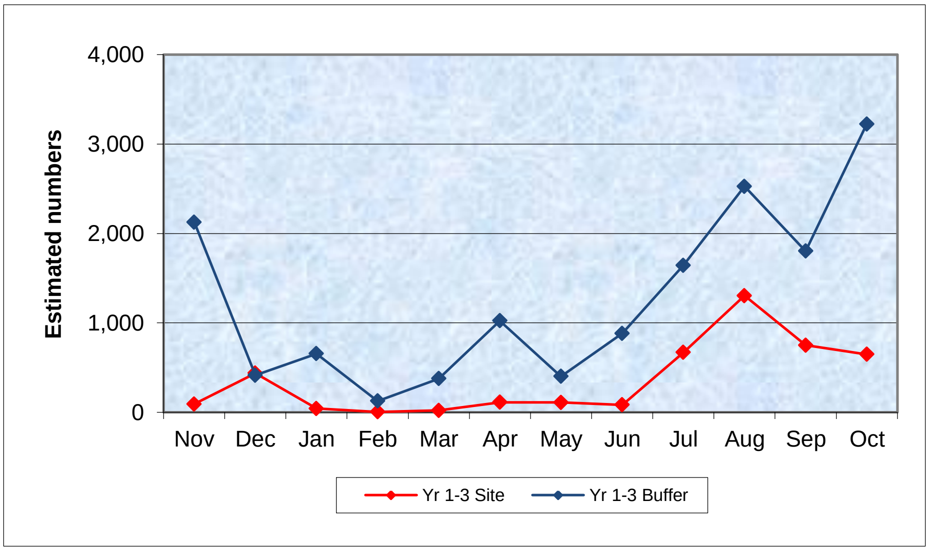


Figure 4.71 Mean monthly estimated numbers of kittiwakes in the Neart na Gaoithe Development & buffer areas in Years 1 to 3 (Three-year mean)

Mean estimated numbers of kittiwakes in the offshore site were very low between January and June, based on the three-year monthly mean (**Figure 4.71**). Mean estimated numbers increased in July and August as adults and juveniles left the breeding colonies and moved out to sea. Mean estimated numbers then decreased again in September and October. In the buffer area, mean estimated numbers were generally higher than in the offshore site, and showed a similar pattern, although highest numbers were recorded in October.

Table 4.98 Estimated numbers of kittiwakes in the offshore site (and 1, 2 & 8 km buffer) in Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 8 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	18	5	67	20	38	45	45	740
Yr1 Dec	22	6	87	14	36	43	72	388
Yr1 Jan	0	0	0	0	0	0	0	0
Yr1 Feb	0	0	0	0	0	0	0	119
Yr1 Mar	0	0	0	0	0	0	7	40
Yr1 Apr	0	0	0	7	7	20	20	2,044
Yr1 May	0	0	0	41	41	61	68	168
Yr1 Jun	35	17	71	20	55	55	151	505
Yr1 Jul	0	0	0	41	41	41	48	196
Yr1 Aug	77	17	351	7	83	407	620	741
Yr1 Sep	2,048	948	4,424	163	2,211	2,513	3,405	6,006
Yr1 Oct	1,688	995	2,864	136	1,824	2,302	4,440	7,322
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	101	43	239	736	837	882	936	1,314
Yr2 Jan	0	0	0	7	7	14	20	1,352
Yr2 Feb	0	0	0	14	14	71	71	135
Yr2 Mar	0	0	0	7	7	33	59	820
Yr2 Apr	256	116	567	48	304	369	444	1,079
Yr2 May	132	49	356	81	213	319	361	691
Yr2 Jun	0	0	0	27	27	47	68	1,427
Yr2 Jul	1,228	485	3,113	223	1,451	1,641	1,708	4,727
Yr2 Aug	18	7	52	27	46	62	149	545
Yr2 Sep	0	0	0	20	20	20	387	716
Yr2 Oct	0	0	0	88	88	115	165	4,161
Yr3 Nov	0	0	0	146	146	191	440	3,697
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	76	34	171	41	117	135	185	735
Yr3 Feb	0	0	0	0	0	14	14	140
Yr3 Mar	0	0	0	55	55	76	90	329
Yr3 Apr	10	2	41	14	24	38	72	281
Yr3 May	0	0	0	76	76	103	117	677
Yr3 Jun	137	71	267	28	165	228	341	962
Yr3 Jul	480	216	1,066	34	515	952	980	2,014
Yr3 Aug	3,763	1,795	7,890	20	3,783	3,903	4,165	10,208
Yr3 Sep	0	0	0	21	21	42	672	938
Yr3 Oct	0	0	0	34	34	34	48	135

Between November and March of Year 1, low densities of kittiwakes were recorded sporadically in the offshore site (Figure 4.72). In the buffer area, kittiwakes were scattered at low to moderate densities, with highest densities in the north and south of the buffer area.

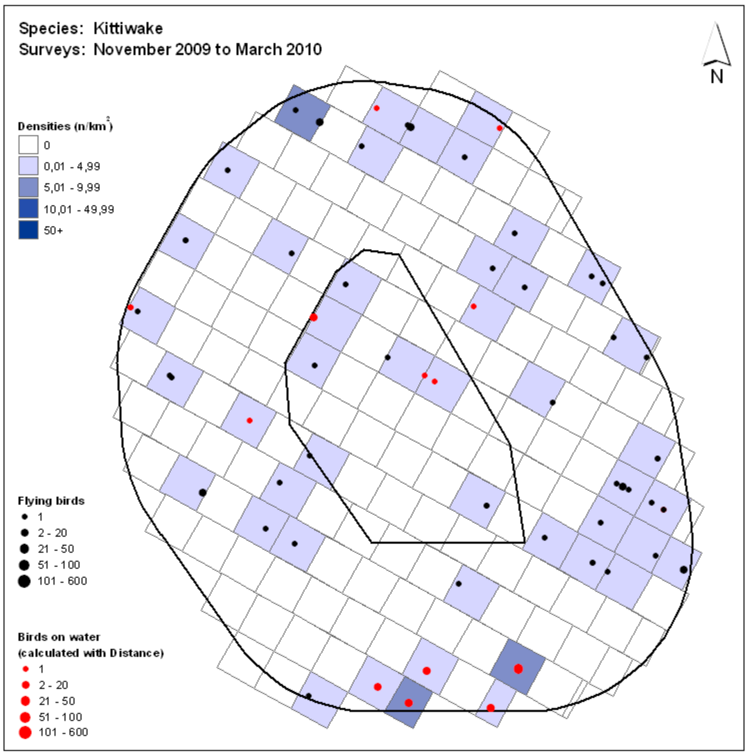


Figure 4.72 Kittiwake density between November and March, Year 1

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 4.73). 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.

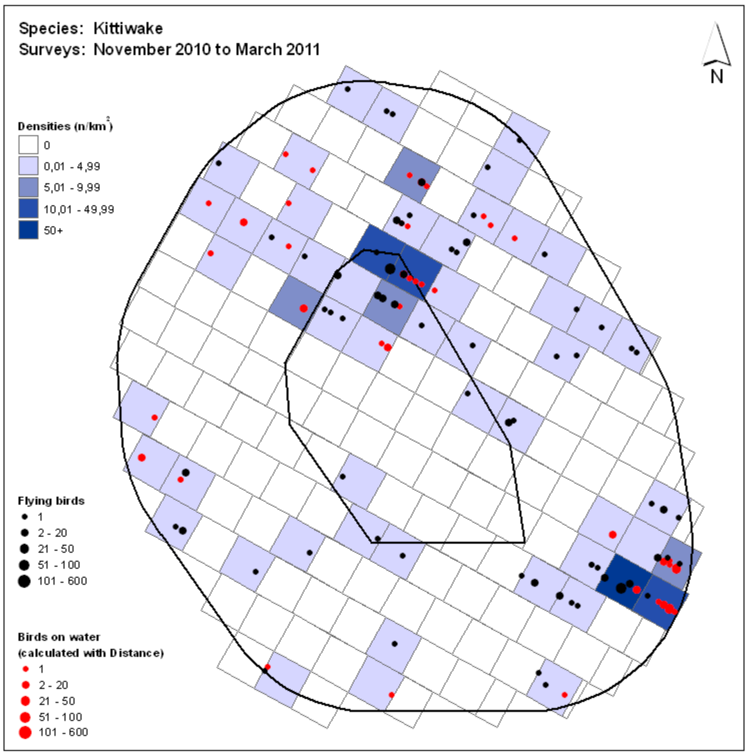


Figure 4.73 Kittiwake density between November and March, Year 2

In the Year 3 non-breeding period, kittiwakes were more widespread in the offshore site, although densities were mostly low (Figure 4.74). In the buffer area, kittiwakes were again more widespread at mostly low densities, with highest densities in the south-west of the buffer area.

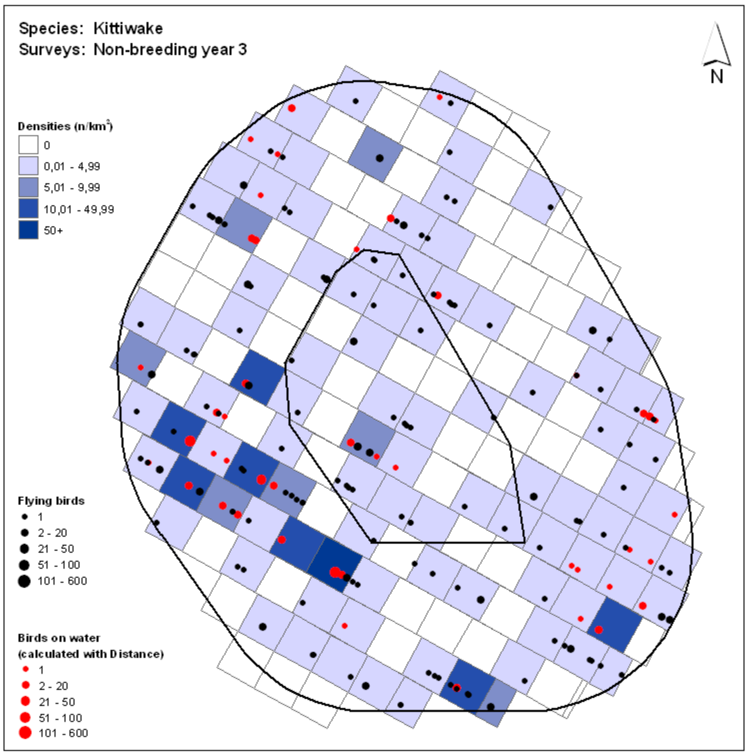


Figure 4.74 Kittiwake density between November and March, Year 3

Between April and August of Year 1, kittiwakes were more widespread across the offshore site and the buffer area, at mostly low to moderate densities (Figure 4.75). Highest densities were recorded in the south of the offshore site and south-east of the buffer area at this time.

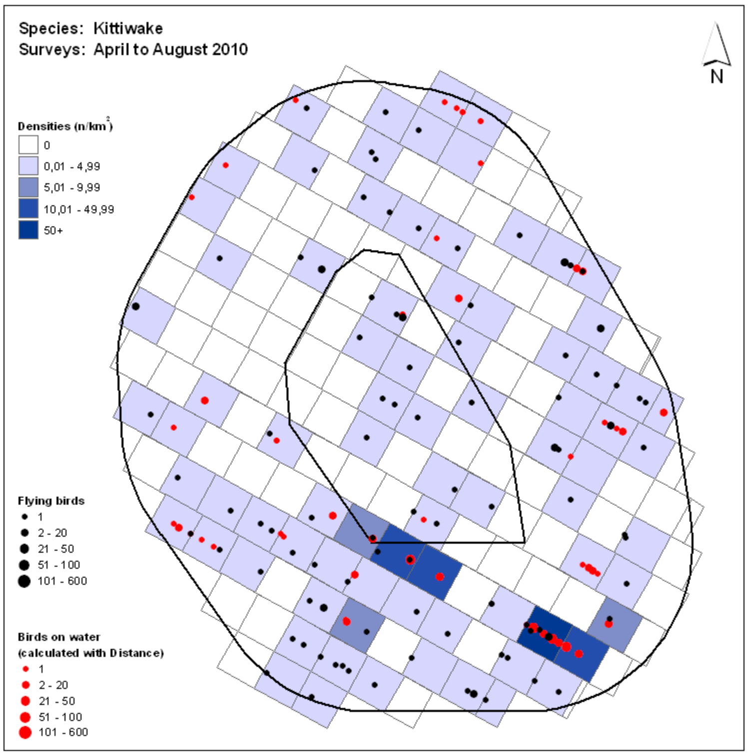


Figure 4.75 Kittiwake density between April and August, Year 1

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 4.76).

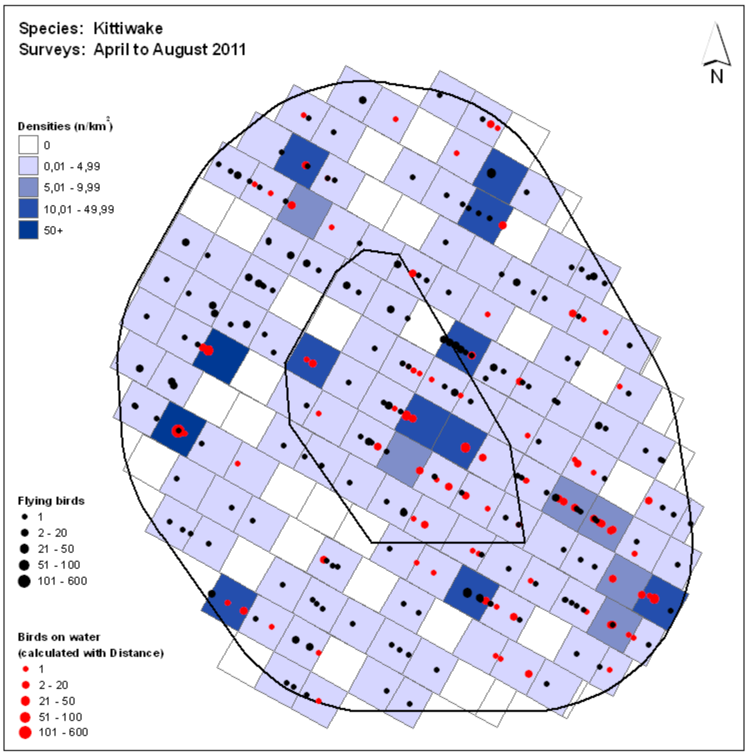


Figure 4.76 Kittiwake density between April and August, Year 2

In the Year 3 breeding season, kittiwakes were less widespread in the offshore site than in Year 2 (Figure 4.77). Birds were widespread across the buffer area at this time, with high densities mainly in the south-east of the buffer area at this time.

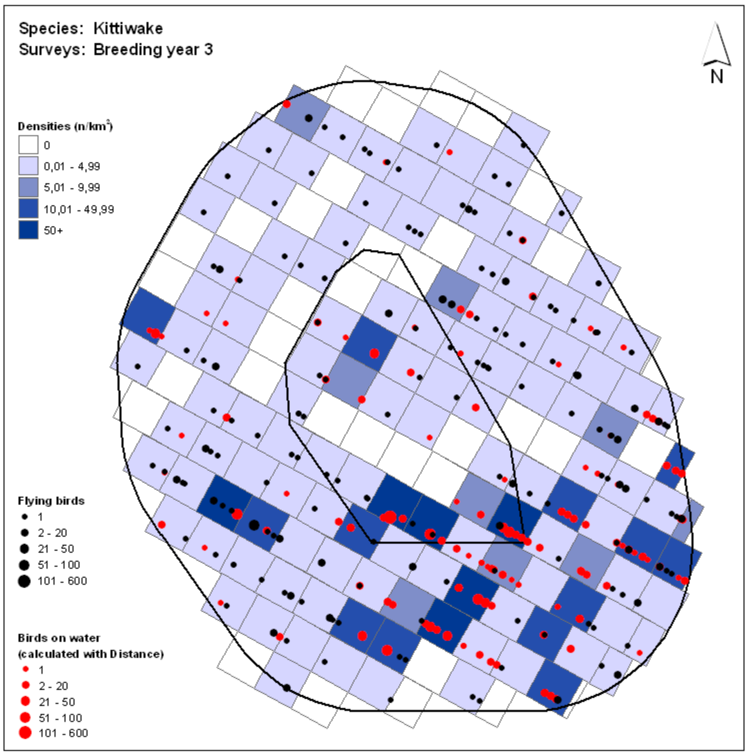


Figure 4.77 Kittiwake density between April and August, Year 3

In the Year 1 post-breeding period, moderate to high densities of kittiwakes were recorded in the offshore site, although distribution was patchy (Figure 4.78). In the buffer area, birds were widespread at mostly low densities at this time, with some higher density concentrations also recorded.

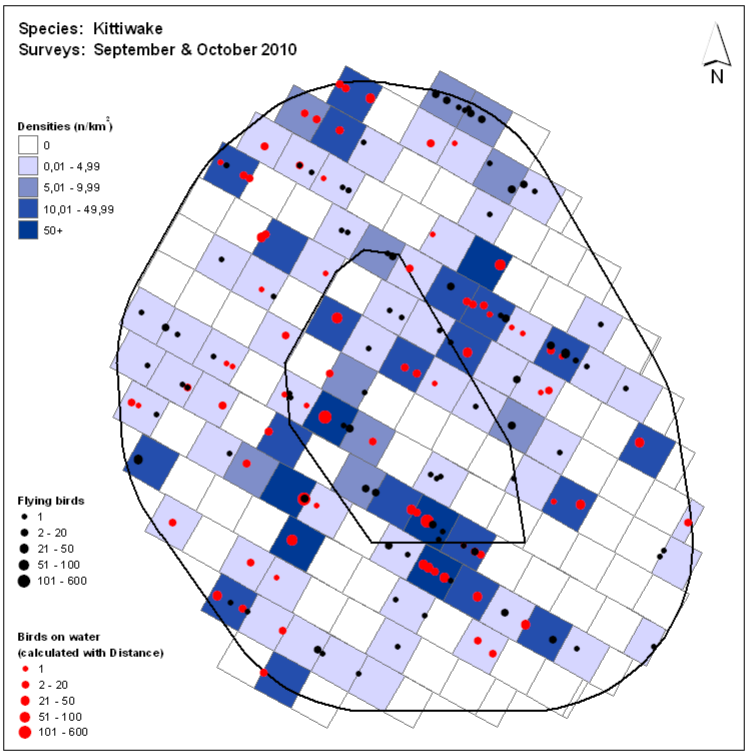


Figure 4.78 Kittiwake density between September and October, Year 1

In the Year 2 post-breeding period, fewer kittiwakes were recorded in the offshore site, with only low densities recorded, mainly in the south of the site (Figure 4.79). 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.

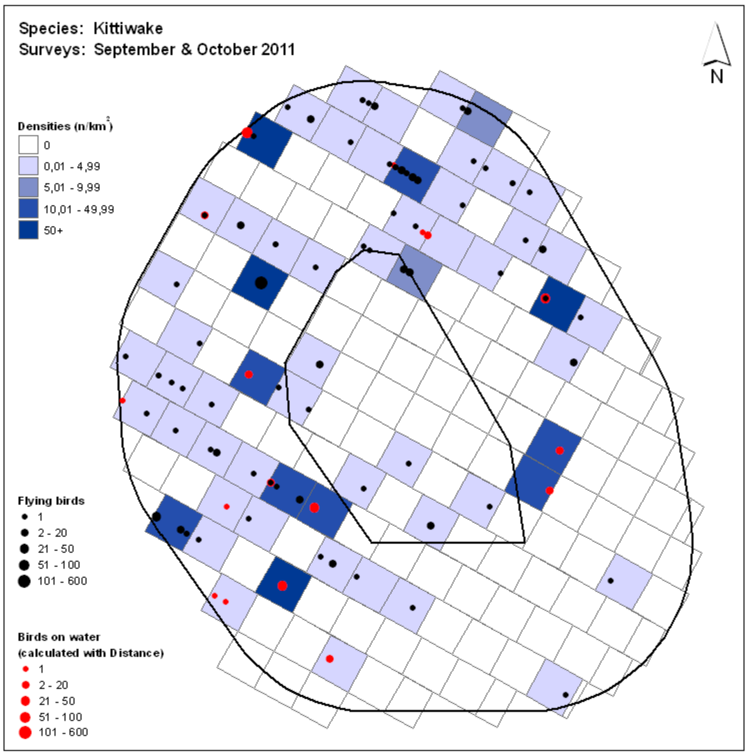


Figure 4.79 Kittiwake density between September and October, Year 2

In the Year 3 post-breeding period, few kittiwakes were again recorded in the offshore site, with only low densities recorded, mainly in the east of the site (Figure 4.80). Kittiwakes were scattered across the buffer area at mostly low densities at this time.

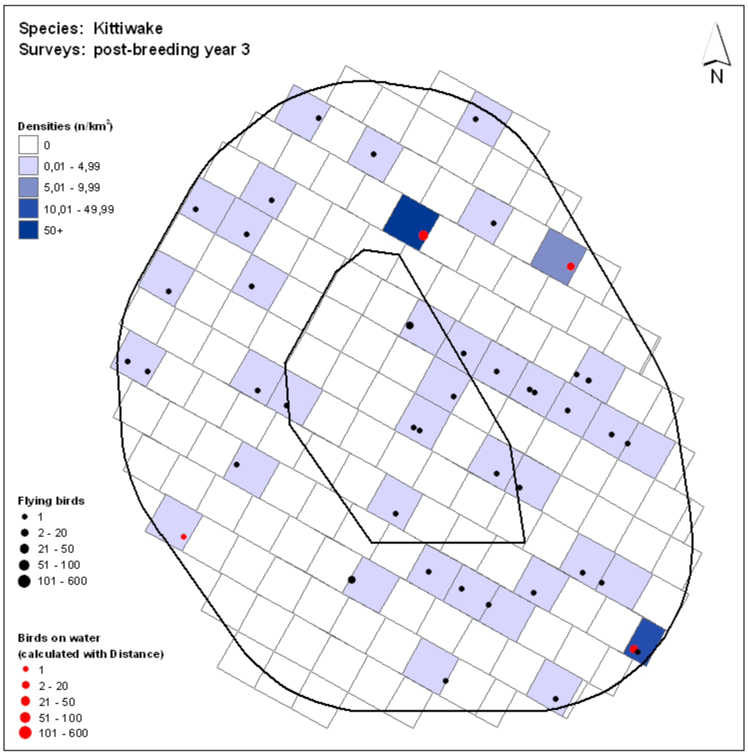


Figure 4.80 Kittiwake density between September and October, Year 3

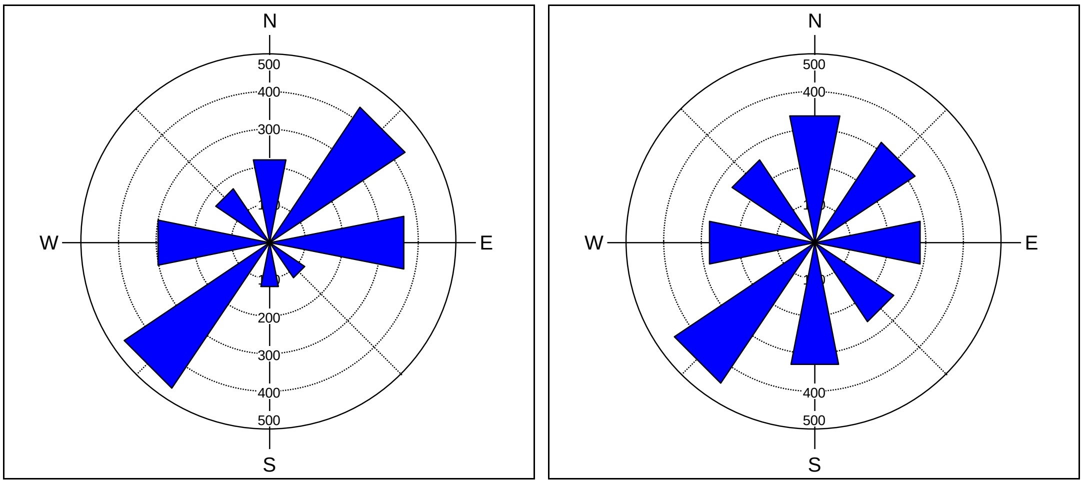
A total of 6,945 kittiwakes were recorded in flight, with the majority of birds (95.2%) recorded flying below 27.5 m in height (Table 4.3). A total of 333 birds (4.8%) were recorded flying above 27.5 m, i.e. within the rotor swept zone, at estimated heights of between 30 m and 70 m.

Foraging behaviour was recorded for 2,241 kittiwakes in the offshore site and 8 km buffer area on baseline surveys, with six types of foraging behaviour recorded, and unspecified feeding behaviour recorded for a further 50 birds (Table 4.99). The majority of all foraging birds were recorded actively searching (27.2%) and dipping (57.4%).

Table 4.99 Kittiwake foraging behaviour in the offshore site and 8 km buffer area in Years 1 to 3

Behaviour	Number of birds
Actively searching	822
Deep plunging	5
Dipping	1,112
Holding fish	2
Surface pecking	299
Scavenging at fishing vessel	1
Feeding method unspecified	50
Total	**2,291**

Flight direction was recorded for 2,194 kittiwakes in the breeding season (April to August), and 2,529 kittiwakes in the post-breeding and non-breeding seasons combined (September to March) (Figure 4.81). In the breeding season, just over a fifth of all birds recorded were flying south-west (21.3%), with 19.7% of birds flying north-east. In the non-breeding season, 17.7% of birds were recorded flying south-west, with 13.6 % flying north, and 13.0% flying south. An additional 2,183 birds were recorded as circling (not shown).



April to August (n=2,194 birds) September to March (n=2,529 birds)

Numbers shown on figures are number of birds recorded

Figure 4.81 Flight direction of kittiwakes in the offshore site and 8 km buffer area in Years 1 to 3

A total of 6,338 kittiwakes were aged during baseline surveys in the offshore site and 8 km buffer area. In the breeding season (April to August) age was recorded for 3,536 kittiwakes, with 227 immature (non-breeding) birds (6.4%) and 3,309 adults (93.6%) aged on surveys (Table 4.100).

Table 4.100 Monthly breakdown of immature and adult kittiwakes in the offshore site and 8 km buffer area in Years 1 to 3 combined

Month	No of immature birds	Number of adult birds	Number of aged birds	Percentage of immature birds
January	91	234	325	28.0%
February	56	106	162	34.6%
March	76	252	328	23.2%
April	10	552	562	1.8%
May	17	676	693	2.5%
June	17	742	759	2.2%
July	46	1,096	1,142	4.0%
August	137	243	380	36.1%
September	128	359	487	26.3%
October	385	510	895	43.0%
November	210	216	426	49.3%
December	133	46	179	74.3%

4.4.18.3 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).

A recent JNCC statistical analysis of ESAS data investigating possible marine SPAs, identified waters to the south and east of the offshore site and 8 km buffer area as an important location for kittiwakes during the breeding season (Kober *et al.,* 2010).

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 4.101). These SPAs held 20.5% of the UK breeding population and 3.2% of the biogeographic population at the time of designation (JNCC, 2013). Since designation, the populations at these SPAs have decreased (SMP, 2013). 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 of this species (60.0 km) (Thaxter *et al.*, 2012). As the distance between the offshore site and Fowlsheugh SPA (62 km) is only slightly greater than the mean maximum foraging range, Fowlsheugh SPA was included as being within mean maximum foraging range for assessment purposes.

The distance to a further two SPAs (Buchan Ness to Collieston Coast 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 4.82.

Table 4.101 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 (pairs) [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*	*4,241*	*2012*
Forth Islands	**16**	**9,380**	**0.3**	**1.9**	**3,766**	**2012**
Fowlsheugh	**62**	**34,870**	**1.1**	**7.1**	**9,337**	**2012**
St Abb's Head to Fast Castle	**31**	**19,600**	**0.6**	**4.0**	**5,409 [3]**	**2012**
Total	**-**	**100,538**	**3.2**	**20.5**	**36,886**	**-**

Sources: 1 JNCC (2013) – SPA online species accounts. 2 SMP (2013) – 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).
3. 2012 count for St Abb's Head NNR (4,314 pairs) (SMP 2013) plus corrected estimate for rest of SPA colony – see text for full explanation

The full extent of the St Abb's Head to Fast Castle SPA has not been surveyed since 2000, for Seabird 2000 (R Mavor pers. Comm.). However, breeding seabirds at St Abb's Head NNR, the major component of the SPA, are counted every year. To estimate the size of the remaining SPA population, the percentage decline in kittiwake numbers at St Abb's Head NNR between 2000 (11,077 pairs) and 2012 (4,314 pairs) was calculated (-61.1% decline).

This was applied to the population of the remainder of the SPA for 2000 (2,814 pairs) to estimate the likely size of the population for this area in 2012 (1,095 pairs). The two 2012 values were then added to get a total estimate for the whole SPA for 2012.

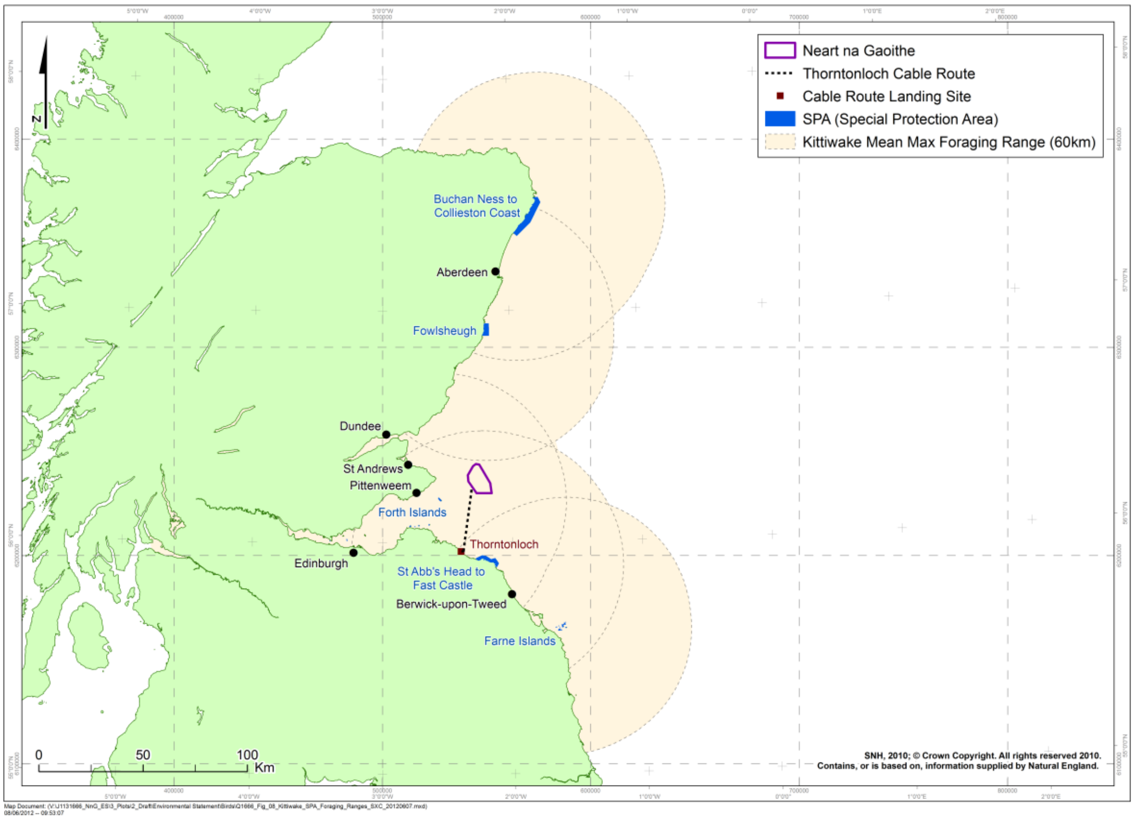


Figure 4.82 Kittiwake mean maximum foraging range from breeding SPAs in relation to the Development

4.4.18.4 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).

Populations

Kittiwakes breeding in eastern Scotland are currently experiencing a prolonged period of population decline (SMP, 2013). The decline in breeding numbers means that published figures on population size for kittiwake (e.g., Mitchell *et al*., 2004) no longer accurately reflect the current population size in eastern Scotland. To prevent this causing the assessment of effects against the regional population to be biased low, the regional total for Peterhead to Blyth derived from Seabird 2000 results (91,586 pairs) (Mitchell *et al.,* 2004) was adjusted, based on the mean change in numbers at regularly counted colonies (Table 4.102). Assuming that the sub-set of colonies that have been recently counted is representative, the regional decline since the Seabird 2000 counts amounts to -30.2%. On this basis, the regional breeding population between Peterhead and Blyth was estimated to be 63,927 pairs (127,854 birds).

Table 4.102 Recent counts & Seabird 2000 counts at main colonies for breeding kittiwakes between Peterhead and Blyth

Colony	Distance to site (km)	Seabird 2000 count	Recent count (birds at colony)	Year	Percentage change since Seabird 2000
Buchan Ness to Collieston Coast	113	14,091 [1]	14,133 [1]	2007	+0.3%
Fowlsheugh	62	18,800 [1]	9,337 [1]	2012	-46.1%
Isle of May	16	3,639 [2]	2,645 [1]	2012	-27.3%
St Abb's Head NNR	33	11,077 [1]	4,314 [1]	2012	-61.1%
Farne Islands	72	5,096 [2]	4,241 [1]	2012	-16.8%
				Mean percentage change	**-30.2%**

Sources: 1 SMP (2013) – Seabird Monitoring Programme Online Database. 2 Mitchell *et al.*, 2004.

The SPA breeding population within mean maximum foraging range of the offshore site was estimated to be 18,512 pairs (37,024 birds), based on most recent available counts (Table 4.101).

The size of the regional kittiwake population in the post-breeding-period (September and October) was assumed to be the same as the regional breeding population, i.e., 127,854 birds (63,927 pairs).

The size of the regional kittiwake population in the non-breeding-period was assumed to be 102,500 birds. This was derived by summing the October to March period estimates for localities 8, and 9 and half the estimates for localities 10 and 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 (Wernham *et al.,* 2002).

Nature conservation importance

The nature conservation importance of kittiwakes using the offshore site is rated as high during the breeding season, 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. The three-year mean peak estimated number of kittiwakes present in the offshore site in the breeding period (April to August) (2,128 birds) is 5.7% of the SPA breeding population within mean maximum foraging range of the offshore site (37,024 birds).

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)

One year of post-construction monitoring at Robin Rigg Offshore Wind Farm Data suggested that kittiwake numbers decreased during the construction phase and to an extent in the first year of operation (compared to pre-construction values) but this pattern was no more pronounced within the turbine area than compared to the entire site and could be regarded as reflecting inter- annual variation (Walls *et al.,* 2013).

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.

Construction Phase

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.

Operational Phase

Displacement

In order to assess the potential impacts from displacement, the approach presented in the interim guidance note on displacement issued by the JNCC and NE has been followed (JNCC & NE 2012).

Peak estimated numbers of kittiwakes in the offshore site in the breeding season (April to August), post-breeding season (September and October) and non-breeding season (December to March) for Years 1 to 3 were averaged to get the three-year mean peak per season. Where peak numbers occurred in different months within the same season across different years, the peak month was used. This was repeated for a 1 km and 2 km buffer around the offshore site (Table 4.103).

Table 4.103 Seasonal three-year mean peak estimated numbers of kittiwakes in the offshore site (plus 1 & 2 km buffer)

Year	Offshore site		
	Breeding	Post-breeding	Non-breeding
Year 1	83	2,211	38
Year 2	1,451	88	837
Year 3	3,783	34	146
3-year mean peak	1,772	778	340
Year	Offshore site + 1 km		
	Breeding	Post-breeding	Non-breeding
Year 1	407	2,513	45
Year 2	1,641	115	882
Year 3	3,903	42	191
3-year mean peak	1,984	890	373
Year	Offshore site + 2 km		
	Breeding	Post-breeding	Non-breeding
Year 1	620	4,440	72
Year 2	1,708	387	936
Year 3	4,165	672	440
3-year mean peak	2,164	1,833	483

Guidance recommends presenting a range of potential displacement and mortality rates and wherever possible selecting a suitable impact based on empirical evidence. Where, there is little evidence to support the assessment a precautionary approach should be taken.

Likely impacts of displacement

Results from bird monitoring at operational wind farms indicate that kittiwakes are not likely to be displaced from offshore wind farms. On this basis, a precautionary level of 10% displacement was assumed for this assessment.

Based on evidence from existing wind farms, it was assumed that there will be 10% displacement of kittiwakes from the offshore site in the breeding, post-breeding and non-breeding seasons.

Breeding period

Assuming 10% of all kittiwakes were displaced from the offshore site during the breeding season, this would affect an estimated 177 birds (Table 4.104), increasing to 216 birds including the 2 km buffer (Table 4.106). However, this estimate includes non-breeding immature birds, as well as breeding adults. During the breeding period (April to August), 6.4% of aged kittiwakes were immature birds (Table 4.100). This percentage was applied to the estimated numbers of displaced kittiwakes in the breeding season to estimate the number of adults potentially displaced (11 birds in offshore site, increasing to 14 birds including the 2 km buffer), as recommended in the draft guidance note on displacement (JNCC & NE, 2012).

Table 4.104 Estimated number of kittiwakes predicted to be at risk of mortality following displacement from offshore site in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	4	9	**18**	35	53	71	89	106	124	142	159	177
	20%	7	18	35	71	106	142	177	213	248	284	319	354
	30%	11	27	53	106	159	213	266	319	372	425	478	532
	40%	14	35	71	142	213	284	354	425	496	567	638	709
	50%	18	44	89	177	266	354	443	532	620	709	797	886
	60%	21	53	106	213	319	425	532	638	744	851	957	1,063
	70%	25	62	124	248	372	496	620	744	868	992	1,116	1,240
	80%	28	71	142	284	425	567	709	851	992	1,134	1,276	1,418
	90%	32	80	159	319	478	638	797	957	1,116	1,276	1,435	1,595
	100%	35	89	177	354	532	709	886	1,063	1,240	1,418	1,595	1,772
Three-year mean peak of 1,772 kittiwakes in the offshore site in the breeding season (April to Aug)													
SPA Population within mean max foraging range (61.1 km) = 18,512 pairs (SMP 2013)													

Table 4.105 Estimated number of kittiwakes predicted to be at risk of mortality following displacement from offshore site plus 1 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	4	10	**20**	40	60	79	99	119	139	159	179	198
	20%	8	20	40	79	119	159	198	238	278	317	357	397
	30%	12	30	60	119	179	238	298	357	417	476	536	595
	40%	16	40	79	159	238	317	397	476	556	635	714	794
	50%	20	50	99	198	298	397	496	595	694	794	893	992
	60%	24	60	119	238	357	476	595	714	833	952	1,071	1,190
	70%	28	69	139	278	417	556	694	833	972	1,111	1,250	1,389
	80%	32	79	159	317	476	635	794	952	1,111	1,270	1,428	1,587
	90%	36	89	179	357	536	714	893	1,071	1,250	1,428	1,607	1,786
	100%	40	99	198	397	595	794	992	1,190	1,389	1,587	1,786	1,984
Three-year mean peak of 1,984 kittiwakes in the offshore site in the breeding season (April to Aug)													
SPA Population within mean max foraging range (61.1 km) = 18,512 pairs (SMP 2013)													

Table 4.106 Estimated number of kittiwakes predicted to be at risk of mortality following displacement from offshore site plus 2 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	4	11	**22**	43	65	87	108	130	151	173	195	216
	20%	9	22	43	87	130	173	216	260	303	346	390	433
	30%	13	32	65	130	195	260	325	390	454	519	584	649
	40%	17	43	87	173	260	346	433	519	606	692	779	866
	50%	22	54	108	216	325	433	541	649	757	866	974	1,082
	60%	26	65	130	260	390	519	649	779	909	1,039	1,169	1,298
	70%	30	76	151	303	454	606	757	909	1,060	1,212	1,363	1,515
	80%	35	87	173	346	519	692	866	1,039	1,212	1,385	1,558	1,731
	90%	39	97	195	390	584	779	974	1,169	1,363	1,558	1,753	1,948
	100%	43	108	216	433	649	866	1,082	1,298	1,515	1,731	1,948	2,164
Three-year mean peak of 2,164 kittiwakes in the offshore site in the breeding season (April to Aug)													
SPA Population within mean max foraging range (61.1 km) = 18,512 pairs (SMP 2013)													

Assuming 6.4% of all birds were non-breeding immature birds, it was estimated that if 10% of all adult kittiwakes were displaced from the offshore site during the breeding season, this would affect an estimated 166 birds, increasing to 202 birds including the 2 km buffer. For the purposes of this assessment, it was assumed that 10% of all adult kittiwakes displaced

from the offshore site and 2 km buffer during the breeding season (up to 22 birds) would die as a result. This corresponds to 0.06% of the regional SPA population within mean maximum foraging range (18,512 pairs) (SMP 2013). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

For the remaining 180 displaced adults that survived, there could potentially be a detrimental impact on their breeding success, as a result of having to travel further on each trip to forage elsewhere. If it was assumed that this affected 180 pairs out of the regional SPA population within mean maximum foraging range (18,512 pairs), then an estimated 1.0% of the regional SPA population within mean maximum foraging range could potentially be affected.

However, it is likely that kittiwakes can compensate for a moderate amount of displacement by choosing to forage elsewhere. Recent satellite tagging studies have been conducted to examine kittiwake foraging movements in the vicinity of the Firth of Forth. GPS tracking was conducted on the Isle of May in June 2010, (36 birds, 91 foraging trips) (Daunt *et al.,* 2011a). Similar tracking studies were repeated in May and June 2011 at Fowlsheugh (35 birds, 93 trips) and St Abb's Head (25 birds, 70 trips) (Daunt *et al.,* 2011b). Data were split into non-flight (foraging and resting), relevant to displacement effects, and flight, relevant to collision risk.

Mean maximum foraging range from the Isle of May colony was 42 km, with a maximum foraging range of 150 km recorded (Daunt *et al.,* 2011a). Foraging trips from Fowlsheugh were concentrated in a north-easterly to south-easterly direction, with a mean maximum foraging range of 35 km, and a maximum foraging range of 141 km recorded (excluding one outlier of 415 km). Foraging range from St Abb's Head was similar (mean maximum range of 32 km; maximum 108 km), but overall distribution was more focussed, in a south-easterly direction (Daunt *et al.,* 2011b).

The offshore site is within the mean maximum foraging ranges recorded for the Isle of May and St Abb's Head by these tagging studies, but is outside the mean maximum foraging range recorded at Fowlsheugh.

Analysis of at-sea distributions of kittiwakes using kernel density estimations found that the offshore site did not overlap to any great extent with the core area used by foraging kittiwakes from the Isle of May (50% kernals), but was within the overall area used by tagged foraging kittiwakes in 2010 (90% kernels). The core area of use (50% kernels) was estimated to cover an area of 1,947 km^2, while the overall area of active use (90% kernels) was estimated at 3,993 km^2 (Daunt *et al.,* 2011a).

Tagging studies at Fowlsheugh and St Abb's Head indicated no overlap between foraging ranges of tagged kittiwakes and the offshore site in 2011 (Daunt *et al.,* 2011b). Based on this, it is likely that birds recorded in the offshore site during May and June will be from the Isle of May colony.

The 2010 study also examined differences in foraging range between 2010 and 2001 for birds from the Isle of May, with a more extensive foraging range recorded in 2010. There was evidence of a negative relationship between range and breeding success, as kittiwakes are generally more sensitive to environmental change than guillemots. The relationship between environmental change, foraging range and breeding success is complex and is likely to depend on the overall abundance of prey species, as well as prey distribution in relation to the colony. (Daunt *et al.,* 2011a).

The main conclusion that can be drawn from these studies is that kittiwakes are clearly capable of travelling and foraging over considerable distances during the breeding season (Daunt *et al.,* 2011a). It is therefore considered that should kittiwakes be partially displaced from the offshore site following construction of the wind farm, any impact on breeding success of these displaced birds is not likely to be significant.

Post-breeding period

For the post-breeding season, all birds (immature and adults) were considered. Assuming 10% of all kittiwakes were displaced from the offshore site during the post-breeding season, this would affect an estimated 78 birds (Table 4.107), increasing to 183 birds including the 2 km buffer (

Table **4.109**). However, given that kittiwakes are not tied to a colony in the non-breeding season, and are therefore free to forage further afield, any additional mortality arising from displacement from the offshore site is likely to be minimal. For the purposes of this assessment, it was assumed that 2% of all kittiwakes displaced from the offshore site and 2 km buffer during the post-breeding season (up to four birds) would die as a result. Such a mortality rate would affect less than 0.01% of the regional population in the post-breeding period (127,854 birds) (Revised from Mitchell *et al,* 2004). It is concluded that the remaining 179 displaced kittiwakes would move to alternative foraging areas.

Table 4.107 Estimated number of kittiwakes predicted to be at risk of mortality following displacement from offshore site in the post-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	**2**	4	8	16	23	31	39	47	54	62	70	78
	20%	3	8	16	31	47	62	78	93	109	124	140	156
	30%	5	12	23	47	70	93	117	140	163	187	210	233
	40%	6	16	31	62	93	124	156	187	218	249	280	311
	50%	8	19	39	78	117	156	195	233	272	311	350	389
	60%	9	23	47	93	140	187	233	280	327	373	420	467
	70%	11	27	54	109	163	218	272	327	381	436	490	545
	80%	12	31	62	124	187	249	311	373	436	498	560	622
	90%	14	35	70	140	210	280	350	420	490	560	630	700
	100%	16	39	78	156	233	311	389	467	545	622	700	778
Three-year mean peak of 778 kittiwakes in the offshore site in the post-breeding season (Sept & Oct)													
Regional population between Peterhead and Blyth = 127,854 birds (Mitchell *et al.,* 2004, corrected for decline)													

Table 4.108 Estimated number of kittiwakes predicted to be at risk of mortality following displacement from offshore site plus 1 km buffer in the post-breeding season

		Mortality (%)											
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	**2**	4	9	18	27	36	45	53	62	71	80	89
	20%	4	9	18	36	53	71	89	107	125	142	160	178
	30%	5	13	27	53	80	107	134	160	187	214	240	267
	40%	7	18	36	71	107	142	178	214	249	285	320	356
	50%	9	22	45	89	134	178	223	267	312	356	401	445
	60%	11	27	53	107	160	214	267	320	374	427	481	534
	70%	12	31	62	125	187	249	312	374	436	498	561	623
	80%	14	36	71	142	214	285	356	427	498	570	641	712
	90%	16	40	80	160	240	320	401	481	561	641	721	801
	100%	18	45	89	178	267	356	445	534	623	712	801	890
Three-year mean peak of 890 kittiwakes in the offshore site in the post-breeding season (Sept & Oct)													
Regional population between Peterhead and Blyth = 127,854 birds (Mitchell *et al.,* 2004, corrected for decline)													

Table 4.109 Estimated number of kittiwakes predicted to be at risk of mortality following displacement from offshore site plus 2 km buffer in the post-breeding season

		Mortality (%)											
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	**4**	9	18	37	55	73	92	110	128	147	165	183
	20%	7	18	37	73	110	147	183	220	257	293	330	367
	30%	11	27	55	110	165	220	275	330	385	440	495	550
	40%	15	37	73	147	220	293	367	440	513	587	660	733
	50%	18	46	92	183	275	367	458	550	642	733	825	917
	60%	22	55	110	220	330	440	550	660	770	880	990	1,100
	70%	26	64	128	257	385	513	642	770	898	1,026	1,155	1,283
	80%	29	73	147	293	440	587	733	880	1,026	1,173	1,320	1,466
	90%	33	82	165	330	495	660	825	990	1,155	1,320	1,485	1,650
	100%	37	92	183	367	550	733	917	1,100	1,283	1,466	1,650	1,833
Three-year mean peak of 1,183 kittiwakes in the offshore site in the post-breeding season (Sept & Oct)													
Regional population between Peterhead and Blyth = 127,854 birds (Mitchell *et al.,* 2004, corrected for decline)													

For the non-breeding season, all birds (immature and adults) were considered. Assuming 10% of all kittiwakes were displaced from the offshore site during the non-breeding season, this would affect an estimated 34 birds (Table 4.110), increasing to 48 birds including the 2 km buffer (

Table **4.112**). However, given that kittiwakes are not tied to a colony in the non-breeding season, and are therefore free to forage further afield, any additional mortality arising from displacement from the offshore site is likely to be minimal. For the purposes of this assessment, it was assumed that 2% of all kittiwakes displaced from the offshore site and 2 km buffer during the non-breeding season (up to one bird) would die as a result. Such a mortality rate would affect less than 0.01% of the regional population in the non-breeding period (102,500 birds) (Skov *et al.,* 1995). It is concluded that the remaining displaced kittiwakes would move to alternative foraging areas.

Table 4.110 Estimated number of kittiwakes predicted to be at risk of mortality following displacement from offshore site in the non-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	2	3	7	10	14	17	20	24	27	31	34
	20%	1	3	7	14	20	27	34	41	48	54	61	68
	30%	2	5	10	20	31	41	51	61	71	82	92	102
	40%	3	7	14	27	41	54	68	82	95	109	122	136
	50%	3	9	17	34	51	68	85	102	119	136	153	170
	60%	4	10	20	41	61	82	102	122	143	163	184	204
	70%	5	12	24	48	71	95	119	143	167	190	214	238
	80%	5	14	27	54	82	109	136	163	190	218	245	272
	90%	6	15	31	61	92	122	153	184	214	245	275	306
	100%	7	17	34	68	102	136	170	204	238	272	306	340
Three-year mean peak of 340 kittiwakes in the offshore site in the non-breeding season (Nov to March)													
Regional population in non-breeding period = 102,500 birds (Skov *et al.,* 1995)													

Table 4.111 Estimated number of kittiwakes predicted to be at risk of mortality following displacement from offshore site in the non-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	2	4	7	11	15	19	22	26	30	34	37
	20%	1	4	7	15	22	30	37	45	52	60	67	75
	30%	2	6	11	22	34	45	56	67	78	90	101	112
	40%	3	7	15	30	45	60	75	90	104	119	134	149
	50%	4	9	19	37	56	75	93	112	131	149	168	187
	60%	4	11	22	45	67	90	112	134	157	179	201	224
	70%	5	13	26	52	78	104	131	157	183	209	235	261
	80%	6	15	30	60	90	119	149	179	209	239	269	298
	90%	7	17	34	67	101	134	168	201	235	269	302	336
	100%	7	19	37	75	112	149	187	224	261	298	336	373
Three-year mean peak of 373 kittiwakes in the offshore site in the non-breeding season (Nov to March))													
Regional population in non-breeding period = 102,500 birds (Skov *et al.,* 1995)													

Table 4.112 Estimated number of kittiwakes predicted to be at risk of mortality following displacement from offshore site in the non-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	2	5	10	14	19	24	29	34	39	43	48
	20%	2	5	10	19	29	39	48	58	68	77	87	97
	30%	3	7	14	29	43	58	72	87	101	116	130	145
	40%	4	10	19	39	58	77	97	116	135	155	174	193
	50%	5	12	24	48	72	97	121	145	169	193	217	242
	60%	6	14	29	58	87	116	145	174	203	232	261	290
	70%	7	17	34	68	101	135	169	203	237	270	304	338
	80%	8	19	39	77	116	155	193	232	270	309	348	386
	90%	9	22	43	87	130	174	217	261	304	348	391	435
	100%	10	24	48	97	145	193	242	290	338	386	435	483
Three-year mean peak of 483 kittiwakes in the offshore site in the non-breeding season (Nov to March)													
Regional population in non-breeding period = 102,500 birds (Skov *et al.,* 1995)													

Based on evidence from tagging studies summarised above (e.g. Daunt *et al.,* 2011a), the regional kittiwake population in the breeding, post-breeding and non-breeding periods 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 population throughout the year.

This impact is categorised as having 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 kittiwake population in the breeding, post-breeding and non-breeding seasons 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 3.7). 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 3.8). For birds breeding on Craigleith, the wind farm acting as a barrier would potentially block approximately 28% of the possible flight directions (Table 3.7). 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 3.8). 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 3.7). 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 3.8).

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

CRM estimated the number of potential kittiwake collisions per season for four wind turbine designs scenarios (Table 3.6). The minimum height for the turbine blades above the sea surface for all four options is 27.5 m at mean sea level (MSL).

There is no specific SNH guidance on avoidance rates for seabirds, and therefore their default value of 98.0% is presented in this assessment for comparison purposes

Highest estimated collisions in the breeding period were for Option 1 (90 x 5 MW turbines). Using a 98% avoidance rate, the number of predicted kittiwake collisions in the breeding period was 26 birds (Table 4.113).

Lowest estimated collisions in the breeding period were for Options 3 and 4 (73 x 6.15 MW turbines). Using a 98% avoidance rate, the number of predicted kittiwake collisions in the breeding period was 20 birds (Table 4.113).

Table 4.113 Number of estimated kittiwake collisions using avoidance rate of 98% for four wind farm options

	Option 1 (90x5MW)	Option 2 (75x6MW)	Option 3 (73x6.15MW)	Option 4 (73x6.15MW)
Collisions in breeding season (April to August), all ages	26	23	20	20
Collisions in breeding season (April to August), adults only	24	22	19	19
Collisions in post-breeding season (September & October), all ages	16	14	12	12
Collisions in non-breeding period (November to March), all ages	44	37	33	33
Total collisions per year, all ages	86	74	65	65

Baseline surveys between April and August recorded the age for a total of 3,536 kittiwakes, with 227 immature (non-breeding) birds (6.4%) and 3,309 adults (93.6%) (Table 4.113). The estimated number of collisions for the breeding period were reduced by 6.4% so that they represented the estimated numbers of adult kittiwakes involved with collisions.

Based on this ratio and using a 98% avoidance rate, the predicted number of adult kittiwake collisions in the breeding season for the worst case design option (Option 1) was 24 birds (Table 4.113). This equates to approximately 0.02% of the regional population (63,927 pairs) (Revised from Mitchell *et al,* 2004), or 0.06% of the regional SPA population (18,512 pairs) (SMP 2013) in the breeding period.

In the post-breeding period (September and October) and using a 98% avoidance rate, the predicted number of kittiwake collisions for the worst case design option (Option 1) was 16 birds (Table 4.113). This equates to approximately 0.01% of the regional population in the non-breeding period (127,854 birds) (Revised from Mitchell *et al,* 2004).

In the non-breeding period (November to March) and using a 98% avoidance rate, the predicted number of kittiwake collisions for the worst case design option (Option 1) was 44 birds (Table 4.113). This equates to approximately 0.04% of the regional population in the non-breeding period (102,500 birds) (Skov *et al.,* 1995).

It is concluded that for the most adverse design (Option 1: 90 x 5 MW turbines), collision mortality for kittiwake is an effect of negligible magnitude, that is temporally long-term and reversible.

It is further concluded that for the least adverse designs (Options 3 and 4: 73 x 6.15MW turbines), collision mortality for kittiwake is an effect of negligible magnitude, that is temporally long-term and reversible.

It is concluded that for the wind farm designs examined here, the effects of collision mortality on kittiwakes from the regional population in the breeding, post-breeding and non-breeding periods are ***not significant*** under the Electricity Regulations.

Decommissioning Phase

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.

Summary of combined effects

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 98%, it is judged that the combined magnitude of the three effects is negligible (Table 4.114). 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.

Table 4.114 Summary of effects on the regional breeding population of kittiwakes 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 using 98.0% A R	Low	Long term	Moderate	***Not significant***
All effects combined	Negligible	Long term	Low/Moderate	***Not significant***

Using an avoidance rate of 98.0%, it is judged that the combined magnitude of the three effects on the regional kittiwake population in the post-breeding and non-breeding periods is negligible (Table 4.115). It is concluded that the overall operational phase impact of the development on the regional kittiwake population in the post-breeding and non-breeding periods is ***not significant*** under the EIA Regulations.

Table 4.115 Summary of effects on the regional population of kittiwakes present in the post-breeding and non-breeding periods

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 (98.0% A R)	Negligible	Long term	Moderate	***Not significant***
All effects combined	Negligible	Long term	Low	***Not significant***

4.4.18.5 Cumulative Impact Assessment

Displacement

No significant impacts were predicted to arise from displacement caused by Neart na Gaoithe for kittiwakes in the breeding, post-breeding or non-breeding periods. This was based on the estimated numbers of birds within the offshore site, and evidence from other wind farms, indicating likely low levels of displacement for this species. It is therefore not plausible that Neart na Gaoithe could contribute to a significant cumulative impact on the regional population of kittiwakes in the breeding, post-breeding or non-breeding periods. As a result, no further cumulative impact assessment for displacement was undertaken for this species.

Collision mortality

The potential cumulative collision risk to kittiwakes from Neart na Gaoithe and other plans or projects is higher than for Neart na Gaoithe alone.

Projects identified during consultation and the undertaking of the EIA for which there is a potential for a cumulative collision risk impact are:

- Inch Cape offshore wind farm;
- Seagreen Project Alpha offshore wind farm; and

- Seagreen Project Bravo offshore wind farm.

Data on the number of predicted collisions arising from the Seagreen Projects Alpha and Bravo offshore wind farms are presented within the applicant's Environmental Statement (Seagreen 2012) and HRA (Seagreen 2013). The proposed Seagreen Round 3 Zone development consists of a three phase programme of six separate developments. Projects Alpha and Bravo are the first developments and the only Seagreen projects for which applications have been made.

No application has been made for proposed Inch Cape offshore wind farm. Unpublished predicted number of collisions arising from the Inch Cape offshore wind farm was obtained through communication with Inch Cape Offshore Limited (ICOL).

The predicted cumulative impacts on kittiwakes from Neart na Gaoithe and Seagreen projects Alpha and Bravo in the breeding and non-breeding periods are presented in Table 4.116.

Table 4.116 Predicted number of cumulative kittiwakes (adult & immature) mortality with Seagreen Projects Alpha and Bravo in the breeding, post-breeding and non-breeding periods

Wind Farm	Breeding	Post-breeding	Non-breeding
Neart na Gaoithe [1]			
98% avoidance	26	16	44
99% avoidance	13	8	22
Seagreen (Project Alpha) [2]			
98% avoidance	201	Not available	474
99% avoidance	101	Not available	237
Seagreen (Project Bravo) [2]			
98% avoidance	263	Not available	361
99% avoidance	132	Not available	181
Total			
98% avoidance	490	Not available	879
99% avoidance	246	Not available	440

1 Based on 3 years data. 2 Based on 2 years data

Using a 98% avoidance behavioural rate, a total of 490 kittiwakes are predicted to collide with Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms during the breeding period.

Not all birds recorded were adults and therefore a proportion of immature birds will not be from the SPA breeding population. Of those that were aged during the breeding period, 6.4% of kittiwakes recorded at Neart na Gaoithe were immature birds. Seagreen have assessed the number of adult kittiwakes predicted to be impacted by Projects Alpha (125 adults) and Bravo (167 adults), taking into account the proportion of immature birds and 98% avoidance (Seagreen 2013).

Using a 98% avoidance behavioural rate, and accounting for the proportion of non-breeding immature birds recorded during the breeding period, the total number of adult kittiwakes at risk of collision during the breeding period at Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms is 316 birds. This equates to approximately 0.2% of the regional population (63,927 pairs) (revised from Mitchell *et al,* 2004), or 0.8% of the regional SPA population (18,512 pairs) (SMP 2013) in the breeding period.

Using a 98% avoidance behavioural rate, a total of 879 kittiwakes are predicted to collide with Neart na Gaoithe and Seagreen Projects Alpha and Bravo offshore wind farms during the non-breeding period. This equates to approximately 0.8% of the regional population in the non-breeding period (102,500 birds) (Skov *et al.,* 1995).

The predicted cumulative impacts on kittiwakes from Neart na Gaoithe and Inch Cape in the breeding and non-breeding periods are presented in Table 4.117.

Table 4.117 Predicted number of cumulative kittiwakes (adult & immature) mortality with Inch Cape in the breeding, post-breeding and non-breeding periods

Wind Farm	Breeding	Post-breeding	Non-breeding
Neart na Gaoithe [1]			
98% avoidance	26	16	44
99% avoidance	13	8	22
Inch Cape [2]			
98% avoidance	18 [3]	Not available	Not available
99% avoidance	9 [3]	Not available	Not available
Total			
98% avoidance	44	16+	44+
99% avoidance	22	8+	22+

1 Based on 3 years data. 2 Based on 2 years data 3 Adults only estimate (ICOL, 2013)

Using a 98% avoidance behavioural rate, a total of 44 kittiwakes are predicted to collide with Neart na Gaoithe and Inch Cape offshore wind farms during the breeding period.

Of those that were aged during the breeding period at Neart na Gaoithe, 6.4% of were immature birds. Using a 98% avoidance behavioural rate, and accounting for the proportion of non-breeding immature birds recorded during the breeding period the total number of adult kittiwakes at risk of collision during the breeding period at Neart na Gaoithe and Inch Cape offshore wind farms is 42 birds. This equates to approximately 0.03% of the regional population (63,927 pairs) (revised from Mitchell *et al,* 2004), or 0.1% of the regional SPA population (18,512 pairs) (SMP 2013) in the breeding period.

Estimated numbers of kittiwake collisions for the post-breeding and non-breeding periods were not available for Inch Cape. It was therefore not possible to calculate the cumulative collision mortality for Neart na Gaoithe and Inch Cape for these periods.

The predicted cumulative impacts on kittiwakes from Neart na Gaoithe, Seagreen projects Alpha and Bravo and Inch Cape in the breeding and non-breeding periods are presented in Table 4.118.

Using a 98% avoidance behavioural rate, a total of 336 kittiwakes are predicted to collide with Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms during the breeding period.

Table 4.118 Predicted number of cumulative kittiwake (adult & immature) mortality with Seagreen Projects Alpha and Bravo and Inch Cape in the breeding, post-breeding and non-breeding periods

Wind Farm	Breeding	Post-breeding	Non-breeding
Neart na Gaoithe [1]			
98% avoidance	26	16	44
99% avoidance	13	8	22
Seagreen (Project Alpha) [2]			
98% avoidance	125 [3]	Not available	474
99% avoidance	63 [3]	Not available	237
Seagreen (Project Bravo) [2]			
98% avoidance	167 [3]	Not available	361
99% avoidance	84 [3]	Not available	181
Inch Cape [2]			
98% avoidance	18 [3]	Not available	Not available
99% avoidance	9 [3]	Not available	Not available
Total			
98% avoidance	336	Not available	879+
99% avoidance	169	Not available	440+

1 Based on 3 years data. 2 Based on 2 years data 3 Adults only estimate

Accounting for the proportion of non-breeding immature birds recorded during the breeding period the total number of adult kittiwakes at risk of collision during the breeding period at Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms is 334 adults. This equates to approximately 0.3% of the regional population (63,927 pairs) (revised from Mitchell *et al,* 2004), or 0.9% of the regional SPA population (18,512 pairs) (SMP 2013) in the breeding period.

It is concluded that the cumulative collision mortality for the regional SPA kittiwake population in the breeding period is an effect of negligible magnitude, that is temporally long-term and reversible.

Estimated numbers of kittiwake collisions for the post-breeding season were not available for Seagreen Projects Alpha or Bravo, or for Inch Cape. It was therefore not possible to calculate the cumulative collision mortality for this period. However, the predicted number of kittiwake collisions in the post-breeding period at Neart na Gaoithe were low (16 birds at 98% avoidance rate) (Table 4.118) and it is therefore not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for the regional population in the post-breeding period (127,854 birds) (revised from Mitchell *et al,* 2004). As a result, no further cumulative impact assessment for collision risk was undertaken for this period.

Estimated numbers of kittiwake collisions for the non-breeding season were not available for for Inch Cape. It was therefore not possible to calculate the cumulative collision mortality for this period for all four projects. However, using a 98% avoidance behavioural rate, and assuming that predicted collisions at Inch Cape were the same as Neart na Gaoithe (44 birds), an estimated total of 923 kittiwakes collisions were predicted for Neart na Gaoithe, Seagreen Projects Alpha and Bravo and Inch Cape offshore wind farms during the non-breeding period. This equates to approximately 0.9% of the regional population in the non-breeding period (102,500 birds) (Skov *et al.,* 1995).

It is concluded that the cumulative collision mortality for the regional kittiwake population in the non-breeding period is an effect of negligible magnitude, that is temporally long-term and reversible.

Overall, it is concluded that for the four projects examined here, the effects of cumulative collision mortality on kittiwakes from the regional population in the breeding, post-breeding and non-breeding periods are ***not significant*** under the Electricity Regulations.

4.4.18.6 Mitigation measures

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.

4.4.19 Common tern *Sterna hirundo*

4.4.19.1 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 offshore site and 8 km buffer area are Leith Docks in Edinburgh (818 pairs in 2010) and the Isle of May (20 pairs in 2012) (SMP, 2013). Common terns have a broad diet compared to other tern species, including sandeels, clupeid and gadoid fish (Mitchell *et al.,* 2004).

4.4.19.2 Offshore site and 8 km buffer area

Low numbers of common terns were recorded in the offshore site and 8 km buffer area in Year 1, with a total of 13 birds recorded between July and November, and highest numbers in September (10 birds) (Table 4.2) (raw numbers, all sea states). Numbers of common terns in Year 2 were slightly higher, with a total of 50 birds recorded, again mostly in September. In Year 3, one common tern was recorded in the buffer area in August (Figure 4.83).

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. No common terns were recorded in the offshore site in Year 3.

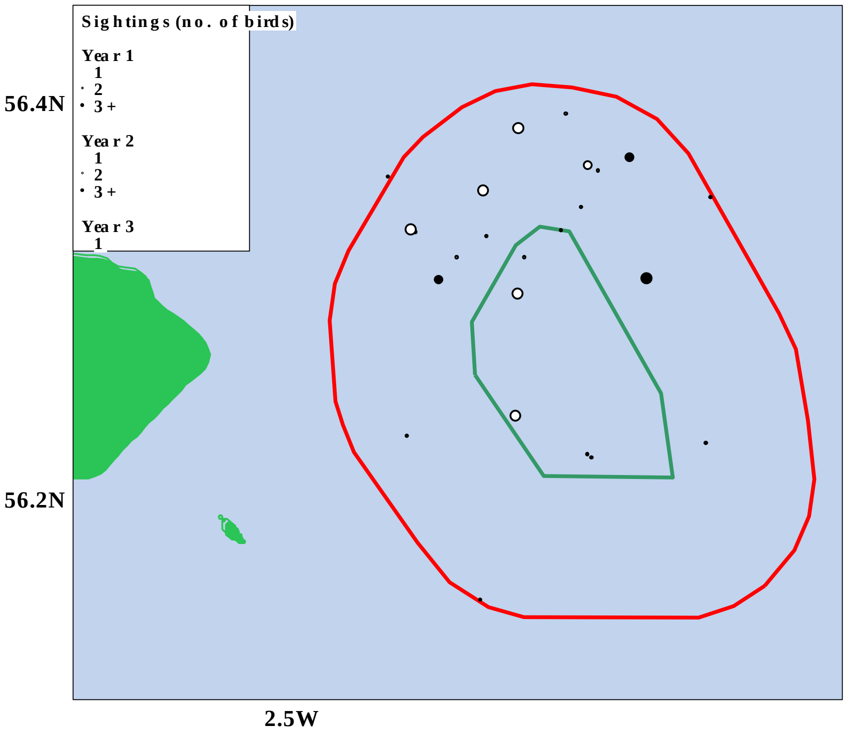


Figure 4.83 Common tern sightings between May and November in Years 1 to 3

A total of 36 common terns were recorded in flight on baseline surveys, with all birds flying below 17.5 m in height (Table 4.3).

4.4.19.3 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).

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 4.119). These SPAs held 21.6 % of the UK breeding population and 1.3 % of the biogeographic population at the time of designation (JNCC, 2013). 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 4.84.

Table 4.119 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	1,158	2012
Farne Islands	72	230	0.1	1.9	88	2012
Firth of Forth Islands	*16*	*800*	*0.4*	*6.5*	*20*	*2012*
Ythan Estuary, Sands of Forvie & Meikle Loch	110	265	0.1	2.2	4	2010
Leith Docks	62	789	0.3	5.0	818	2010
Total	**-**	**2,824**	**1.3**	**21.6**	**2,088**	**-**

Sources: 1 JNCC (2013) – SPA online species accounts. 2 SMP (2013) – 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).

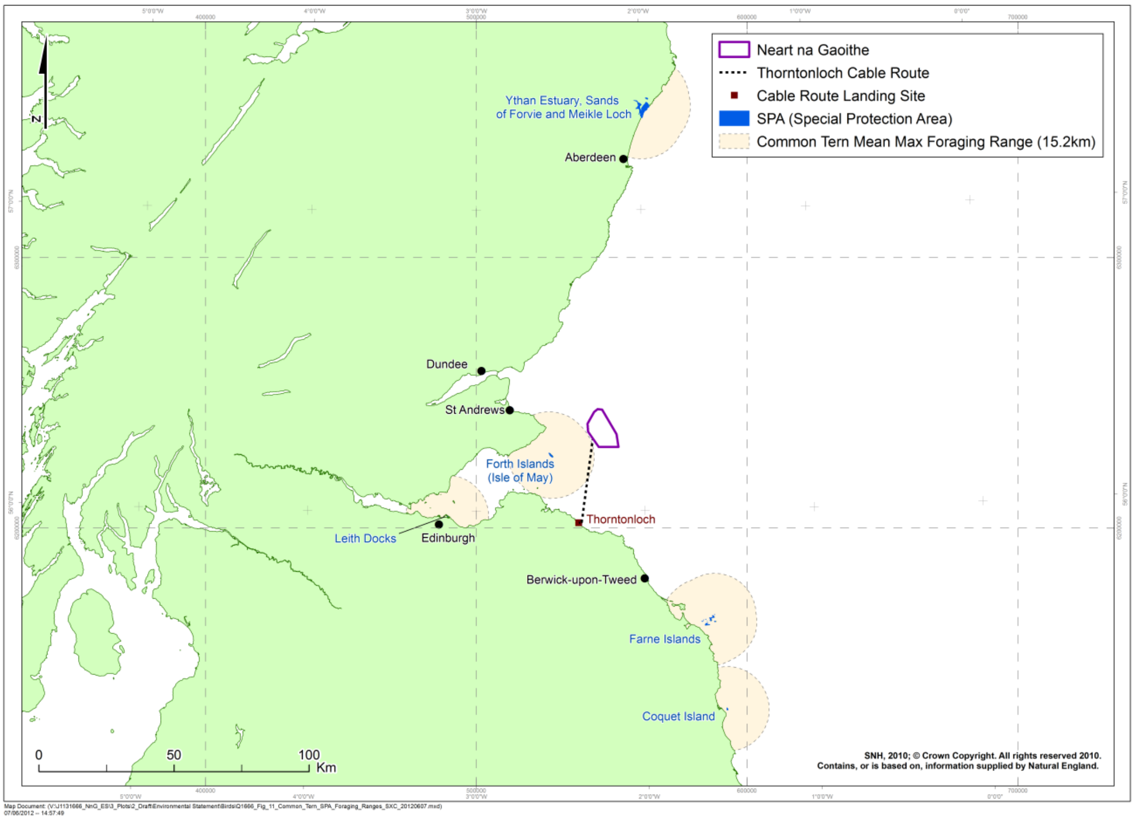


Figure 4.84 Common tern mean maximum foraging range from breeding SPAs in relation to the Development

4.4.19.4 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. No common terns were recorded in the offshore site in Year 3.

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 (818 pairs in 2010) and the Isle of May (20 pairs in 2012) (SMP, 2013). 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 offshore site 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 36 common terns in flight, with all birds flying below 17.5 m in height, i.e. well below rotor 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%. Further details are presented in Ornithology Appendix 2: Collision Rate Estimates of Seabirds at Neart na Gaoithe.

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 collisions per annum for a 98.0% avoidance rate. This corresponds to approximately 0.5% of the eastern Scotland and north-east England estimated population (8,126 adults).

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, 100% of 36 flying common terns recorded in baseline surveys were below 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 on common terns 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.

4.4.19.5 Cumulative Impact Assessment

There were no significant impacts predicted for the regional population of common terns in the breeding or non-breeding periods from the proposed Neart na Gaoithe development. The predicted effects of the development on the regional population of common terns in the autumn passage period arising from construction, operation and decommissioning are very close to no effect.

Based on these findings, it was considered that it is not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for this population, and therefore no further cumulative impact assessment was undertaken for this species.

4.4.19.6 Mitigation measures

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

4.4.20 Arctic tern *Sterna paradisaea*

4.4.20.1 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 offshore site and 8 km buffer area is the Isle of May, with 265 pairs in 2012 (SMP, 2013). 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).

4.4.20.2 Offshore site and 8 km buffer area

Arctic terns were only recorded in the offshore site between May and September on baseline surveys, although numbers varied between years. In Year 1, a total of 857 Arctic terns were recorded on surveys in the offshore site and 8 km buffer between May and August, with 205 birds recorded in the offshore site in August (raw numbers, all sea states) (Table 4.2). Fewer birds were recorded in Year 2, with 329 birds recorded between May and October, with a total of 37 Arctic terns recorded in the offshore site between May and September. A total of 549 Arctic terns were recorded between May and August of Year 3, with 90 birds recorded in the offshore site. Peak numbers were recorded in August. It is likely that these birds were passing through the offshore site and 8 km buffer area on southward migration at the end of the breeding season. The majority of aged birds were adults (95.7%).

During the Year 1 autumn migration period (August), the estimated number of Arctic terns in the offshore site was 1,114 birds (Table 4.120). In August of Year 2, the estimated number of Arctic terns in the offshore site was 55 birds. In August of Year 3, the estimated number of Arctic terns in the offshore site was 601 birds.

In the offshore site, the mean estimated number of Arctic terns peaked in August (590 birds, three-year mean), with a smaller peak in May (56 birds, three-year mean) (Figure 4.85). In the buffer area, the mean estimated number also peaked in August (1,625 birds, three-year mean). This exceeded the 1% nationally important threshold of 1,052 birds (Mitchell *et al.,* 2004).

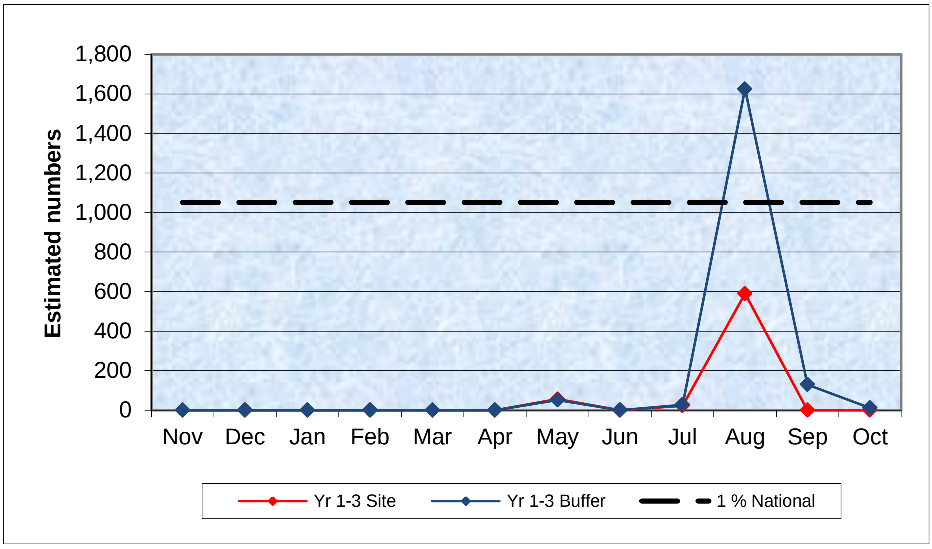


Figure 4.85 Mean monthly estimated numbers of Arctic terns in the offshore site & buffer area in Years 1 to 3 (Three-year mean)

Table 4.120 Estimated numbers of Arctic terns in the offshore site (plus 1 & 2 km buffer) in Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 8 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	0	0	0	0	0	0	0	0
Yr1 Dec	0	0	0	0	0	0	0	0
Yr1 Jan	0	0	0	0	0	0	0	0
Yr1 Feb	0	0	0	0	0	0	0	0
Yr1 Mar	0	0	0	0	0	0	0	0
Yr1 Apr	0	0	0	0	0	0	0	0
Yr1 May	0	0	0	0	0	0	0	0
Yr1 Jun	0	0	0	0	0	0	0	0
Yr1 Jul	0	0	0	0	0	0	0	0
Yr1 Aug	637	189	2,143	477	1,114	1,162	1,762	3,323
Yr1 Sep	0	0	0	0	0	0	0	0
Yr1 Oct	0	0	0	0	0	0	0	0
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	0	0	0	0	0	0	0	0
Yr2 Jan	0	0	0	0	0	0	0	0
Yr2 Feb	0	0	0	0	0	0	0	0
Yr2 Mar	0	0	0	0	0	0	0	0
Yr2 Apr	0	0	0	0	0	0	0	0
Yr2 May	0	0	0	168	168	168	168	323
Yr2 Jun	0	0	0	0	0	0	0	0
Yr2 Jul	0	0	0	0	0	0	0	0
Yr2 Aug	0	0	0	55	55	109	123	396
Yr2 Sep	0	0	0	0	0	0	57	388
Yr2 Oct	0	0	0	0	0	0	0	34
Yr3 Nov	0	0	0	0	0	0	0	0
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	0	0	0	0	0	0	0	0
Yr3 Feb	0	0	0	0	0	0	0	0
Yr3 Mar	0	0	0	0	0	0	0	0
Yr3 Apr	0	0	0	0	0	0	0	0
Yr3 May	0	0	0	0	0	0	0	0
Yr3 Jun	0	0	0	0	0	0	0	0
Yr3 Jul	69	12	401	0	69	69	69	151
Yr3 Aug	594	243	1,454	7	601	693	992	2,927
Yr3 Sep	0	0	0	0	0	0	0	0
Yr3 Oct	0	0	0	0	0	0	0	0

Arctic terns were scattered sporadically in the offshore site and buffer area at moderate to high densities in August of Year 1, with the majority of birds recorded outside the offshore site (Figure 4.86).

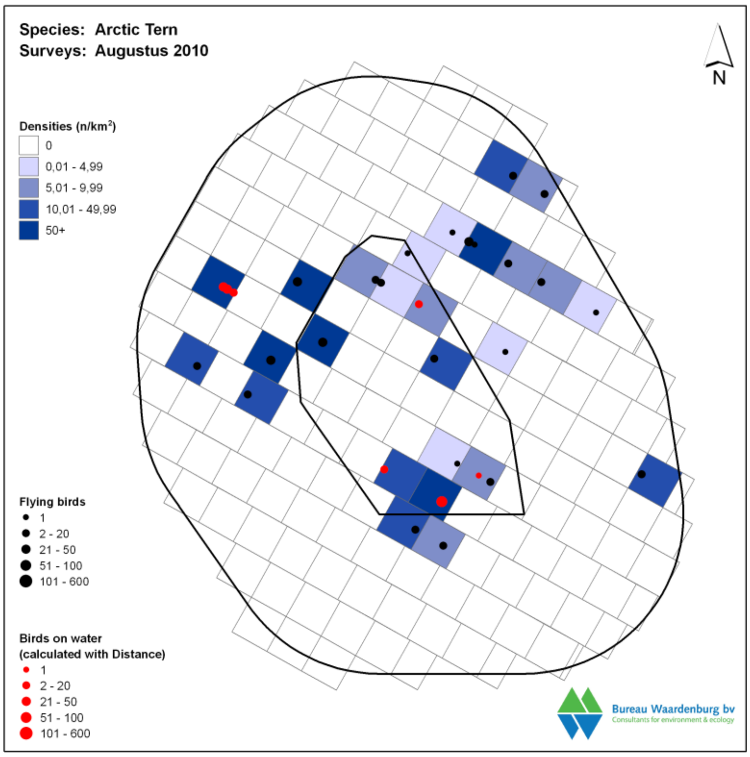


Figure 4.86 Arctic tern density in August, Year 1

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 4.87). Densities were generally low to moderate in August, with most birds recorded west of the offshore site.

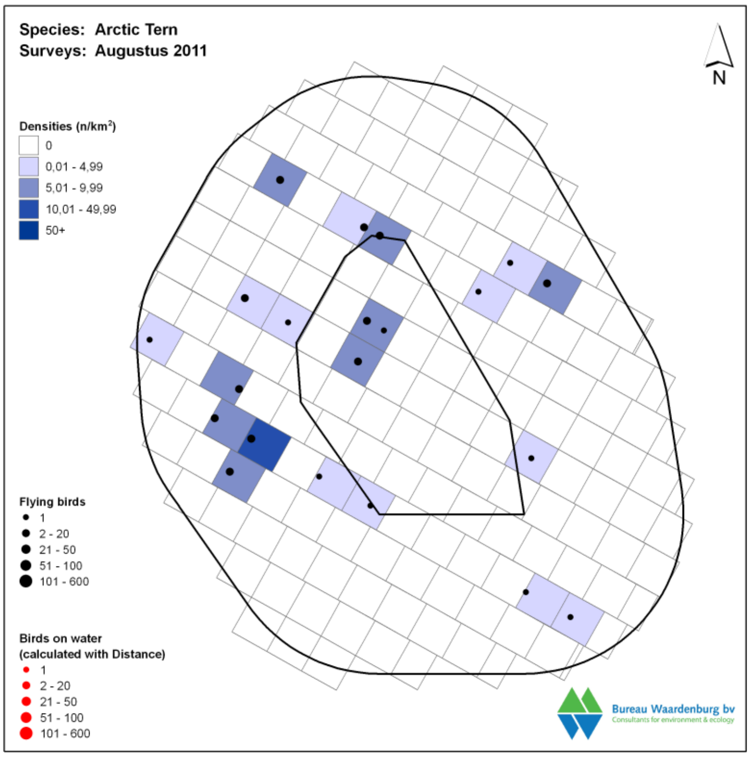


Figure 4.87 Arctic tern density in August, Year 2

In August of Year 3, Arctic terns were recorded in highest density in the south of the offshore site and buffer area, with fewer birds recorded elsewhere (Figure 4.88).

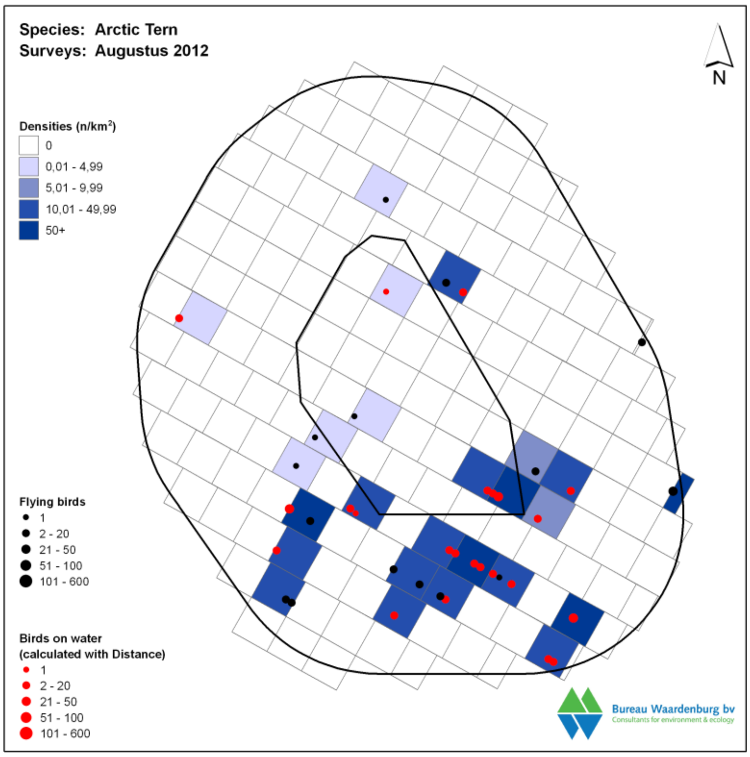


Figure 4.88 Arctic tern density in August, Year 3

A total of 1,186 Arctic terns were recorded in flight on baseline surveys, with almost all birds (99.8%) flying below 27.5 m in height (Table 4.3). A total of two birds (0.2%) were recorded flying above 27.5 m, at an estimated heights of 30 m. In addition, a further 216 unidentified common/Arctic terns and 34 unidentified tern species were recorded in flight on baseline surveys. All unidentified terns were recorded flying below 27.5 m.

Foraging behaviour was recorded for 694 Arctic terns in the offshore site and 8 km buffer area on baseline surveys, with five types of foraging behaviour recorded (Table 4.121). The majority of all foraging birds were recorded actively searching (48.7%) and dipping (49.1%). 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 4.121 Arctic tern foraging behaviour in the offshore site and 8 km buffer area in Years 1 to 3

Behaviour	Number of birds
Actively searching	338
Deep plunging	7
Dipping	341
Shallow plunging	1
Surface pecking	7
Total	**694**

4.4.20.3 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).

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 4.122). 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 4.89.

Table 4.122 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	1,275	2012
Farne Islands	72	2,840	0.3	6.5	1,866	2012
Firth of Forth Islands	**16**	**540**	**<0.1**	**1.2**	**265**	**2012**
Total	**-**	**4,080**	**>0.4**	**9.3**	**3,406**	**-**

Sources: 1 JNCC (2013) – SPA online species accounts. 2 SMP (2013) – 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).

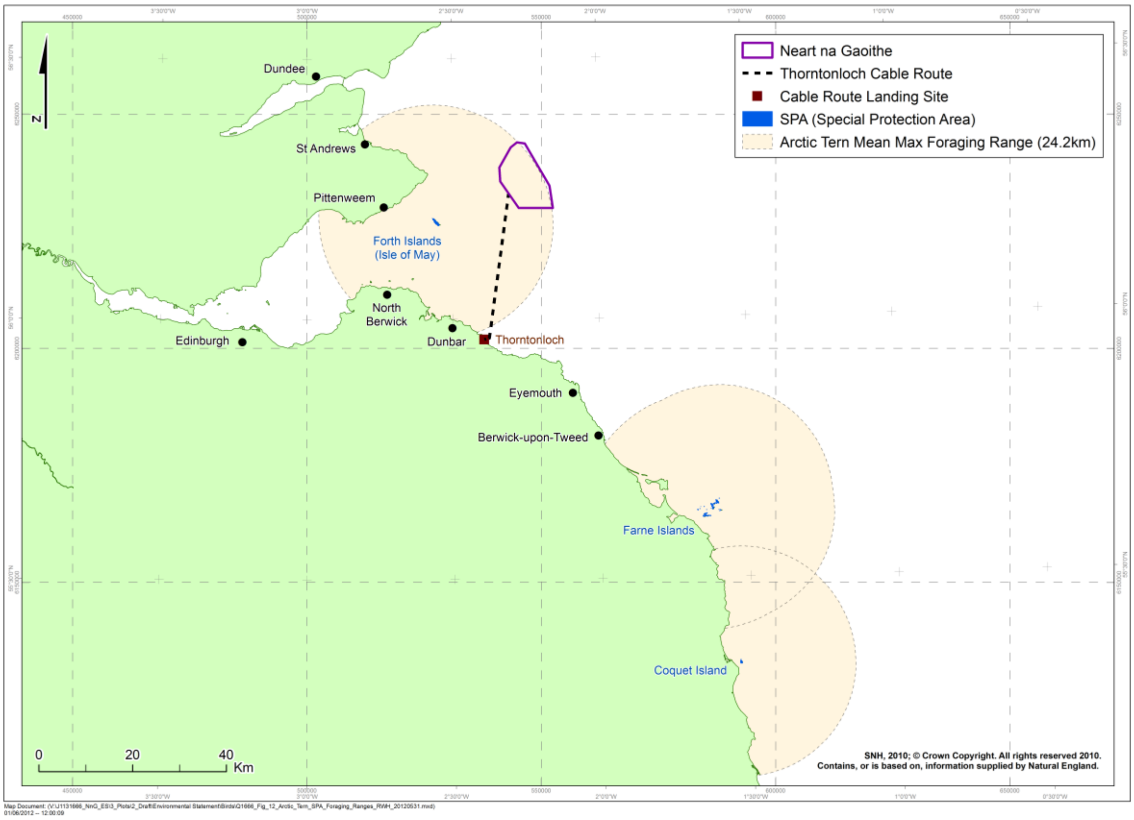


Figure 4.89 Arctic tern mean maximum foraging range from breeding SPAs in relation to the Development

4.4.20.4 Assessment

Definition of seasons

Arctic terns were mainly recorded in the offshore site in August, which was considered the autumn passage period for this assessment. 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.

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 was estimated to be 4,627 pairs during Seabird 2000 (Mitchell *et al.,* 2004). 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 use for the assessment is the number breeding on the east coast of Scotland, including Orkney and Shetland, and the north-east of England. Seabird 2000 estimated this number to be 44,231 pairs or 88,462 adults (Mitchell *et al.*, 2004).

Nature conservation importance

The Nature Conservation Importance of Arctic tern using the offshore site is rated as high, 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.

Offshore wind farm studies

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).

Construction Phase

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.

Operational Phase

Displacement

It is not known whether Arctic terns will be displaced by the proposed Neart na Gaoithe development. The review undertaken by Langston (2010) suggested that Arctic terns are at low risk of displacement effects. The majority of Arctic terns in the offshore site on baseline surveys were recorded 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 Arctic terns is ***not significant*** under the terms of 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%. Further details are presented in Ornithology Appendix 2: Collision Rate Estimates of Seabirds at Neart na Gaoithe.

Scaling this up to the size of the Arctic tern population of 88,462 adults that might pass through would give a worst case scenario of 98 collisions per annum for a 98.0% avoidance rate. This corresponds to approximately 0.1% of the eastern Scotland and north-east England breeding population (88,462 adults).

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. Firstly, it is likely that a smaller proportion of Arctic tern flight activity would be at rotor height, as just 0.2% of 1,186 flying Arctic terns recorded in baseline surveys were above the proposed rotor height, compared to 3% assumed here. Secondly, 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 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.

Decommissioning Phase

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.

Summary of combined effects

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 in the autumn passage period 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 4.123).

Table 4.123 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

4.4.20.5 Cumulative Impact Assessment

There were no significant impacts predicted for the regional population of Arctic terns in the breeding or non-breeding periods from the proposed Neart na Gaoithe development. The predicted effects of the development on the regional population of Arctic terns in the autumn passage period arising from construction, operation and decommissioning are very close to no effect.

Based on these findings, it was considered that it is not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for this population, and therefore no further cumulative impact assessment was undertaken for this species.

4.4.20.6 Mitigation measures

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

4.4.21 Common Guillemot *Uria aalge*

4.4.21.1 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 offshore site and 8 km buffer 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).

4.4.21.2 Offshore site and 8 km buffer area

Guillemot was one of the most frequently recorded seabirds on surveys in the Neart na Gaoithe study area during the baseline surveys, with a total of 7,898 birds recorded in Year 1, 11,730 birds in Year 2 and 11,557 birds in Year 3 (raw numbers, all sea states). The majority of guillemots were recorded in the buffer area (Table 4.2).

A further 3,323 unidentified guillemot/razorbills, and 1,348 unidentified auks were also seen on Year 1 surveys, with 1,532 unidentified guillemots/razorbills and 827 unidentified auks seen on the Year 2 surveys. In Year 3, a total of 1,767 unidentified guillemots/razorbills and 186 unidentified auks were recorded on surveys.

During the Year 1 "at-colony" part of the breeding season (April to June), the mean estimated number of guillemots in the offshore site was 260 birds, with a peak of 387 birds in June (Table 4.124). In the same period of Year 2, the mean estimated number of guillemots in the offshore site was 1,436 birds, with a peak of 3,789 birds in April. In Year 3, the mean estimated number of guillemots during the "at-colony" period was 1,246 birds, with a peak of 1,511 birds in April.

During the Year 1 "chicks-at-sea" period (July and August), the mean estimated number of guillemots in the offshore site was 86 birds, with a peak of 145 birds in July (Table 4.124). In the same period of Year 2, the mean estimated number of guillemots in the offshore site was 739 birds, with a peak of 1,129 birds in July. In Year 3, the mean estimated number of guillemots during the "chicks-at-sea" period was 3,022 birds, with a peak of 4,857 birds in August.

In the Year 1 post-breeding period (September and October), the mean estimated number of guillemots in the offshore site was 4,210 birds, with a peak of 7,020 birds in October (Table 4.124). In the same period of Year 2, the mean estimated number of guillemots in the offshore site was 1,641 birds, with a peak of 2,222 birds in September. In Year 3, the mean estimated number of guillemots during the post-breeding period was 1,585 birds, with a peak of 2,108 birds in September.

In the Year 1 non-breeding period (November to March), the mean estimated number of guillemots in the offshore site was 338 birds, with a peak of 994 birds in November (Table 4.124). In the same period of Year 2, the mean estimated number of guillemots in the offshore site was 581 birds, with a peak of 868 birds in December, although there was no November survey. In Year 3, the mean estimated number of guillemots during the non-breeding period was 828 birds, with a peak of 1,317 birds in November, although there was no December survey.

Mean estimated numbers of guillemots in the offshore site were highest in April at the start of the breeding season, and in the post-breeding period, with a three-year peak mean of 3,047 birds in October (Figure 4.90). Mean estimated numbers in the buffer zone were higher in all months, but showed a similar pattern, with a three-year peak mean in September (12,362 birds). The three-year mean estimates in the buffer area for September and October were slightly below 1% of the national breeding population (13,228 birds) (Mitchell *et al.,* 2004).

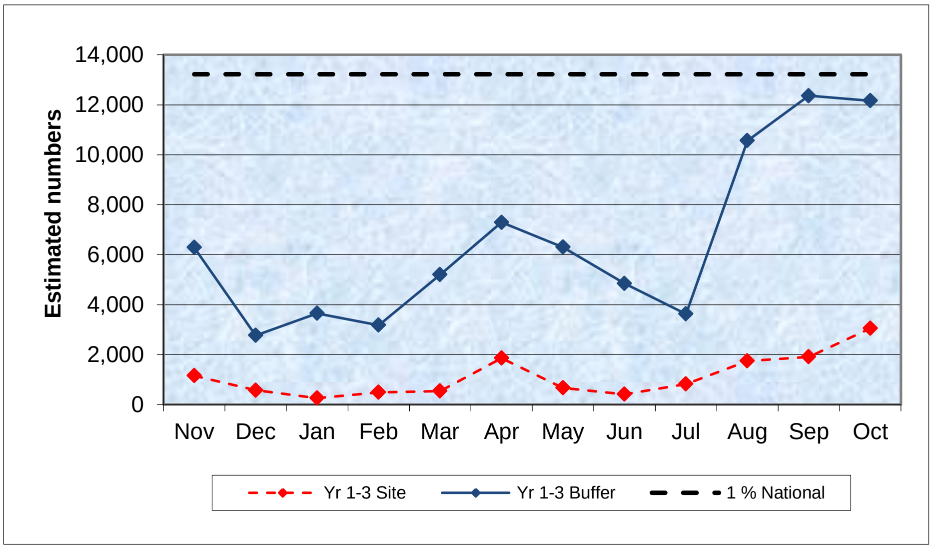


Figure 4.90 Mean monthly estimated numbers of guillemots in the Neart na Gaoithe Development & buffer areas in Years 1 to 3 (Three-year mean)

Table 4.124 Estimated numbers of guillemots in the offshore site (and 1, 2 & 8 km buffer) in Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 8 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	980	676	1,420	14	994	1,214	1,453	4,266
Yr1 Dec	270	160	456	7	277	317	337	1,774
Yr1 Jan	28	14	53	0	28	75	117	1,155
Yr1 Feb	68	40	116	34	102	129	176	1,641
Yr1 Mar	278	143	540	13	291	394	504	3,401
Yr1 Apr	129	67	250	143	272	542	924	3,418
Yr1 May	11	5	24	109	120	154	179	3,383
Yr1 Jun	380	273	529	7	387	461	698	5,256
Yr1 Jul	145	100	210	0	145	193	242	987
Yr1 Aug	27	14	52	0	27	40	174	1,522
Yr1 Sep	1,400	976	2,008	0	1,400	1,987	3,098	11,425
Yr1 Oct	6,986	5,406	9,027	34	7,020	9,491	11,174	24,017
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	847	600	1,196	21	868	1,277	1,708	4,911
Yr2 Jan	187	66	528	48	235	312	491	5,319
Yr2 Feb	535	268	1,070	20	555	756	1,079	3,657
Yr2 Mar	635	333	1,210	33	667	1,001	1,573	8,410
Yr2 Apr	3,531	1,551	8,036	259	3,789	4,100	4,323	8,131
Yr2 May	314	175	565	94	409	746	917	8,840
Yr2 Jun	36	20	65	74	110	171	241	4,597
Yr2 Jul	1,095	742	1,617	34	1,129	1,436	1,827	7,530
Yr2 Aug	328	194	555	20	349	553	949	5,857
Yr2 Sep	2,222	1,485	3,325	0	2,222	3,839	7,140	22,042
Yr2 Oct	1,053	707	1,568	7	1,060	1,396	2,048	13,061
Yr3 Nov	1,268	960	1,676	49	1,317	2,140	2,958	10,621
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	377	263	541	130	507	702	1,020	5,259
Yr3 Feb	784	538	1,141	35	818	1,109	1,659	5,715
Yr3 Mar	657	423	1,019	14	670	957	1,374	5,415
Yr3 Apr	1,470	758	2,848	41	1,511	2,243	2,965	15,909
Yr3 May	1,381	740	2,575	96	1,477	1,979	2,445	8,683
Yr3 Jun	660	465	937	90	750	1,094	1,521	5,936
Yr3 Jul	1,186	909	1,546	0	1,186	1,581	1,758	4,824
Yr3 Aug	4,857	2,468	9,557	0	4,857	6,891	9,081	29,553
Yr3 Sep	2,108	1,153	3,853	0	2,108	2,739	3,610	9,348
Yr3 Oct	1,054	685	1,623	7	1,061	1,855	2,513	8,558

Between November and March of Year 1, guillemots were widespread throughout the offshore site and buffer area at mostly low to moderate densities, with high densities in the north-west of the offshore site and in the east and west of the buffer area (Figure 4.91).

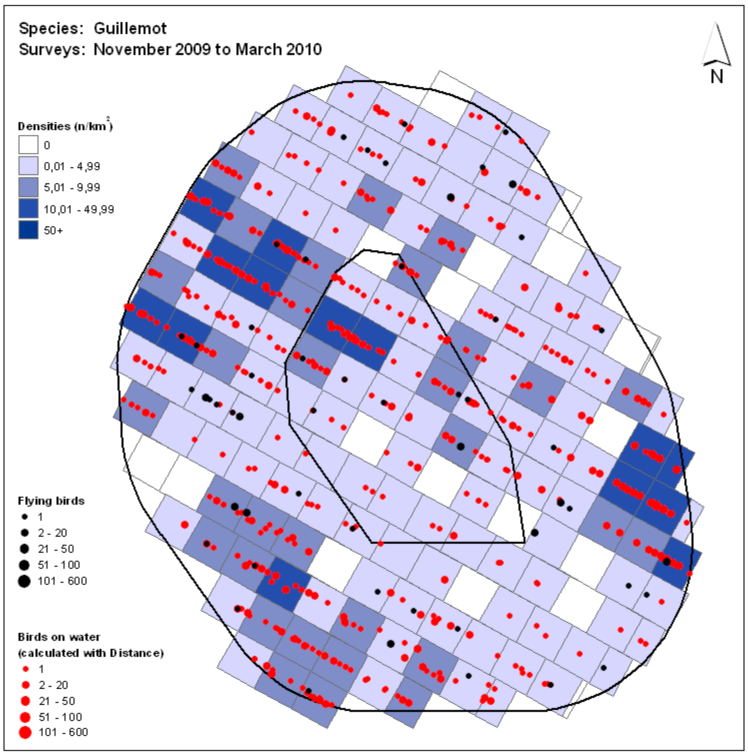


Figure 4.91 Guillemot density between November and March, Year 1

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 4.92).

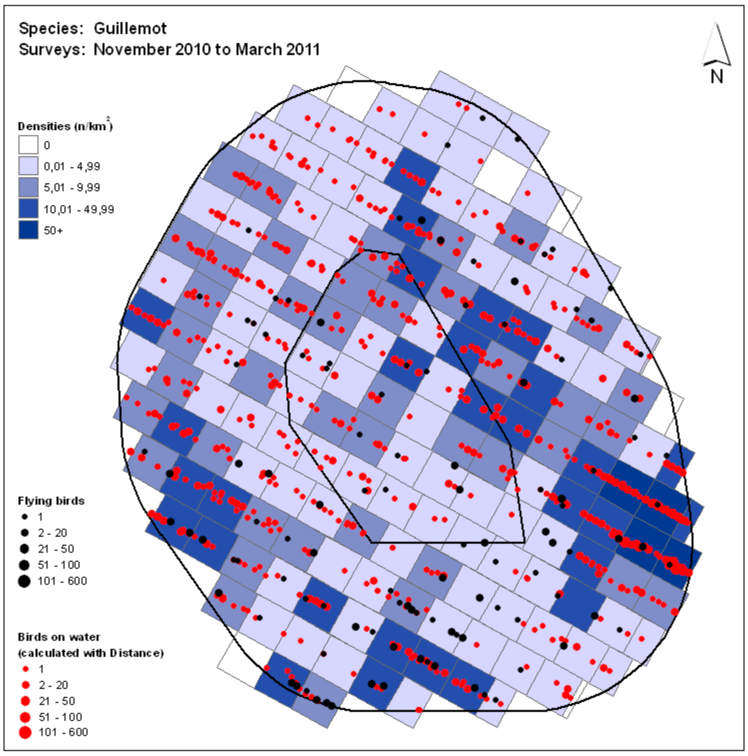


Figure 4.92 Guillemot density between November and March, Year 2

Guillemots were more widespread at higher densities across the offshore site and the buffer area in the Year 3 non-breeding period compared to the same period in Years 1 and 2 (Figure 4.93).

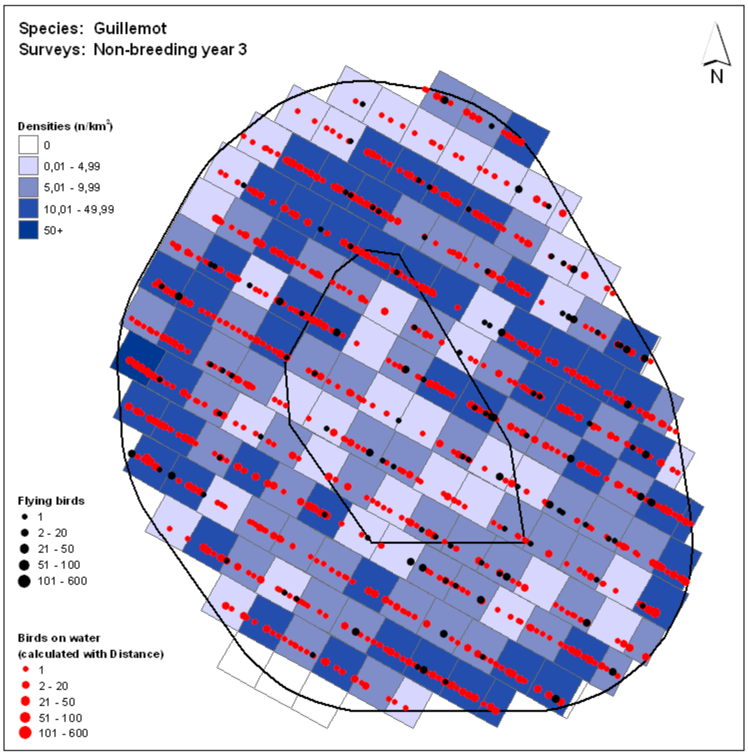


Figure 4.93 Guillemot density between November and March, Year 3

During the Year 1 "at-colony" part of the breeding season, guillemots were widespread across the offshore site at mostly low densities (Figure 4.94). Guillemots were also widespread in the buffer area, with highest densities in the south-east at this time.

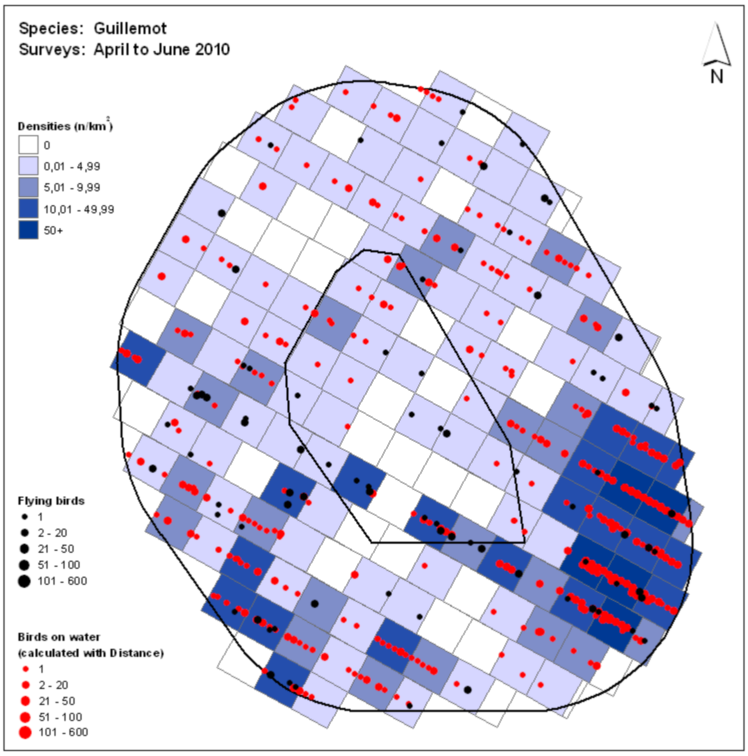


Figure 4.94 Guillemot density between April and June, Year 1

A similar distribution pattern was recorded over the same period in Year 2, with highest densities of guillemots in the south-east of the buffer area (Figure 4.95). Densities in the offshore site were mostly low to moderate at this time.

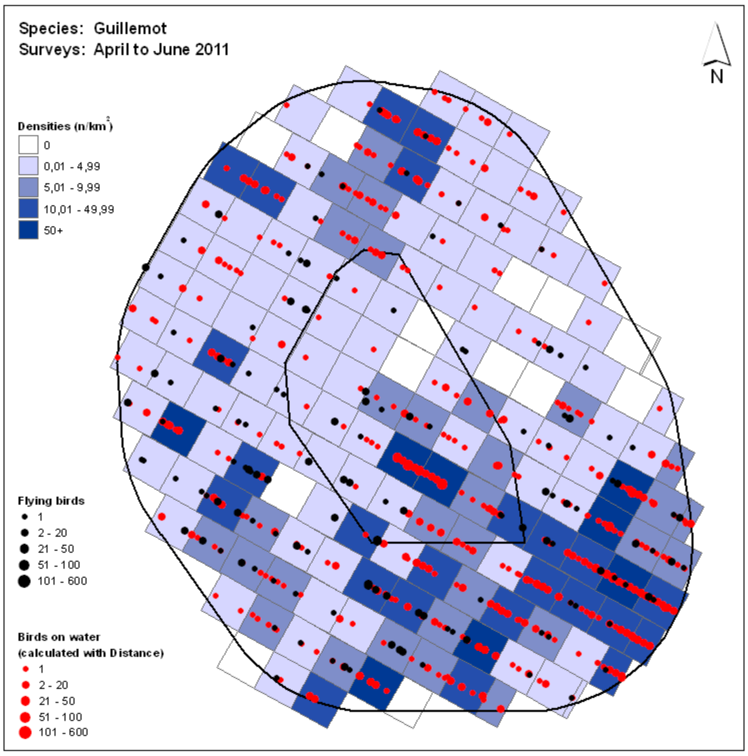


Figure 4.95 Guillemot density between April and June, Year 2

Over the same period in Year 3, guillemots were again widespread throughout the offshore site and buffer area, although the distribution of the high density areas was more extensive compared to the same period in Years 1 and 2, with a band of high density running from the north-west to the south-east through the centre of the site and buffer area (Figure 4.96). Lower densities were recorded to the north-east and south-west at this time.

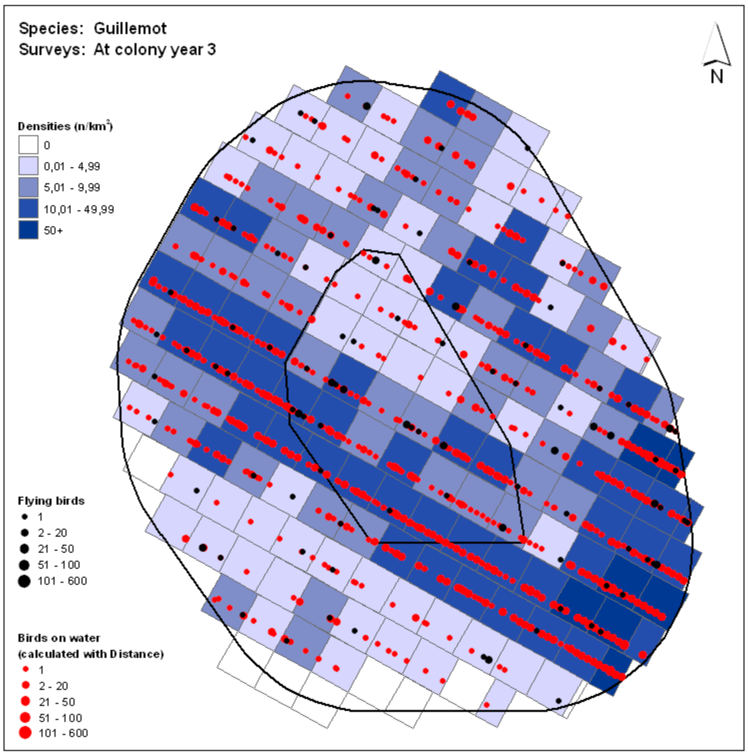


Figure 4.96 Guillemot density between April and June, Year 3

Guillemots were least widespread with lowest density in the “chicks-at-sea” period of the breeding season in Year 1 (Figure 4.97). Birds were largely absent from the offshore site at this time, apart from low densities in the north-east. 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.

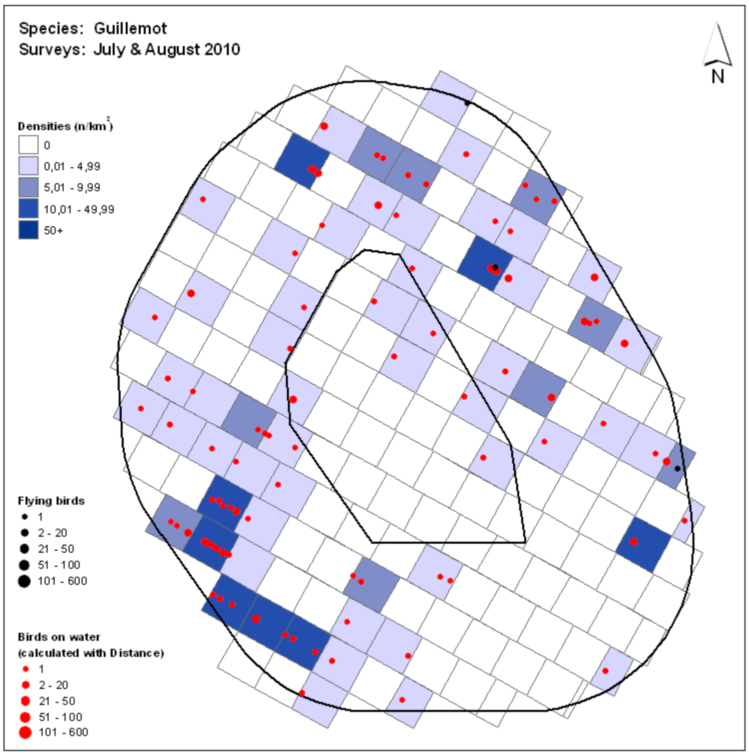


Figure 4.97 Guillemot density in July and August, Year 1

Guillemots were more widespread at high densities in July and August of Year 2, compared to Year 1 (Figure 4.98). 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-east and south-west, with lower densities elsewhere.

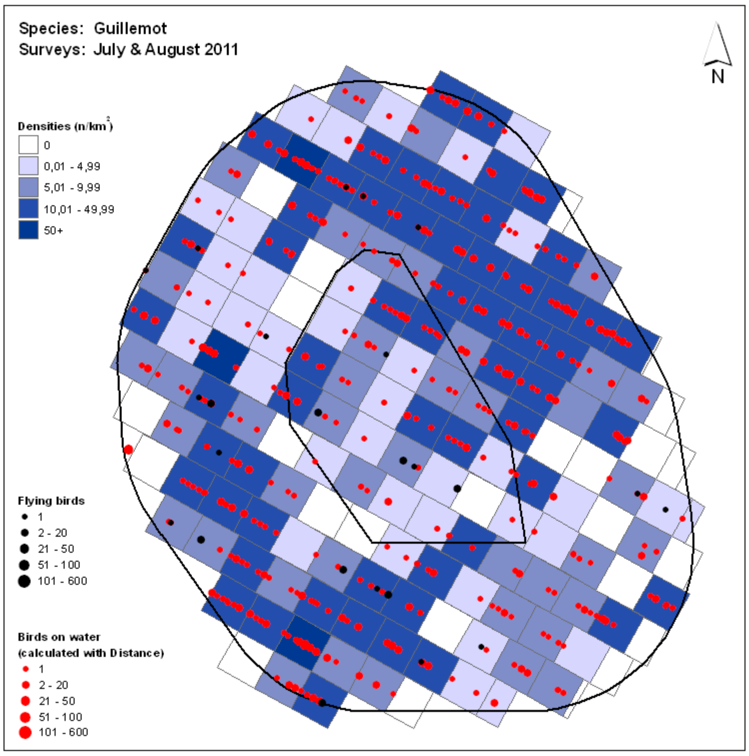


Figure 4.98 Guillemot density in July and August, Year 2

Guillemots were again widespread in the offshore site and buffer area over the same period in Year 3, although the distribution of the high density areas was different to Year 2 (Figure 4.99). Highest densities were concentrated in the south-west of the offshore site and the buffer area, with lower densities to the north and east.

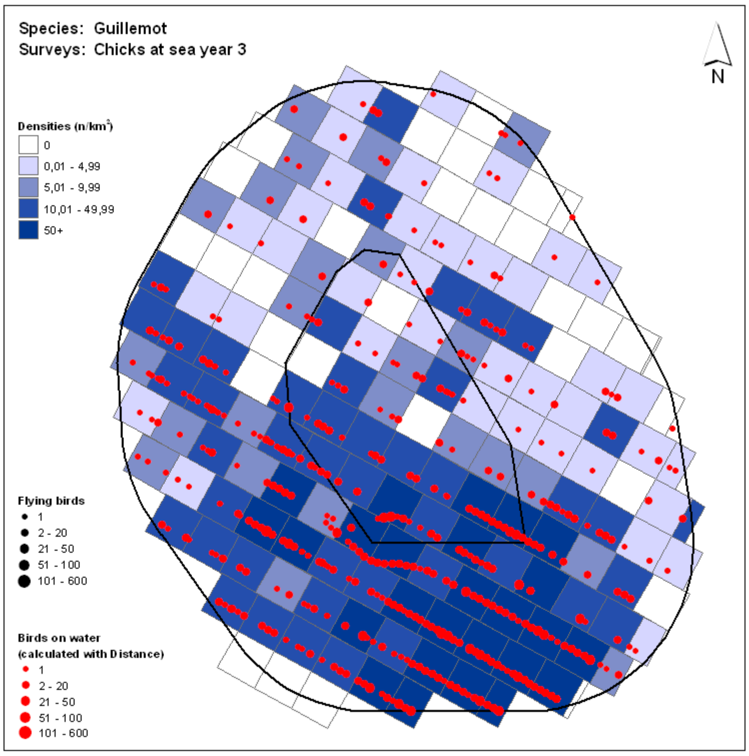


Figure 4.99 Guillemot density in July and August, Year 3

In the Year 1 post-breeding period, birds were widespread at mostly high densities across the offshore site and most of the buffer area, apart from the south-east of the buffer area, where densities were low or zero (Figure 4.100).

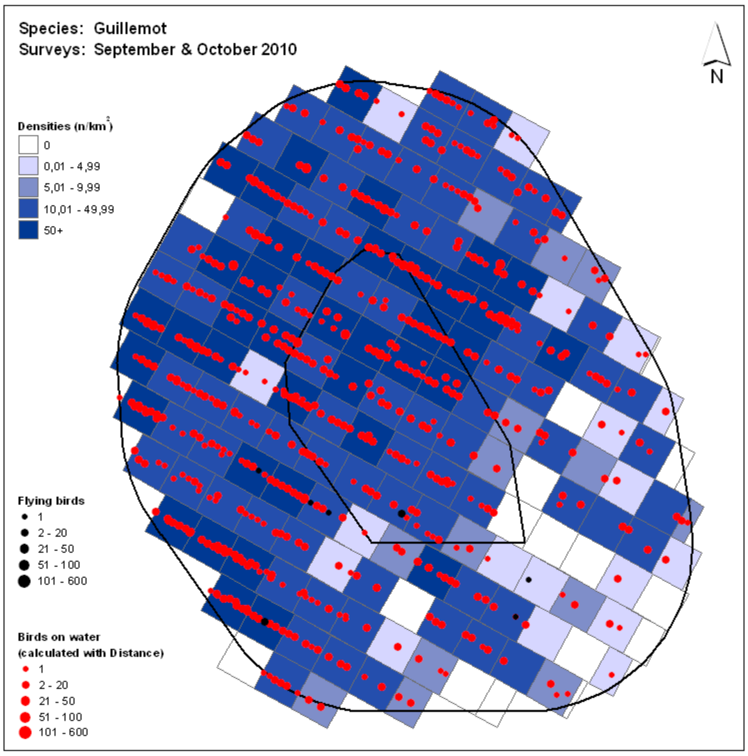


Figure 4.100Guillemot density in September and October, Year 1

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 4.101). In the east of the offshore site and buffer area, densities were lower or zero at this time.

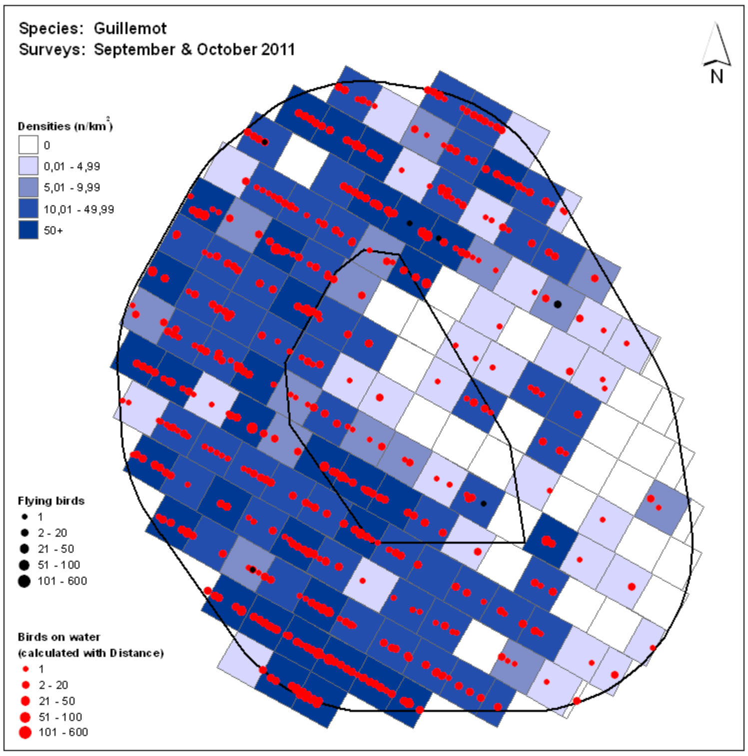


Figure 4.101 Guillemot density in September and October, Year 2

In the Year 3 post-breeding period, guillemots were again widespread at mostly high densities in the offshore site and buffer area (Figure 4.102). The main concentrations of guillemots were in the south-west of the offshore site and buffer area, with lower densities in the north-east of the buffer area at this time.

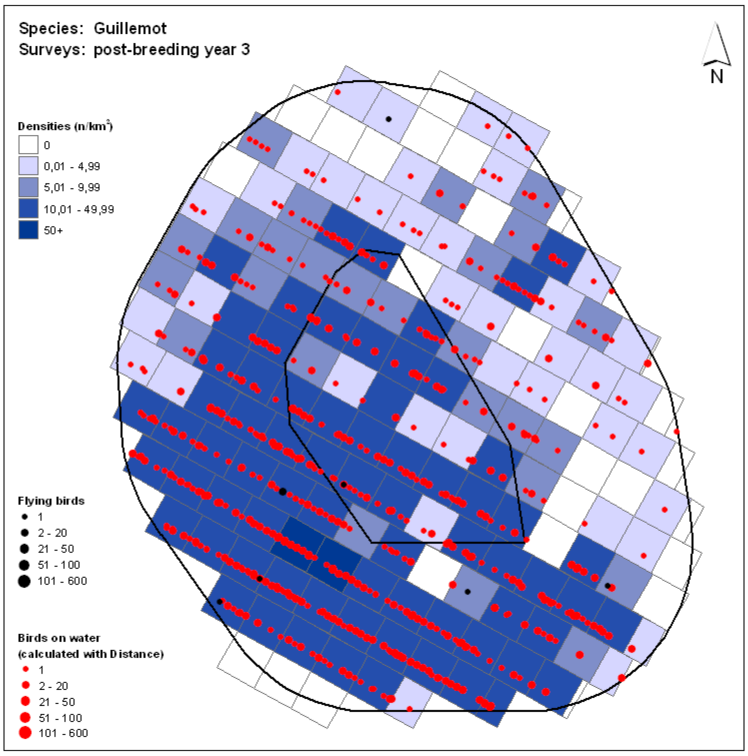


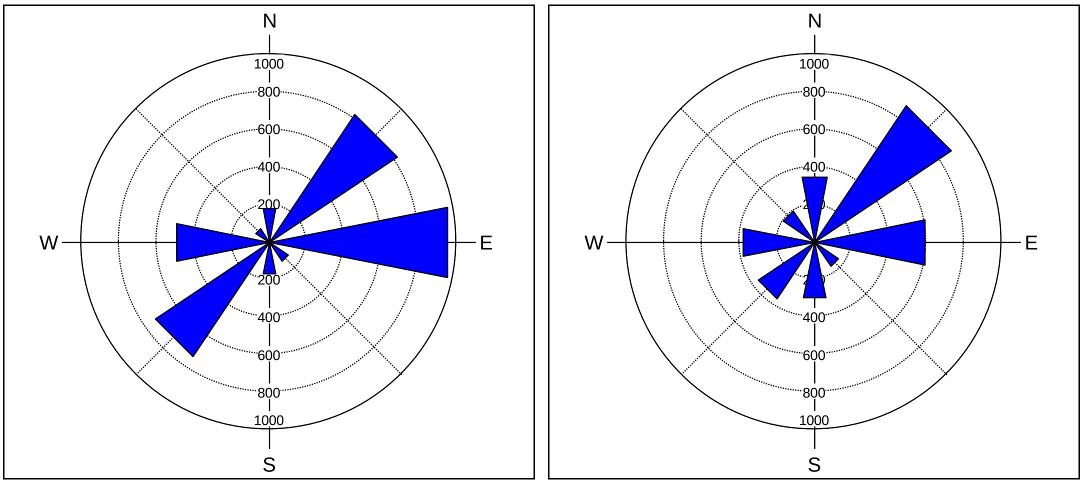
Figure 4.102 Guillemot density in September and October, Year 3

A total of 6,812 guillemots were recorded in flight, with almost all birds recorded flying below 27.5 m in height (Table 4.3). The majority of birds (98.6%) were recorded flying below 7.5 m in height. One bird (0.01%) was recorded flying above 27.5 m i.e. within the rotor-swept zone, at an estimated height of 30 m.

A further 1,453 unidentified guillemots/razorbills and 141 unidentified auk species were also recorded on baseline surveys. All unidentified auks were recorded flying below 27.5 m (Table 4.3).

Flight direction was recorded for 3,571 guillemots in the "at colony" period of the breeding season (April to June), with direction recorded for 3,238 guillemots in the "chicks-at-sea", post-breeding and non-breeding periods combined (July to March) (Figure 4.103).

In the breeding season, just over a quarter of all birds recorded were flying east (27.0%), with 22.8% of birds flying north-east. In the post-breeding and non-breeding seasons, just over a quarter of birds were recorded flying north east (27.0%), with 18.6% flying east.



April to June (n=3,571 birds) July to March (n=3,238 birds)
Numbers shown on figures are number of birds recorded

Figure 4.103 Flight direction of guillemots in the offshore site and 8 km buffer area in Years 1 to 3

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).

A recent JNCC statistical analysis of ESAS data investigating possible marine SPAs, identified waters to the east of the offshore site and 8 km buffer area as an important area for guillemots during the breeding season (Kober *et al.,* 2010). The waters around the offshore site and 8 km buffer area were also identified as an important area for guillemots during the non-breeding season (October to April) (Kober *et al.,* 2010).

Foraging behaviour was recorded for 269 guillemots in the offshore site and 8 km buffer area on baseline surveys, with five types of foraging behaviour recorded (Table 4.125). The majority of all foraging birds were recorded holding fish (55.0%) and pursuit diving (34.2%). Prey was identified for 25 prey items, with 24 sandeels and one herring/sprat recorded.

Table 4.125 Guillemot foraging behaviour in the offshore site and 8 km buffer area in Years 1 to 3

Behaviour	Number of birds
Actively searching	14
Pursuit diving	92
Pursuit plunging	3
Holding fish	148
Surface pecking	12
Total	**269**

4.4.21.3 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).

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 4.126). These SPAs held 65.2% of the UK breeding population and 20.4% of the biogeographic population at the time of designation (JNCC, 2013). 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 4.104.

The full extent of the St Abb's Head to Fast Castle SPA guillemot colony has not been surveyed since 1998, for Seabird 2000 (R Mavor pers. comm.). However, breeding guillemots at St Abb's Head NNR, the major component of the SPA, were counted in 2008. To estimate the size of the remaining SPA population, the percentage decline in guillemot numbers at St Abb's Head NNR between 1998 (40,720 birds) and 2008 (33,181 birds) was calculated (-18.5% decline). This was applied to the population of the remainder of the SPA for 1998 (1,514 birds) to estimate the likely size of the population for this area in 2008 (1,234 birds). The two 2008 values were then added to get a total estimate for the whole SPA for 2008 (34,415 birds).

Table 4.126 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**	**49,076**	**2012**
Forth Islands	**16**	**22,452**	**1.0**	**3.2**	**22,553**	**2012**
Fowlsheugh	**62**	**40,140**	**1.8**	**5.7**	**44,920**	**2012**
St Abb's Head to Fast Castle	**31**	**20,971**	**0.9**	**3.0**	**34,415 [3]**	**2008**
Total	**-**	**115,702**	**5.1**	**16.4**	**171,822**	**-**

Sources: 1 JNCC (2013) – SPA online species accounts. 2 SMP (2013) – 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).
3. 2008 count for St Abb's Head NNR (33,181 birds) (SMP 2013) plus corrected estimate for rest of SPA colony – see text for full explanation

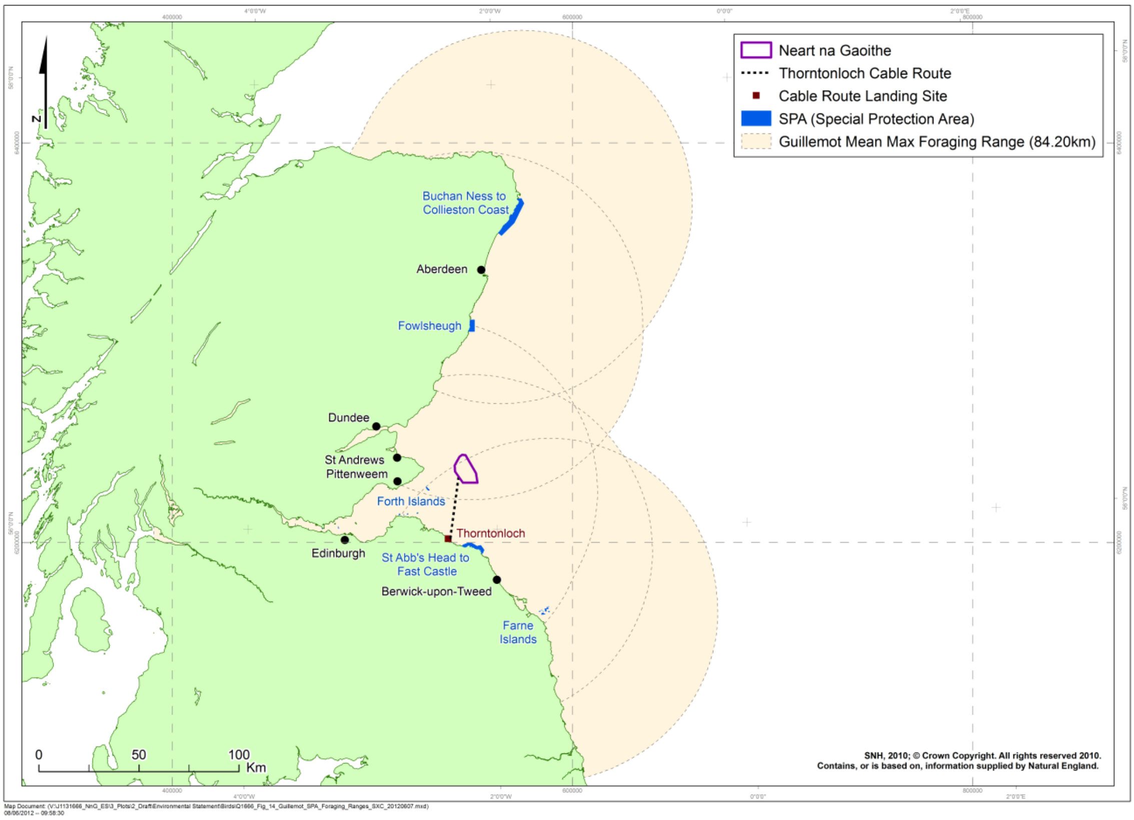


Figure 4.104 Guillemot mean maximum foraging range from breeding SPAs in relation to the Development

4.4.21.4 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.

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 at-colony period, when breeding adults are attending colonies, was 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 chicks-at-sea period was defined as July and August and corresponds to the time when male adults have accompanying dependent young with them on the sea. In this period of the breeding season, 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 chicks-at-sea period are from breeding sites within the region, particularly the relatively close large colonies. Some adult guillemot undergo wing moult in this 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 majority of individuals present at this time are still from colonies 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 non-breeding period was 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).

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, 2013) (Table 4.127).

Table 4.127 Recent counts & Seabird 2000 counts at main colonies for breeding guillemot between Peterhead and Blyth

Colony	Distance to site (km)	Seabird 2000 count	Recent count (birds at colony)	Year	Percentage change since Seabird 2000
Buchan Ness to Collieston Coast	113	29,389 [1]	20,858 [1]	2007	-29.0%
Fowlsheugh	62	61,420 [2]	44,920 [1]	2012	-26.9%
Isle of May	16	28,103 [2]	16,991 [1]	2012	-39.5%
St Abb's Head NNR	33	40,720 [1]	33,181 [1]	2008	-18.5%
Farne Islands	72	35,436 [1]	49,076 [1]	2012	+38.5%
				Mean percentage change	**-15.1%**

Sources: 1 SMP (2013) – Seabird Monitoring Programme Online Database. 2 Mitchell *et al.*, 2004.

Assuming that the sub-set of colonies that have been recently counted is representative, the regional decline since the Seabird 2000 counts amounts to -15.1%. 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 for Peterhead to Blyth derived from Seabird 2000 results (218,905 birds) (Mitchell *et al.,* 2004) was adjusted downwards by 15.1%. On this basis, the regional breeding population was estimated to be 185,973 birds, taking into account recent changes in numbers at the main breeding colonies in the region.

The SPA breeding population within mean maximum foraging range of the offshore site was estimated to be 171,822 birds, based on most recent available counts (Table 4.126).

The size of the post-breeding-period regional guillemot population is estimated to be the same as the regional population between Peterhead and Blyth (185,973 birds).

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 birds from more distant colonies.

Nature conservation importance

The nature conservation importance of guillemot using the offshore site is categorised as high during the breeding season, 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. The three-year mean peak estimated number of guillemots present in the offshore site between April and June (the at-colony period of the breeding season) (1,654 birds) is 1.0% of the SPA breeding population within mean maximum foraging range of the offshore site (171,822 birds).

Offshore wind farm studies

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).

One year of post-construction monitoring at Robin Rigg Offshore Wind Farm Data suggested a decline in guillemot abundance during the construction phase with numbers increasing again during operational year one, although a smaller increase was observed within the turbine area, indicating potential displacement (Walls *et al.,* 2013).

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.

Construction Phase

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.

Operational Phase

Displacement

In order to assess the potential impacts from displacement, the approach presented in the interim guidance note on displacement issued by the JNCC and NE has been followed (JNCC & NE 2012).

Peak estimated numbers of guillemots in the offshore site in the at-colony period of the breeding season (April to June), the chicks-at-sea period (July and August), the post-breeding season (September and October) and the non-breeding season (November to March) for Years 1 to 3 were averaged to get the three-year mean peak per season. This was repeated for a 1 km and 2 km buffer around the offshore site (Table 4.128).

The three-year mean peak estimated number of guillemots was then used to predict the estimated number of birds at potential risk of mortality following displacement in the at-colony period of the breeding season in the offshore site (plus 1 km and 2 km buffers), as recommended in the draft guidance note on displacement (JNCC & NE, 2012).

This was repeated for the chicks-at-sea period of the breeding season, the post-breeding period and the non-breeding period.

Table 4.128 Seasonal three-year mean peak estimated numbers of guillemots in the offshore site (plus 1 & 2 km buffer)

Year	Offshore site			
	At colony	Chicks-at-sea	Post-breeding	Non-breeding
Year 1	387	145	7,020	994
Year 2	3,789	1,129	2,222	868
Year 3	1,511	4,857	2,108	1,317
3-year mean peak	1,896	2,044	3,783	1,060
Year	Offshore site + 1 km			
	At colony	Chicks-at-sea	Post-breeding	Non-breeding
Year 1	542	193	9,491	1,214
Year 2	4,100	1,436	3,839	1,277
Year 3	2,243	6,891	2,739	2,140
3-year mean peak	2,295	2,840	5,356	1,544
Year	Offshore site + 2 km			
	At colony	Chicks-at-sea	Post-breeding	Non-breeding
Year 1	924	242	11,174	1,453
Year 2	4,323	1,827	7,140	1,708
Year 3	2,965	9,081	3,610	2,958
3-year mean peak	2,737	3,717	7,308	2,040

Table 4.129 Estimated number of guillemots at risk of mortality following displacement from offshore site in "at-colony" period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	4	9	19	38	57	76	95	114	133	152	171	190
	20%	8	19	38	76	114	152	190	228	265	303	341	379
	30%	11	28	57	114	171	228	284	341	398	455	512	569
	40%	15	38	**76**	152	228	303	379	455	531	607	683	758
	50%	19	47	95	190	284	379	474	569	664	758	853	948
	60%	23	57	114	228	341	455	569	683	796	910	1,024	1,138
	70%	27	66	133	265	398	531	664	796	929	1,062	1,194	1,327
	80%	30	76	152	303	455	607	758	910	1,062	1,213	1,365	1,517
	90%	34	85	171	341	512	683	853	1,024	1,194	1,365	1,536	1,706
	100%	38	95	190	379	569	758	948	1,138	1,327	1,517	1,706	1,896
Three-year mean peak of 1,896 guillemots in the offshore site in "at-colony" period													
SPA breeding population within mean max foraging range (84.2 km) = 171,822 birds (JNCC 2013, SMP 2013)													

Table 4.130 Estimated number of guillemots at risk of mortality following displacement from offshore site plus 1 km buffer in "at-colony" period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	5	11	23	46	69	92	115	138	161	184	207	230
	20%	9	23	46	92	138	184	230	275	321	367	413	459
	30%	14	34	69	138	207	275	344	413	482	551	620	689
	40%	18	46	**92**	184	275	367	459	551	643	734	826	918
	50%	23	57	115	230	344	459	574	689	803	918	1,033	1,148
	60%	28	69	138	275	413	551	689	826	964	1,102	1,239	1,377
	70%	32	80	161	321	482	643	803	964	1,125	1,285	1,446	1,607
	80%	37	92	184	367	551	734	918	1,102	1,285	1,469	1,652	1,836
	90%	41	103	207	413	620	826	1,033	1,239	1,446	1,652	1,859	2,066
	100%	46	115	230	459	689	918	1,148	1,377	1,607	1,836	2,066	2,295
Three-year mean peak of 2,295 guillemots in offshore site & 1 km buffer in "at-colony" period													
SPA breeding population within mean max foraging range (84.2 km) = 171,822 birds (JNCC 2013, SMP 2013)													

Table 4.131 Estimated number of guillemots at risk of mortality following displacement from offshore site plus 2 km buffer in "at-colony" period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	5	14	27	55	82	109	137	164	192	219	246	274
	20%	11	27	55	109	164	219	274	328	383	438	493	547
	30%	16	41	82	164	246	328	411	493	575	657	739	821
	40%	22	55	**109**	219	328	438	547	657	766	876	985	1,095
	50%	27	68	137	274	411	547	684	821	958	1,095	1,232	1,369
	60%	33	82	164	328	493	657	821	985	1,150	1,314	1,478	1,642
	70%	38	96	192	383	575	766	958	1,150	1,341	1,533	1,724	1,916
	80%	44	109	219	438	657	876	1,095	1,314	1,533	1,752	1,971	2,190
	90%	49	123	246	493	739	985	1,232	1,478	1,724	1,971	2,217	2,463
	100%	55	137	274	547	821	1,095	1,369	1,642	1,916	2,190	2,463	2,737
Three-year mean peak of 2,737 guillemots in offshore site & 2 km buffer in "at-colony" period													
SPA breeding population within mean max foraging range (84.2 km) = 171,822 birds (JNCC 2013, SMP 2013)													

Likely impacts of displacement

Colony attendance period

There is evidence from existing offshore wind farms indicating that displacement levels for guillemots are likely to be below 50% (Leopold *et al.,* 2011). For this assessment, it was assumed that there will be 40% displacement of guillemots from the offshore site in all seasons. Additional scenarios considering 40% displacement out to a 1 km and a 2 km buffer are also presented.

Assuming 40% of all guillemots were displaced from the offshore site during the "at-colony" part of the breeding season (April to June), this would affect an estimated 758 birds (Table 4.129), increasing to 1,095 birds if displacement is assumed to affect the offshore site and a 2 km buffer (Table 4.131).

Assuming that 30% of the population present are immature birds (Wanless *et al.,* 1998), then this would mean 531 breeding adults would be displaced from the offshore site, increasing to 766 breeding adults if displacement is assumed to affect the offshore site and a 2 km buffer.

For the purposes of this assessment, it was assumed that 10% of all guillemots displaced from the offshore site during the breeding season (up to 109 birds) would die as a result. Applying the correction for immature birds being present in the colony attendance part of the breeding season (Wanless *et al.,* 1998), gives an estimated mortality of up to 76 breeding adults. This corresponds to approximately 0.04% of the regional breeding SPA population within mean maximum foraging range of the offshore site (171,822 birds). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

For the remaining 766 breeding adult guillemots displaced from the offshore site and 2 km buffer that survived, there could potentially be a detrimental impact on their breeding success, as a result of having to forage elsewhere. This number corresponds to approximately 0.4% of the regional breeding SPA population within mean maximum foraging range of the offshore site (171,822 birds).

However, it is likely that guillemots can compensate for a moderate amount of displacement by choosing to forage elsewhere. Guillemot GPS tracking data from the Isle of May in June 2010, based on a sample size of 33 individuals (112 foraging trips), found that guillemots showed a strong affinity to coastal as well as offshore regions. Birds departed from and returned to the Isle of May primarily on a bearing between north and east, and favoured depths of 40-50 m over depths of 60-70 m. Mean maximum foraging range from the Isle of May colony was 18 km, with a maximum foraging range of 61 km recorded (Daunt *et al.,* 2011a). The offshore site is within the mean maximum foraging range recorded by this study, as it is approximately 16 km from the Isle of May.

However, analysis of at-sea distributions of guillemots using kernel density estimations found that the offshore site did not overlap to any great extent with the core area used by foraging guillemots (50% kernals). The offshore site was within the overall area used by tagged foraging guillemots in 2010 (90% kernels). The core area of use (50% kernels) was estimated to cover an area of 540 km^2, while the overall area of active use (90% kernels) was estimated at 1,845 km^2 (Daunt *et al.,* 2011a).

The 2010 study also examined differences in foraging range between 2010 and previous breeding seasons, and found substantial inter-annual variation falling into two clear categories; more restricted distributions in 1997, 2002 and 2003 (< 5% of fixes >25 km from colony) and more extensive foraging ranges in 1998, 1999 and 2010 (> 35% of fixes >25 km from colony). However, breeding success did not differ significantly between the three years of broader foraging range versus the three years of more restricted foraging range. Analysis of this data suggested that 2010 had the most extensive foraging range recorded across the six study years (Daunt *et al.,* 2011a).

Daytime watches at Fowlsheugh and St Abb's Head in summer 2011 recording trip duration and flight direction of individual birds were compared with GPS tracking data from the Isle of May in 2010 to give estimated foraging ranges of guillemots from these two SPA colonies. The results suggested that breeding guillemots from Fowlsheugh did not use the offshore site (approximately 62 km away) in 2011. Mean maximum foraging range for guillemots from Fowlsheugh in 2011 was estimated at 12.2 km, with a maximum foraging range of 55.6 km. The majority of birds flew east from the colony (Daunt *et al.,* 2011b).

Mean maximum foraging range for guillemots from St Abb's Head in 2011 was estimated at 16.6 km, with a maximum foraging range of 55.0 km. The majority of birds flew north-east from the colony. St Abb's Head is approximately 31 km

However, it should be noted that calculations for these results were based on GPS data from the Isle of May in 2010, and there may be differences between colonies that were not taken into account in this study. Also, there is likely to be variations in foraging range between years (Daunt *et al.,* 2011b).

Other studies have estimated mean maximum foraging range for guillemot as 84.2 km (Thaxter *et al.,* 2012), which suggests that guillemots from these two colonies may have a more restricted distribution than the broader picture, although this estimate was only based on one breeding season (Daunt *et al.,* 2011b).

The main conclusion that can be drawn from these studies is that guillemots are clearly capable of travelling considerable distances during the breeding season (Daunt *et al.,* 2011a). Studies also indicate that breeding success of guillemots is not significantly affected by having to forage at greater distances from the colony. It is therefore considered that should guillemots be partially displaced from the offshore site following construction of the wind farm, any impact on breeding success of these displaced birds is not likely to be significant.

Based on the above, the impact of displacement on the "at-colony" period 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-at-sea period

Assuming 40% of all guillemots were displaced from the offshore site during the "chicks-at-sea" part of the breeding season (July and August), this would affect an estimated 818 birds (Table 4.132), increasing to 1,487 birds if displacement is assumed to affect the offshore site and a 2 km buffer (Table 4.134).

Assuming that 30% of the population present are immature birds (Wanless *et al.,* 1998), then this would mean 573 breeding adults would be displaced from the offshore site, increasing to 1,041 breeding adults if displacement is assumed to affect the offshore site and a 2 km buffer.

Table 4.132 Estimated number of guillemots at risk of mortality following displacement from offshore site in "chicks-at-sea" period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	4	10	20	41	61	82	102	123	143	164	184	204
	20%	8	20	41	82	123	164	204	245	286	327	368	409
	30%	12	31	61	123	184	245	307	368	429	491	552	613
	40%	16	41	**82**	164	245	327	409	491	572	654	736	818
	50%	20	51	102	204	307	409	511	613	715	818	920	1,022
	60%	25	61	123	245	368	491	613	736	858	981	1,104	1,226
	70%	29	72	143	286	429	572	715	858	1,002	1,145	1,288	1,431
	80%	33	82	164	327	491	654	818	981	1,145	1,308	1,472	1,635
	90%	37	92	184	368	552	736	920	1,104	1,288	1,472	1,656	1,840
	100%	41	102	204	409	613	818	1,022	1,226	1,431	1,635	1,840	2,044
Three-year mean peak of 2,044 guillemots in the offshore site in "chicks-at-sea" period													
SPA breeding population within mean max foraging range (84.2 km) = 171,822 birds (JNCC 2013, SMP 2013)													

Table 4.133 Estimated number of guillemots at risk of mortality following displacement from offshore site plus 1 km buffer in "chicks-at-sea" period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	6	14	28	57	85	114	142	170	199	227	256	284
	20%	11	28	57	114	170	227	284	341	398	454	511	568
	30%	17	43	85	170	256	341	426	511	596	682	767	852
	40%	23	57	**114**	227	341	454	568	682	795	909	1,022	1,136
	50%	28	71	142	284	426	568	710	852	994	1,136	1,278	1,420
	60%	34	85	170	341	511	682	852	1,022	1,193	1,363	1,534	1,704
	70%	40	99	199	398	596	795	994	1,193	1,392	1,590	1,789	1,988
	80%	45	114	227	454	682	909	1,136	1,363	1,590	1,818	2,045	2,272
	90%	51	128	256	511	767	1,022	1,278	1,534	1,789	2,045	2,300	2,556
	100%	57	142	284	568	852	1,136	1,420	1,704	1,988	2,272	2,556	2,840
Three-year mean peak of 2,840 guillemots in offshore site & 1 km buffer in "chicks-at-sea" period													
SPA breeding population within mean max foraging range (84.2 km) = 171,822 birds (JNCC 2013, SMP 2013)													

Table 4.134 Estimated number of guillemots at risk of mortality following displacement from offshore site plus 2 km buffer in "chicks-at-sea" period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	7	19	37	74	112	149	186	223	260	297	335	372
	20%	15	37	74	149	223	297	372	446	520	595	669	743
	30%	22	56	112	223	335	446	558	669	781	892	1,004	1,115
	40%	30	74	**149**	297	446	595	743	892	1,041	1,189	1,338	1,487
	50%	37	93	186	372	558	743	929	1,115	1,301	1,487	1,673	1,859
	60%	45	112	223	446	669	892	1,115	1,338	1,561	1,784	2,007	2,230
	70%	52	130	260	520	781	1,041	1,301	1,561	1,821	2,082	2,342	2,602
	80%	59	149	297	595	892	1,189	1,487	1,784	2,082	2,379	2,676	2,974
	90%	67	167	335	669	1,004	1,338	1,673	2,007	2,342	2,676	3,011	3,345
	100%	74	186	372	743	1,115	1,487	1,859	2,230	2,602	2,974	3,345	3,717
Three-year mean peak of 3,717 guillemots in offshore site & 2 km buffer in "chicks-at-sea" period													
SPA breeding population within mean max foraging range (84.2 km) = 171,822 birds (JNCC 2013, SMP 2013)													

For the purposes of this assessment, it was assumed that 10% of all guillemots displaced from the offshore site during the "chicks-at-sea" part of the breeding season (up to 149 birds) would die as a result. Applying the correction for 30% immature birds being present in the breeding season (Wanless *et al.,* 1998), gives an estimated mortality of up to 104 breeding adults. This corresponds to approximately 0.06% of the regional breeding SPA population within mean maximum foraging range of the offshore site (171,822 birds). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

For the remaining 937 breeding adult guillemots displaced from the offshore site and 2 km buffer that survived, there could potentially be a detrimental impact on their breeding success, as a result of having to forage elsewhere. This number corresponds to approximately 0.5% of the regional breeding SPA population within mean maximum foraging range of the offshore site (171,822 birds).

However, for the reasons outlined above, it is likely that guillemots can compensate for a moderate amount of displacement by choosing to forage elsewhere. In addition, as they are no longer tied to the colony, birds are less restricted in where they can forage. Comparing the distribution of guillemots in the study area from Years 1 to 3 at this time of year shows that there is considerable variation between years. In July and August of Year 1, very few guillemots were present in the north and east of the offshore site, with no birds recorded elsewhere. Density elsewhere in the study area at this time was also generally low (Figure 4.97). In Year 2, guillemots were widespread throughout the offshore site and buffer area, with highest densities recorded in the north-east and south-west of the buffer area (Figure 4.98). In the same period of Year 3, highest densities were concentrated in the south-west of the offshore site and the buffer area, with lower densities to the north and east (Figure 4.99). These changes in seasonal distribution between years are most likely influenced by changes in the distribution of prey, and show that guillemots are not regularly relying on the offshore site exclusively at this time of year.

Based on the above, the impact of displacement on the "chicks-at-sea" part of the breeding season 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-at-sea" period of the breeding season are ***not significant*** under the EIA Regulations.

Post-breeding period

Assuming 40% of all guillemots were displaced from the offshore site during the post-breeding period (September and October), this would affect an estimated 1,513 birds (Table 4.135), increasing to 2,923 birds if displacement is assumed to affect the offshore site and a 2 km buffer (Table 4.137). For the purposes of this assessment, it was assumed that 2% of all guillemots displaced from the offshore site during the post-breeding period (up to 58 birds) would die as a result. This corresponds to approximately 0.03% of the regional population in the post-breeding period (185,973 birds). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be lower than this. It was assumed that the remaining 2,865 guillemots displaced from the offshore site and 2 km buffer would move to alternative foraging areas in the post-breeding period.

Table 4.135 Estimated number of guillemots at risk of mortality following displacement from offshore site in post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	8	19	38	76	113	151	189	227	265	303	340	378
	20%	15	38	76	151	227	303	378	454	530	605	681	757
	30%	23	57	113	227	340	454	567	681	794	908	1,021	1,135
	40%	**30**	76	151	303	454	605	757	908	1,059	1,211	1,362	1,513
	50%	38	95	189	378	567	757	946	1,135	1,324	1,513	1,702	1,892
	60%	45	113	227	454	681	908	1,135	1,362	1,589	1,816	2,043	2,270
	70%	53	132	265	530	794	1,059	1,324	1,589	1,854	2,118	2,383	2,648
	80%	61	151	303	605	908	1,211	1,513	1,816	2,118	2,421	2,724	3,026
	90%	68	170	340	681	1,021	1,362	1,702	2,043	2,383	2,724	3,064	3,405
	100%	76	189	378	757	1,135	1,513	1,892	2,270	2,648	3,026	3,405	3,783
Three-year mean peak of 3,783 guillemots in the offshore site in post-breeding period													
Regional population in post-breeding season = 185,973 birds (Revised from Mitchell *et al.,* 2004)													

Table 4.136 Estimated number of guillemots at risk of mortality following displacement from offshore site plus 1 km buffer in post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	11	27	54	107	161	214	268	321	375	428	482	536
	20%	21	54	107	214	321	428	536	643	750	857	964	1,071
	30%	32	80	161	321	482	643	803	964	1,125	1,285	1,446	1,607
	40%	**43**	107	214	428	643	857	1,071	1,285	1,500	1,714	1,928	2,142
	50%	54	134	268	536	803	1,071	1,339	1,607	1,875	2,142	2,410	2,678
	60%	64	161	321	643	964	1,285	1,607	1,928	2,250	2,571	2,892	3,214
	70%	75	187	375	750	1,125	1,500	1,875	2,250	2,624	2,999	3,374	3,749
	80%	86	214	428	857	1,285	1,714	2,142	2,571	2,999	3,428	3,856	4,285
	90%	96	241	482	964	1,446	1,928	2,410	2,892	3,374	3,856	4,338	4,820
	100%	107	268	536	1,071	1,607	2,142	2,678	3,214	3,749	4,285	4,820	5,356
Three-year mean peak of 5,356 guillemots in offshore site & 1 km buffer in post-breeding period													
Regional population in post-breeding season = 185,973 birds (Revised from Mitchell *et al.,* 2004)													

Table 4.137 Estimated number of guillemots at risk of mortality following displacement from offshore site plus 2 km buffer in post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	15	37	73	146	219	292	365	438	512	585	658	731
	20%	29	73	146	292	438	585	731	877	1,023	1,169	1,315	1,462
	30%	44	110	219	438	658	877	1,096	1,315	1,535	1,754	1,973	2,192
	40%	**58**	146	292	585	877	1,169	1,462	1,754	2,046	2,339	2,631	2,923
	50%	73	183	365	731	1,096	1,462	1,827	2,192	2,558	2,923	3,289	3,654
	60%	88	219	438	877	1,315	1,754	2,192	2,631	3,069	3,508	3,946	4,385
	70%	102	256	512	1,023	1,535	2,046	2,558	3,069	3,581	4,092	4,604	5,116
	80%	117	292	585	1,169	1,754	2,339	2,923	3,508	4,092	4,677	5,262	5,846
	90%	132	329	658	1,315	1,973	2,631	3,289	3,946	4,604	5,262	5,919	6,577
	100%	146	365	731	1,462	2,192	2,923	3,654	4,385	5,116	5,846	6,577	7,308
Three-year mean peak of 7,308 guillemots in offshore site & 2 km buffer in post-breeding period													
Regional population in post-breeding season = 185,973 birds (Revised from Mitchell *et al.,* 2004)													

Comparing the distribution of guillemots in the study area from Years 1 to 3 at this time of year shows that there is considerable variation between years. In September and October of

Year 1, guillemots were widespread throughout the offshore site and buffer area, at mostly high densities, apart from in the south-east of the buffer area, where densities were lower (Figure 4.100). In Year 2, guillemots were less widespread, with highest densities recorded in the north, west and south of the offshore site and buffer area. Birds were absent or present only in low densities in the eastern half of the offshore site and in the south-east of the buffer area at this time (Figure 4.101).

In the same period of Year 3, the main concentrations of guillemots were in the south-west of the offshore site and buffer area, with lower densities in the north-east of the buffer area at this time (Figure 4.102). These changes in distribution in the post-breeding period between years are most likely influenced by changes in the distribution of prey, and demonstrate that guillemots are not regularly relying on the offshore site exclusively at this time of year.

The displacement impact in the post-breeding season is therefore 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. 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

Assuming 40% of all guillemots were displaced from the offshore site during the non-breeding period (November to March), this would affect an estimated 424 birds (Table 4.138), increasing to 816 birds if displacement is assumed to affect the offshore site and a 2 km buffer (Table 4.140). For the purposes of this assessment, it was assumed that 2% of all guillemots displaced from the offshore site and 2 km buffer during the non-breeding period (up to 16 birds) would die as a result. This corresponds to approximately <0.01% of the regional population in the non-breeding period (521,000 birds). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be lower than this. It was assumed that the remaining displaced guillemots would move to alternative foraging areas in the non-breeding period.

Although there was less variation in guillemot distribution in the study area between Years 1 and 3 at this time of year, analysis shows that fewer birds are present in the study area compared to other seasons. Although guillemots were widespread between November and March throughout the study area in all three years, densities in the offshore site at this time were generally lower than in the surrounding buffer area. This suggests that guillemots are not regularly relying on the offshore site at this time of year, and that any displacement effects from this area will be small.

The displacement impact in the non-breeding season is therefore 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. 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.

Table 4.138 Estimated number of guillemots at risk of mortality following displacement from offshore site in non-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	2	5	11	21	32	42	53	64	74	85	95	106
	20%	4	11	21	42	64	85	106	127	148	170	191	212
	30%	6	16	32	64	95	127	159	191	223	254	286	318
	40%	**8**	21	42	85	127	170	212	254	297	339	382	424
	50%	11	27	53	106	159	212	265	318	371	424	477	530
	60%	13	32	64	127	191	254	318	382	445	509	572	636
	70%	15	37	74	148	223	297	371	445	519	594	668	742
	80%	17	42	85	170	254	339	424	509	594	678	763	848
	90%	19	48	95	191	286	382	477	572	668	763	859	954
	100%	21	53	106	212	318	424	530	636	742	848	954	1,060
Three-year mean peak of 1,060 guillemots in the offshore site in non-breeding period													
Regional population in the non-breeding season = 521,000 birds (Skov *et al*., 1995)													

Table 4.139 Estimated number of guillemots at risk of mortality following displacement from offshore site plus 1 km buffer in non-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	3	8	15	31	46	62	77	93	108	124	139	154
	20%	6	15	31	62	93	124	154	185	216	247	278	309
	30%	9	23	46	93	139	185	232	278	324	371	417	463
	40%	**12**	31	62	124	185	247	309	371	432	494	556	618
	50%	15	39	77	154	232	309	386	463	540	618	695	772
	60%	19	46	93	185	278	371	463	556	648	741	834	926
	70%	22	54	108	216	324	432	540	648	757	865	973	1,081
	80%	25	62	124	247	371	494	618	741	865	988	1,112	1,235
	90%	28	69	139	278	417	556	695	834	973	1,112	1,251	1,390
	100%	31	77	154	309	463	618	772	926	1,081	1,235	1,390	1,544
Three-year mean peak of 1,544 guillemots in offshore site & 1 km buffer in non-breeding period													
Regional population in the non-breeding season = 521,000 birds (Skov *et al*., 1995)													

Table 4.140 Estimated number of guillemots at risk of mortality following displacement from offshore site plus 2 km buffer in non -breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	4	10	20	41	61	82	102	122	143	163	184	204
	20%	8	20	41	82	122	163	204	245	286	326	367	408
	30%	12	31	61	122	184	245	306	367	428	490	551	612
	40%	**16**	41	82	163	245	326	408	490	571	653	734	816
	50%	20	51	102	204	306	408	510	612	714	816	918	1,020
	60%	24	61	122	245	367	490	612	734	857	979	1,102	1,224
	70%	29	71	143	286	428	571	714	857	1,000	1,142	1,285	1,428
	80%	33	82	163	326	490	653	816	979	1,142	1,306	1,469	1,632
	90%	37	92	184	367	551	734	918	1,102	1,285	1,469	1,652	1,836
	100%	41	102	204	408	612	816	1,020	1,224	1,428	1,632	1,836	2,040
Three-year mean peak of 2,040 guillemots in offshore site & 2 km buffer in non-breeding period													
Regional population in the non-breeding season = 521,000 birds (Skov *et al*., 1995)													

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 3.7 and Table 3.8. 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 3.7). 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 3.8). 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 3.7). 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 3.8). 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 3.7). 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 3.8). 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. 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

CRM estimated the number of potential guillemot collisions per season for four wind turbine designs scenarios (Table 3.6). The minimum height for the turbine blades above the sea surface for all four options is 27.5 m at mean sea level (MSL).

There is no specific SNH guidance on avoidance rates for seabirds, and therefore their default value of 98.0% is presented in this assessment for comparison purposes

Highest estimated collisions in the "colony attendance" and "chicks at sea" periods of the breeding season combined (April to August) were for Option 1 (90 x 5 MW turbines). Using a 98% avoidance rate, the number of predicted guillemot collisions in the breeding period was 0.7 birds (Table 4.141). This equates to less than 0.01% of the regional SPA population (171,822 birds) in the breeding period.

Table 4.141 Number of estimated guillemot collisions using avoidance rate of 98% for four wind farm options

	Option 1 (90x5MW)	Option 2 (75x6MW)	Option 3 (73x6.15MW)	Option 4 (73x6.15MW)
Collisions in breeding season (April to August), all ages	0.7	0.6	0.5	0.5
Collisions in post- and non-breeding period (September to March), all ages	0.3	0.3	0.3	0.3
Total collisions per year, all ages	1.1	0.9	0.8	0.8

In the post-breeding and non-breeding periods combined (October to March) and using a 98% avoidance rate, the predicted number of guillemot collisions for the worst case design option (Option 1) was 0.3 birds (Table 4.141). This equates to less than 0.01% of the regional population in the non-breeding period (521,000 birds) (Skov *et al.,* 1995).

It is concluded that for the most adverse design (Option 1: 90 x 5 MW turbines), collision mortality for guillemot is an effect of negligible magnitude, that is temporally long-term and reversible.

It is further concluded that for the wind farm designs examined here, the effects of collision mortality on guillemots from the regional population in the breeding, post-breeding and non-breeding periods are ***not significant*** under the EIA Regulations.

Decommissioning Phase

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.

Summary of combined effects

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 4.142). It is concluded that the overall impact on the regional population of guillemots in the breeding season is **not significant** under the EIA regulations.

Table 4.142 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***

Similarly, it is judged that the magnitude of the effects on the regional population of guillemots in the post-breeding and non-breeding-periods is negligible (Table 4.143). It is concluded that the overall impact on the regional population of guillemots in the post-breeding and non-breeding-periods is ***not significant*** under the EIA regulations.

Table 4.143 Summary of effects on the regional population of guillemot in the post-breeding and 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***

4.4.21.5 Cumulative Impact Assessment

Displacement

The potential cumulative displacement risk to guillemots from Neart na Gaoithe and other plans or projects is higher than for Neart na Gaoithe alone.

Projects identified during consultation and the undertaking of the EIA for which there is a potential for a cumulative displacement impact are:

- Inch Cape offshore wind farm;
- Seagreen Project Alpha offshore wind farm; and
- Seagreen Project Bravo offshore wind farm.

Evidence from existing offshore wind farms support the potential for a moderate level of displacement behaviour (e.g. Leopold et al. 2011) and a predicted 40% displacement effect has been used in this assessment.

Data presented in the Seagreen HRA (Seagreen 2012) report peak numbers of guillemots in the Seagreen Project Alpha site boundary during the colony attendance part of the breeding period. Peak estimated numbers within the site boundary occurred in June with 5,502 birds in 2010, and 10,811 birds in 2011. This provides a peak mean across two years, of 8,156 guillemots during the breeding period within the Seagreen Project Alpha boundary.

Assuming that 30% of the population present are immature birds (Wanless *et al.,* 1998), then of the peak mean of 8,156 individuals at project Alpha, 5,709 may be breeding adults.

Assuming 40% of all guillemots were displaced from the Seagreen project Alpha and a 2 km buffer during the breeding season, this would affect an estimated 2,284 breeding adults (Table 4.144).

Assuming that there is the potential for up to 10% rate of mortality during the breeding period then up to 228 breeding adult guillemots may be impacted by displacement from Seagreen project Alpha and a 2 km buffer (Table 4.144).

Table 4.144 Estimated number of adult guillemots predicted to be at risk of mortality following displacement from Seagreen Project Alpha plus 2 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	11	29	57	114	171	228	285	343	400	457	514	571
	20%	23	57	114	228	343	457	571	685	799	913	1,028	1,142
	30%	34	86	171	343	514	685	856	1,028	1,199	1,370	1,541	1,713
	40%	46	114	**228**	457	685	913	1,142	1370	1,599	1827	2,055	2,284
	50%	57	143	285	571	856	1,142	1,427	1,713	1,998	2,284	2,569	2,855
	60%	69	171	343	685	1,028	1,370	1,713	2,055	2,398	2,740	3,083	3,425
	70%	80	200	400	799	1,199	1,599	1,998	2,398	2,797	3,197	3,597	3,996
	80%	91	228	457	913	1,370	1,827	2,284	2,740	3,197	3,654	4,110	4,567
	90%	103	257	514	1,028	1,541	2,055	2,569	3,083	3,597	4,110	4,624	5,138
	100%	114	285	571	1,142	1,713	2,284	2,855	3,425	3,996	4,567	5,138	5,709
Two-year mean peak of 5,709 adult guillemots in Seagreen Project Alpha & 2 km buffer in "at-colony" period													
SPA breeding population within mean max foraging range (84.2 km) = 171,822 birds (JNCC 2013, SMP 2013)													

In project Bravo the peak estimated number of guillemots also occurred during June 2011 with an estimated 10,569 individuals (no peak numbers for 2010 are presented in the ES and so calculating a peak mean is not possible for Project Bravo) (Seagreen 2012). Assuming that 30% of the population present are immature birds (Wanless *et al.,* 1998), then of the 10,569 individuals at project Bravo, 7,398 birds may be breeding adults.

Assuming 40% of all guillemots were displaced from the Seagreen project Bravo and a 2 km buffer during the breeding season, this would affect an estimated 2,959 breeding adults (Table 4.145).

Assuming that there is the potential for up to 10% rate of mortality during the breeding period then up to 296 breeding adult guillemots may be impacted by displacement from Seagreen project Bravo and a 2 km buffer (Table 4.145).

Table 4.145 Estimated number of adult guillemots predicted to be at risk of mortality following displacement from Seagreen Project Bravo plus 2 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	15	37	74	148	222	296	370	444	518	592	666	740
	20%	30	74	148	296	444	592	740	888	1,036	1,184	1,332	1,480
	30%	44	111	222	444	666	888	1,110	1,332	1,554	1,776	1,997	2,219
	40%	59	148	**296**	592	888	1,184	1,480	1,776	2,071	2,367	2,663	2,959
	50%	74	185	370	740	1,110	1,480	1,850	2,219	2,589	2,959	3,329	3,699
	60%	89	222	444	888	1,332	1,776	2,219	2,663	3,107	3,551	3,995	4,439
	70%	104	259	518	1,036	1,554	2,071	2,589	3,107	3,625	4,143	4,661	5,179
	80%	118	296	592	1,184	1,776	2,367	2,959	3,551	4,143	4,735	5,327	5,918
	90%	133	333	666	1,332	1,997	2,663	3,329	3,995	4,661	5,327	5,992	6,658
	100%	148	370	740	1,480	2,219	2,959	3,699	4,439	5,179	5,918	6,658	7,398
One year peak of 7,398 adult guillemots in Seagreen Project Bravo & 2 km buffer in "at-colony" period													
SPA breeding population within mean max foraging range (84.2 km) = 171,822 birds (JNCC 2013, SMP 2013)													

Based on the above, the predicted cumulative displacement impacts on adult guillemots from Neart na Gaoithe and Seagreen projects Alpha and Bravo and a 2 km buffer in the breeding period involve a predicted 6,009 adult birds, with 10% mortality of 601 adults. This corresponds to approximately 0.3% of the regional breeding SPA population within mean maximum foraging range of the offshore site (171,822 birds). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

The predicted number of adult guillemots displaced from the Inch Cape site during the breeding season was 1,411 birds (ICOL, 2013). Assuming 40% of all guillemots were displaced from the Inch Cape site during the breeding season, this would affect an estimated 564 breeding adults (Table 4.146).

Assuming that there is the potential for up to 10% rate of mortality during the colony attendance part of the breeding period then up to 56 breeding adult guillemots may be impacted by displacement from Inch Cape (Table 4.146).

Based on the above, the predicted cumulative displacement impacts on adult guillemots from Neart na Gaoithe and a 2 km buffer and the Inch Cape site in the breeding period involve a predicted 1,330 adult birds, with 10% mortality of 165 adults. This corresponds to approximately 0.01% of the regional breeding SPA population within mean maximum foraging range of the offshore site (171,822 birds). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

Table 4.146 Estimated number of adult guillemots predicted to be at risk of mortality following displacement from Inch Cape site in the colony attendance part of the breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	3	7	14	28	42	56	71	85	99	113	127	141
	20%	6	14	28	56	85	113	141	169	198	226	254	282
	30%	8	21	42	85	127	169	212	254	296	339	381	423
	40%	11	28	**56**	113	169	226	282	339	395	452	508	564
	50%	14	35	71	141	212	282	353	423	494	564	635	706
	60%	17	42	85	169	254	339	423	508	593	677	762	847
	70%	20	49	99	198	296	395	494	593	691	790	889	988
	80%	23	56	113	226	339	452	564	677	790	903	1,016	1,129
	90%	25	63	127	254	381	508	635	762	889	1,016	1,143	1,270
	100%	28	71	141	282	423	564	706	847	988	1,129	1,270	1,411
One year peak of 11,137 adult guillemots in Inch Cape site in “at-colony” period													
SPA breeding population within mean max foraging range (84.2 km) = 171,822 birds (JNCC 2013, SMP 2013)													

Based on the predicted numbers reported for the four proposed development areas a total of 6,573 adult guillemots may be displaced from the development sites and a 2 km buffer during the colony attendance part of the breeding period, with 10% mortality of 656 adult birds. The 10% mortality corresponds to approximately 0.4% of the regional breeding SPA population within mean maximum foraging range of the offshore site (171,822 birds). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

This impact is categorised as having negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative effects of displacement on the regional guillemot population in the colony-attendance part of the breeding season are ***not significant*** under the EIA Regulations.

A separate CIA for displacement has not been undertaken for the regional guillemot population in the “chicks-at-sea” part of the breeding-period (July and August), as data was not available for these months for the Seagreen or Inch Cape projects. The predicted displacement of guillemots due to Neart na Gaoithe wind farm in the “chicks-at-sea” part of the breeding period was similar to the colony attendance part of the breeding season, and so it was assumed that any cumulative displacement impact for the regional guillemot population in the “chicks-at-sea” part of the breeding-period would be similar to that of the colony attendance period. This impact was categorised as having negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative effects of displacement on the regional guillemot population in the “chicks-at-sea” part of the breeding season (July and August) are ***not significant*** under the EIA Regulations.

A separate CIA for displacement has not been undertaken for the regional guillemot population in the post-breeding (September and October) or non-breeding periods (November to March), as data was not available for these months for the Seagreen or Inch Cape projects. The predicted displacement of guillemots due to Neart na Gaoithe wind farm in the post-breeding and non-breeding periods is negligible and the sensitivity of the population to displacement at this time of year is considered to be negligible. Therefore it is not plausible that the Neart na Gaoithe Development could contribute to a significant cumulative impact for the regional populations during these periods.

Collision mortality

No significant impacts were predicted to arise from collision mortality caused by Neart na Gaoithe for guillemots in the breeding, post-breeding or non-breeding periods. This was based on the very low numbers of bird collisions predicted by collision risk modelling, indicating very low levels of collision mortality for this species. It is therefore not plausible that Neart na Gaoithe could contribute to a significant cumulative collision mortality impact for the regional population of guillemots in the breeding, post-breeding or non-breeding periods. As a result, no further cumulative impact assessment for collision mortality was undertaken for this species.

4.4.21.6 Mitigation measures

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

4.4.22 Razorbill *Alca torda*

4.4.22.1 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 offshore site and 8 km buffer 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).

4.4.22.2 Offshore site and 8 km buffer area

Razorbill was one of the most frequently recorded seabirds on surveys in the Neart na Gaoithe study area during the baseline surveys, with a total of 3,980 birds recorded in Year 1, 3,323 birds in Year 2 and 1,915 birds in Year 3 (raw numbers, all sea states). The majority of guillemots were recorded in the buffer area (Table 4.2).

During the Year 1 “at-colony” part of the breeding season (April to June), the mean estimated number of razorbills in the offshore site was 30 birds, with a peak of 44 birds in June (Table 4.147). In the same period of Year 2, the mean estimated number of razorbills in the offshore site was 173 birds, with a peak of 364 birds in May. In Year 3, the mean estimated number of razorbills during the “at-colony” period was 116 birds, with a peak of 227 birds in May.

During the Year 1 “chicks-at-sea” period (July and August), the mean estimated number of razorbills in the offshore site was 768 birds, with a peak of 1,529 birds in August (Table 4.147) In the same period of Year 2, the mean estimated number of razorbills in the offshore site was 223 birds, with a peak of 367 birds in July. In Year 3, the mean estimated number of razorbills during the “chicks-at-sea” period was 840 birds, with a peak of 1,412 birds in August.

In the Year 1 post-breeding period (September and October), the mean estimated number of razorbills in the offshore site was 1,689 birds, with a peak of 2,655 birds in October (Table 4.147). In the same period of Year 2, the mean estimated number of razorbills in the offshore site was 517 birds, with a peak of 852 birds in October. In Year 3, the mean estimated number of razorbills during the post-breeding period was 327 birds, with a peak of 571 birds in September.

In the Year 1 non-breeding period (November to March), the mean estimated number of razorbills in the offshore site was 96 birds, with a peak of 274 birds in November (Table 4.147). In the same period of Year 2, the mean estimated number of razorbills in the offshore site was 105 birds, with a peak of 327 birds in December, although there was no November survey. In Year 3, the mean estimated number of razorbills during the non-breeding period was 46 birds, with a peak of 79 birds in November, although there was no December survey.

Table 4.147 Estimated numbers of razorbills in the offshore site (and 1, 2 & 8 km buffer) in Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 8 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	268	171	418	7	274	328	368	971
Yr1 Dec	142	74	275	0	142	168	227	597
Yr1 Jan	0	0	0	7	7	7	7	29
Yr1 Feb	15	7	32	0	15	46	46	362
Yr1 Mar	37	22	61	7	43	158	177	910
Yr1 Apr	0	0	0	20	20	20	61	437
Yr1 May	0	0	0	27	27	41	41	308
Yr1 Jun	44	20	98	0	44	44	65	282
Yr1 Jul	0	0	0	7	7	7	7	189
Yr1 Aug	1,529	1,184	1,975	0	1,529	2,388	2,919	5,972
Yr1 Sep	723	532	983	0	723	1,046	1,252	3,409
Yr1 Oct	2,655	1,479	4,765	0	2,655	3,316	4,664	19,985
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	320	194	529	7	327	632	632	906
Yr2 Jan	0	0	0	0	0	0	0	199
Yr2 Feb	42	22	78	0	42	139	174	872
Yr2 Mar	50	24	104	0	50	50	50	543
Yr2 Apr	107	46	250	14	120	187	214	655
Yr2 May	323	174	602	40	364	410	472	1,466
Yr2 Jun	15	6	36	20	35	42	63	455
Yr2 Jul	367	211	639	0	367	419	590	2,013
Yr2 Aug	78	41	149	0	78	104	143	611
Yr2 Sep	182	107	312	0	182	257	815	3,122
Yr2 Oct	770	590	1,006	81	852	1,785	2,944	14,578
Yr3 Nov	65	39	109	14	79	135	152	597
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	58	23	145	0	58	58	58	272
Yr3 Feb	16	7	34	0	16	55	71	248
Yr3 Mar	31	15	66	0	31	47	109	396
Yr3 Apr	84	43	167	0	84	205	229	579
Yr3 May	200	82	484	27	227	256	341	732
Yr3 Jun	15	7	35	21	36	36	65	186
Yr3 Jul	246	145	417	21	267	458	458	1,374
Yr3 Aug	1,412	903	2,208	0	1,412	2,507	3,388	9,645
Yr3 Sep	571	340	961	0	571	800	1,231	4,879
Yr3 Oct	76	45	128	7	82	82	143	1,085

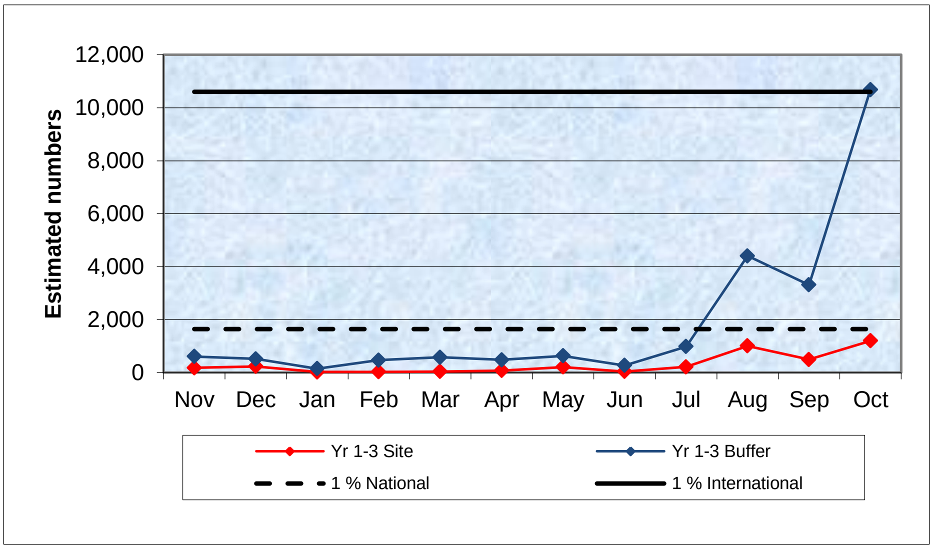


Figure 4.105 Mean monthly estimated numbers of razorbills in the Neart na Gaoithe Development & buffer areas in Years 1 to 3 (Three-year mean)

Estimated mean numbers of razorbills in the offshore site were low between November and July across all three baseline years (Figure 4.105). Numbers increased in August, and peaked in October (1,196 birds, three-year mean). Mean estimated numbers in the buffer zone showed a similar pattern, although estimated numbers between August and October were higher, especially the October peak (10,686 birds, three-year mean). Mean estimated numbers in the buffer zone between August and October exceeded 1% of the national breeding population (1,646 birds) (Mitchell *et al.,* 2004), while the three-year mean estimated number for the buffer zone in October exceeded 1% of the bio-geographic breeding population (10,600 birds) (Mitchell *et al.,* 2004).

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 4.106). Generally fewer razorbills were recorded in the south-west of the buffer area at this time.

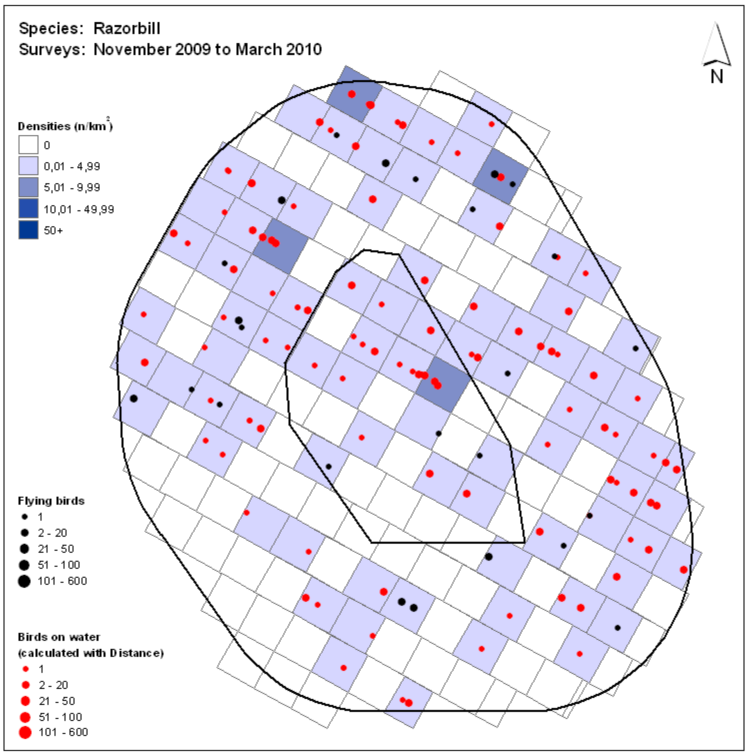


Figure 4.106 Razorbill density between November and March, Year 1

Razorbill distribution over the same period in Year 2 was broadly similar, with mostly low densities of razorbills scattered across 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 4.107).

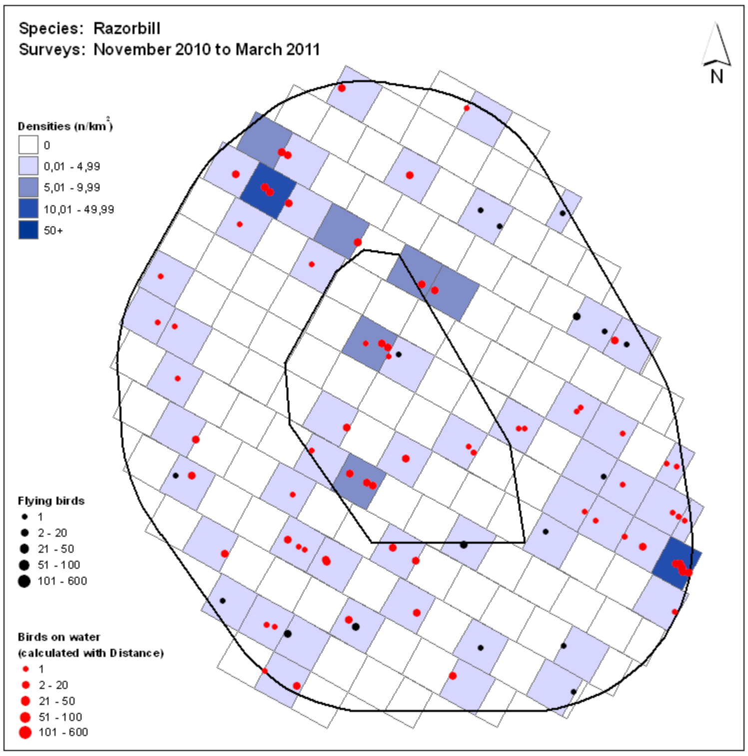


Figure 4.107 Razorbill density between November and March, Year 2

Razorbill distribution over the same period in Year 3 was broadly similar, with low densities of razorbills scattered across the offshore site and the buffer area (Figure 4.108).

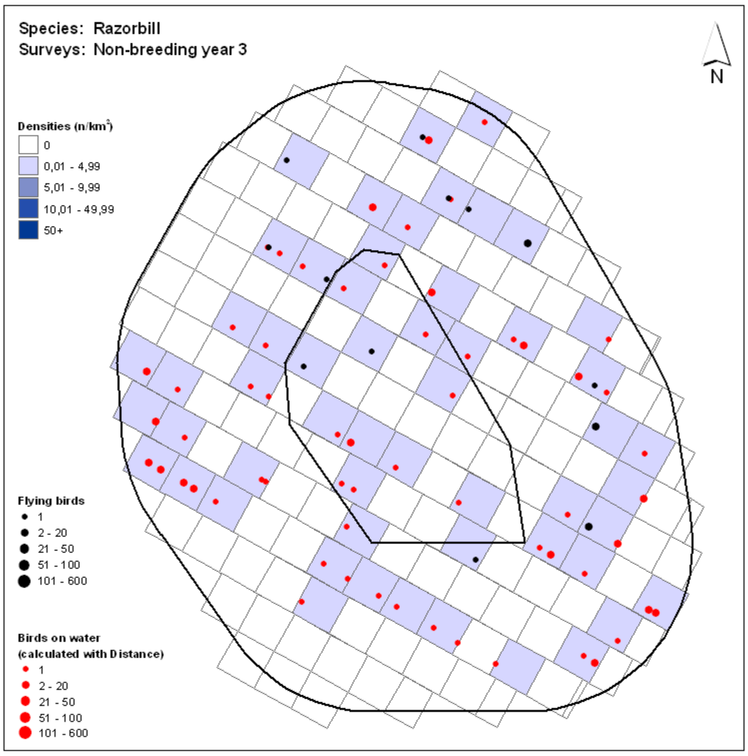


Figure 4.108 Razorbill density between November and March, Year 3

During the Year 1 "at-colony" period of the breeding season, (April to June), razorbills were scattered at low densities across the offshore site (Figure 4.109). In the buffer area, highest densities were recorded in the south-east at this time.

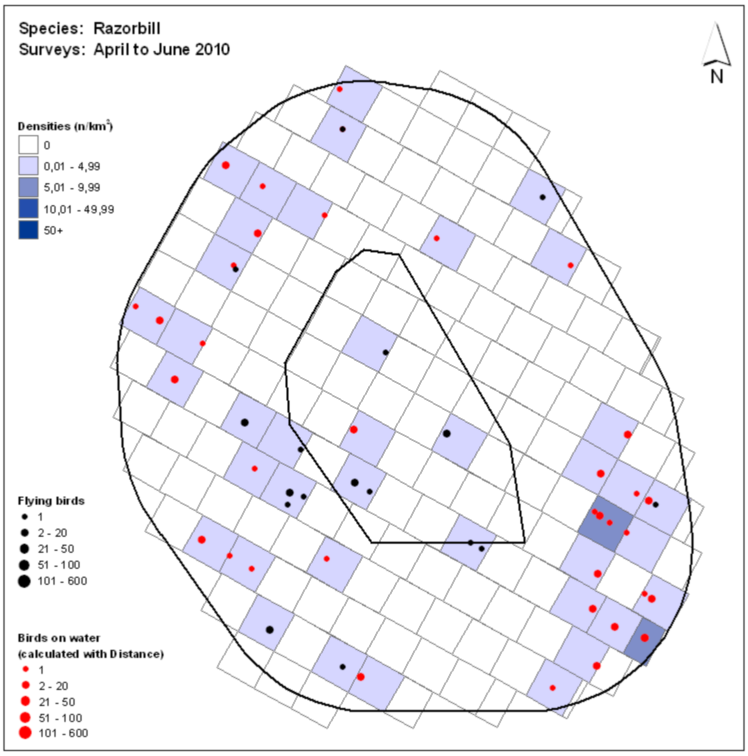


Figure 4.109 Razorbill density between April and June, Year 1

Razorbill distribution between April and June of Year 2 was similar to the same period in Year 1, with low to moderate densities in the offshore site (Figure 4.110). In the buffer area, highest densities were recorded in the south-east at this time.

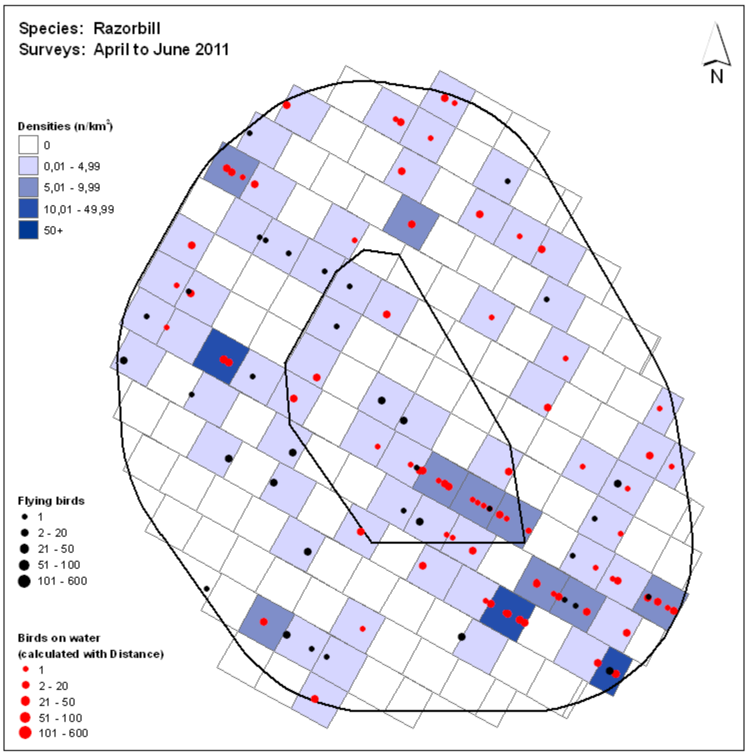


Figure 4.110Razorbill density between April and June, Year 2

Razorbill distribution between April and June of Year 3, was similar to previous years, with low densities in the offshore site and the buffer area (Figure 4.111). Highest densities were recorded in the north-west of the buffer area at this time.

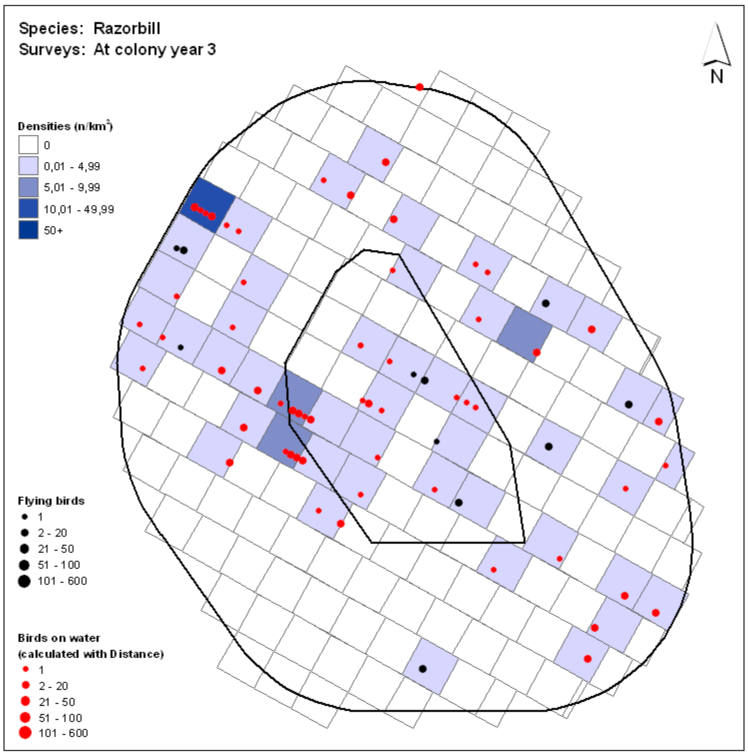


Figure 4.111Razorbill density between April and June, Year 3

Razorbills were more widespread at higher densities during the Year 1 "chicks-at-sea" period of the breeding season (July and August), compared to when birds were at the breeding colonies. Birds occurred throughout the offshore site at mostly moderate to high densities at this time (Figure 4.112). In the buffer area, razorbills were similarly distributed, although birds were absent from parts of the buffer area at this time.

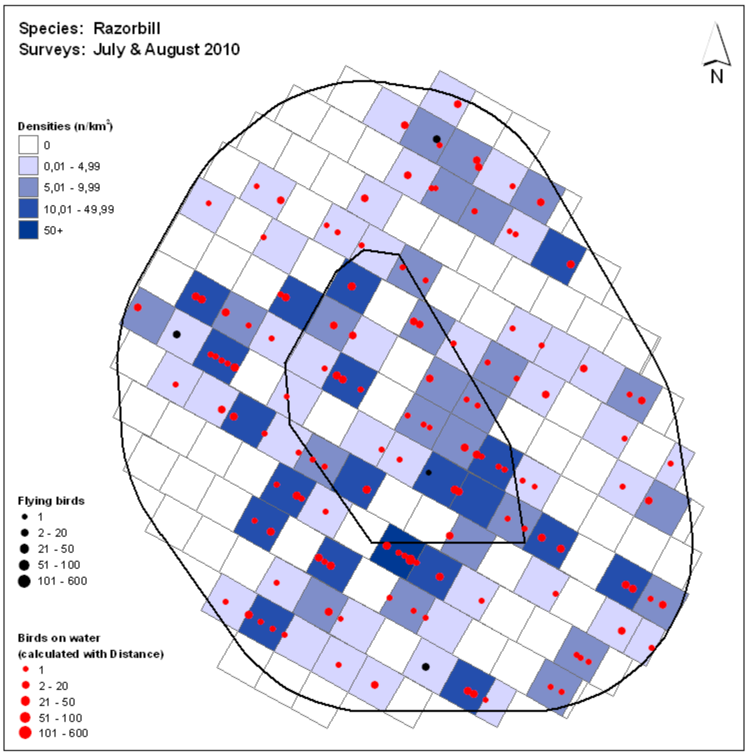


Figure 4.112 Razorbill density in July and August, Year 1

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 and west (Figure 4.113). Fewer razorbills were recorded in the offshore site at this time, with highest densities in the east of the site at this time.

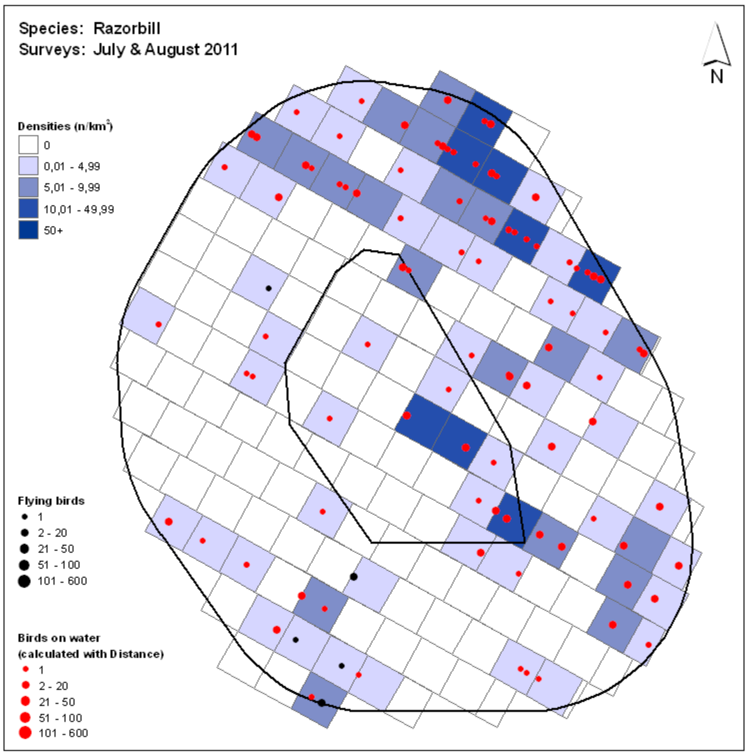


Figure 4.113 Razorbill density in July and August, Year 2

Razorbills were more widespread at moderate to high densities in the offshore site and buffer area in July and August of Year 3, compared to the same period in Year 2 (Figure 4.114). Highest densities of razorbills were recorded in the south-west of the buffer area, with fewer birds in the north and east at this time.

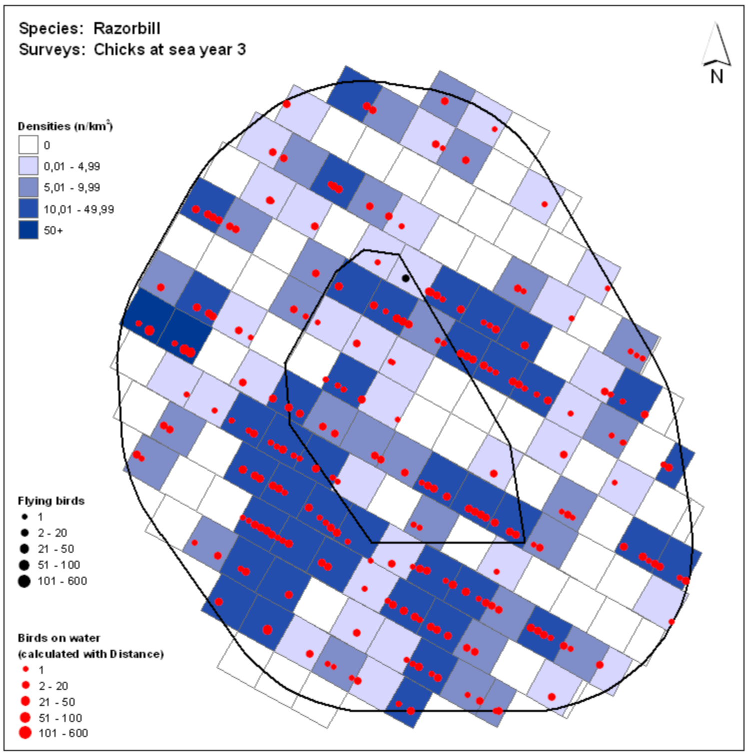


Figure 4.114 Razorbill density in July and August, Year 3

In the Year 1 post-breeding period (September and October), razorbills were widespread at mostly high densities across the offshore site and in the north and west of the buffer area (Figure 4.115). Densities were lower or zero in the south-east of the buffer area at this time.

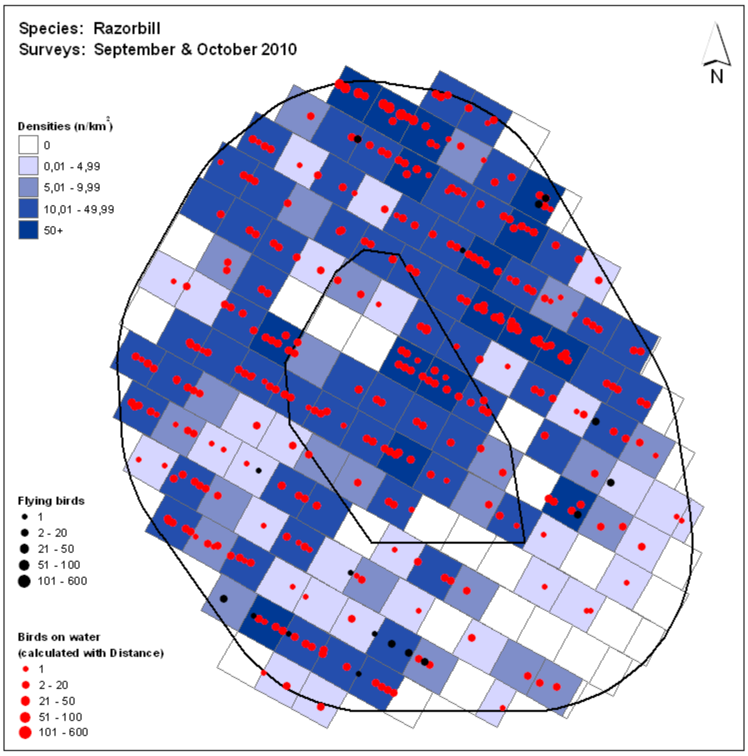


Figure 4.115 Razorbill density in September and October, Year 1

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 smaller areas of high densities in the south and east (Figure 4.116). Razorbill densities in the offshore site at this time were lower than in the same period of Year 1.

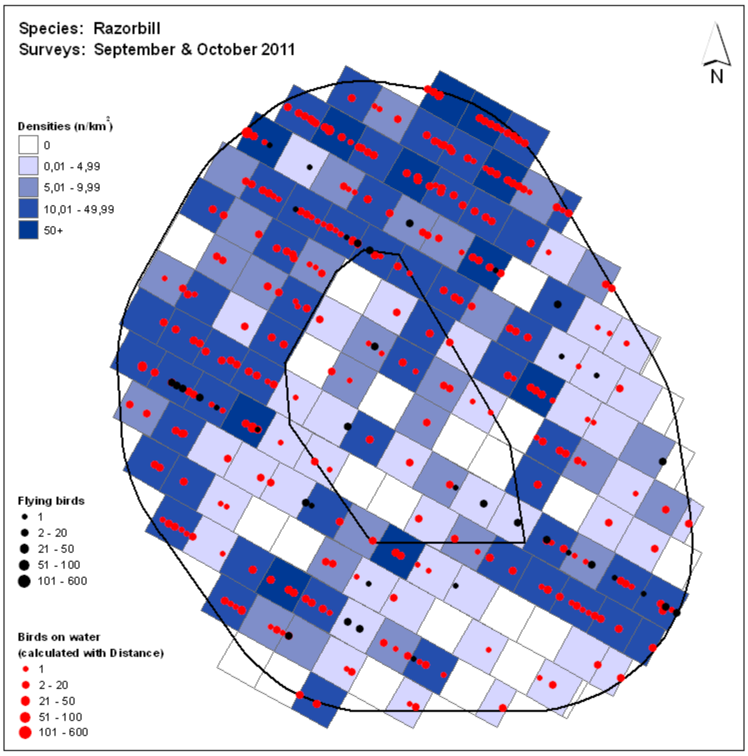


Figure 4.116Razorbill density in September and October, Year 2

In the Year 3 post-breeding period, razorbill density in the offshore site was lower than in the previous two years, with highest densities recorded in the north of the site (Figure 4.117). In the buffer area, there were also fewer high density concentrations of razorbills. Highest densities were recorded in the north of the buffer area, with fewer birds in the south.

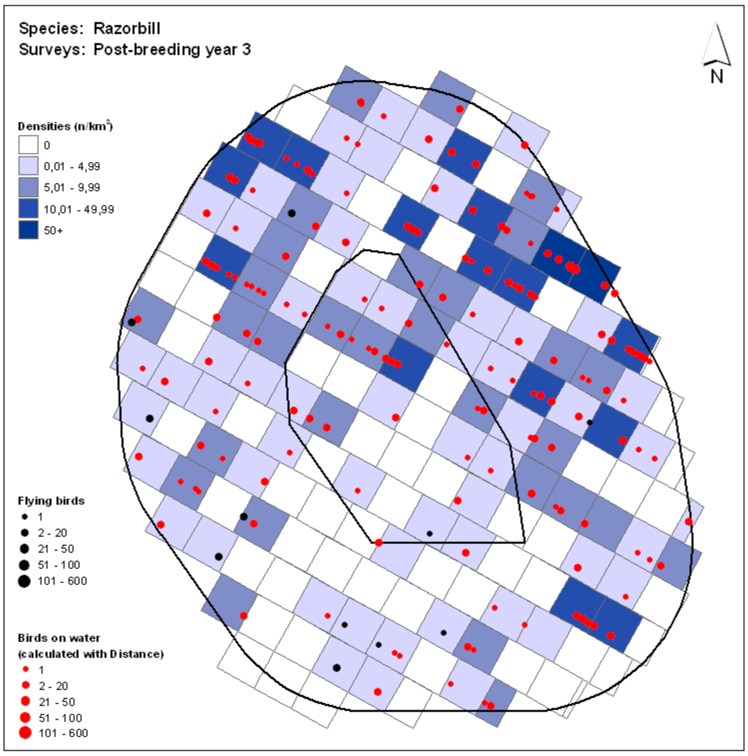
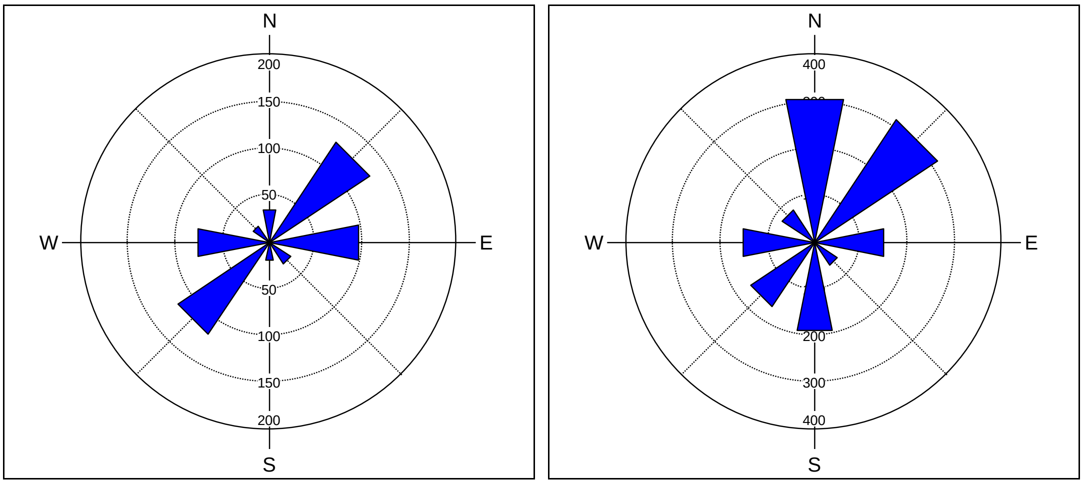


Figure 4.117Razorbill density in September and October, Year 3

A total of 1,949 razorbills were recorded in flight on baseline surveys, with all birds recorded flying below 27.5 m in height. The majority (98.6%) were recorded flying below 7.5 m in height (Table 4.3).

Flight direction was recorded for 522 razorbills in the colony attendance period of the breeding season (April to June), with direction recorded for 1,423 razorbills in the "chicks-at-sea", post-breeding and non-breeding periods combined (July to March) (Figure 4.118).



April to June (n=522 birds) July to March (n=1,423 birds)
Numbers shown on figures are number of birds recorded

Figure 4.118 Flight direction of razorbills in the offshore site and 8 km buffer area in Years 1 to 3

In the breeding season, just under a quarter of all birds recorded were flying north-east (24.5%), with 22.4% flying south-west and 18.6% flying east. In the non-breeding season, just over a fifth of all birds recorded were flying north-east (22.0%) and north (21.9%).

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).

4.4.22.3 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).

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 4.148). These SPAs held 8.7% of the UK breeding population and 1.5% of the biogeographic population at the time of designation (JNCC, 2013). 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).

Table 4.148 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**	**3,704**	**2012**
Fowlsheugh	*62*	*4,576*	*0.8*	*4.6*	*5,260*	*2012*
St Abb's Head to Fast Castle	**31**	**1,407**	**0.2**	**1.4**	**2,406 [3]**	**2008**
Total	**-**	**8,676**	**1.5**	**8.7**	**11,370**	**-**

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).
3. 2008 count for St Abb's Head NNR (birds) (SMP 2013) plus corrected estimate for rest of SPA colony – see text for full explanation.

The full extent of the St Abb's Head to Fast Castle SPA razorbill colony has not been surveyed since 1998, for Seabird 2000 (R Mavor pers. comm.). However, breeding razorbills at St Abb's Head NNR, the major component of the SPA, were counted in 2008. To estimate the size of the remaining SPA population, the percentage decline in razorbill numbers at St Abb's Head NNR between 1998 (2,214 birds) and 2008 (1,687 birds) was calculated (-23.8% decline). This was applied to the population of the remainder of the SPA for 1998 (943 birds) to estimate the likely size of the population for this area in 2008 (719 birds). The two 2008 values were then added to get a total estimate for the whole SPA for 2008 (2,406 birds).

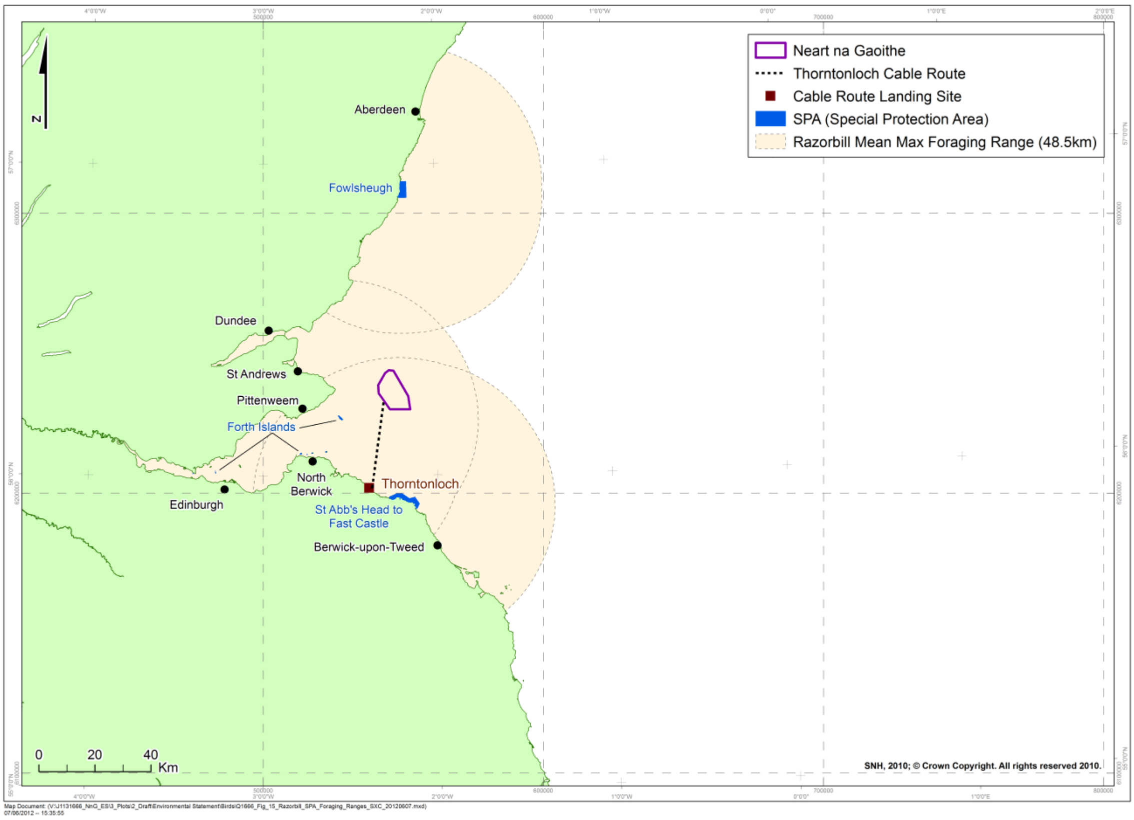


Figure 4.119 Razorbill mean maximum foraging range from breeding SPAs in relation to the Development

Razorbill mean maximum foraging range from breeding SPAs in relation to the offshore site are shown in Figure 4.119.

4.4.22.4 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.

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 "chicks-at-sea" 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).

Populations

Since Seabird 2000, the breeding population of razorbills in Scotland has undergone a period of decline (SMP, 2013) (Table 4.149).

Table 4.149 Recent counts & Seabird 2000 counts at main colonies for breeding razorbills between Peterhead and Blyth

Colony	Distance to site (km)	Seabird 2000 count	Recent count (birds at colony)	Year	Percentage change since Seabird 2000
Fowlsheugh	62	6,425 [2]	5,260 [1]	2012	-18.1%
Isle of May	16	4,114 [1]	3,305 [1]	2012	-19.7%
St Abb's Head NNR	33	2,214 [1]	1,687 [1]	2008	-23.8%
				Mean percentage change	**-20.5%**

Sources: 1 SMP (2013) – Seabird Monitoring Programme Online Database. 2 Mitchell *et al.*, 2004.

Assuming that the sub-set of colonies that have been recently counted is representative, the regional decline since the Seabird 2000 counts amounts to -20.5%. 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 for Peterhead to Blyth derived from Seabird 2000 results (22,646 birds) (Mitchell *et al.,* 2004) was adjusted downwards by -20.5%. On this basis, the regional breeding population was estimated to be 18,004 birds, taking into account recent changes in numbers at the main breeding colonies in the region.

The SPA breeding population within mean maximum foraging range of the offshore site was estimated to be 6,110 birds, based on most recent available counts (Table 4.148).

The size of the post-breeding-period regional razorbill population is estimated to be the same as the regional population between Peterhead and Blyth (18,004 birds).

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).

An unknown proportion of the regional breeding population may remain in the region through the non-breeding period but these are joined by birds from more distant colonies.

Nature conservation importance

The nature conservation importance of razorbill using the offshore site is categorised as high during the breeding season, 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. The three-year mean peak estimated number of razorbills present in the offshore site between April and June (the at-colony period of the breeding season) (201 birds) is 3.3% of the SPA breeding population within mean maximum foraging range of the offshore site (6,110 birds).

Offshore wind farm studies

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%.

One year of post-construction monitoring at Robin Rigg Offshore Wind Farm Data suggested a decline in razorbill numbers during the construction phase, both within the turbine area and across the site as a whole. Numbers appear to have increased again post construction, although further monitoring is required (Walls *et al.,* 2013).

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.

Construction Phase

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.

Operational Phase

Displacement

In order to assess the potential impacts from displacement, the approach presented in the interim guidance note on displacement issued by the JNCC and NE has been followed (JNCC & NE 2012).

Peak estimated numbers of razorbills in the offshore site in the at-colony period of the breeding season (April to June), the chicks-at-sea period (July and August), the post-breeding season (September and October) and the non-breeding season (November to March) for Years 1 to 3 were averaged to get the three-year mean peak per season. This was repeated for a 1 km and 2 km buffer around the offshore site (Table 4.150).

The three-year mean peak estimated number of razorbills was then used to predict the estimated number of birds at potential risk of mortality following displacement in the at-colony period of the breeding season in the offshore site (plus 1 km and 2 km buffers), as recommended in the draft guidance note on displacement (JNCC & NE, 2012).

This was repeated for the chicks-at-sea period of the breeding season, the post-breeding period and the non-breeding period.

Table 4.150 Seasonal three-year mean peak estimated numbers of razorbills in the offshore site (plus 1 & 2 km buffer)

Year	Offshore site			
	At colony	Chicks-at-sea	Post-breeding	Non-breeding
Year 1	44	1,529	2,655	274
Year 2	364	367	852	327
Year 3	227	1,412	571	79
3-year mean peak	212	1,103	1,359	227
Year	Offshore site + 1 km			
	At colony	Chicks-at-sea	Post-breeding	Non-breeding
Year 1	44	2,388	3,316	328
Year 2	410	419	1,785	632
Year 3	256	2,507	800	135
3-year mean peak	237	1,771	1,967	365
Year	Offshore site + 2 km			
	At colony	Chicks-at-sea	Post-breeding	Non-breeding
Year 1	65	2,919	4,664	368
Year 2	472	590	2,944	632
Year 3	341	3,388	1,231	152
3-year mean peak	293	2,299	2,946	384

Likely impacts of displacement

Colony attendance period

There is evidence from existing offshore wind farms indicating that displacement levels for razorbills are likely to be below 50% (Leopold *et al.,* 2011). For this assessment, it was assumed that there will be 40% displacement of razorbills from the offshore site in all seasons. Additional scenarios considering 40% displacement out to a 1 km and a 2 km buffer are also presented.

Assuming 40% of all razorbills were displaced from the offshore site during the “at-colony” part of the breeding season (April to June), this would affect an estimated 85 birds (Table 4.151), increasing to 117 birds if displacement is assumed to affect the offshore site and a 2 km buffer (Table 4.153). Assuming that 30% of the population present are immature birds (Wanless *et al.,* 1998), this would mean 59 breeding adults would be displaced from the offshore site, increasing to 82 breeding adults if displacement is assumed to affect the offshore site and a 2 km buffer.

For the purposes of this assessment, it was assumed that 10% of all razorbills displaced from the offshore site during the breeding season (up to 12 birds) would die as a result. Applying the correction for immature birds being present in the breeding season (Wanless *et al.,* 1998), gives an estimated mortality of up to eight breeding adults. This corresponds to approximately 0.1% of the regional breeding SPA population within mean maximum foraging range of the offshore site (6,110 birds). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

For the remaining 74 breeding adult razorbills displaced from the offshore site and 2 km buffer that survived, there could potentially be a detrimental impact on their breeding success, as a result of having to forage elsewhere. This number corresponds to approximately 1.2% of the regional breeding SPA population within mean maximum foraging range of the offshore site (6,110 birds).

Table 4.151 Estimated number of razorbills at risk of mortality following displacement from offshore site in “at-colony” period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	1	2	4	6	8	11	13	15	17	19	21
	20%	1	2	4	8	13	17	21	25	30	34	38	42
	30%	1	3	6	13	19	25	32	38	45	51	57	64
	40%	2	4	**8**	17	25	34	42	51	59	68	76	85
	50%	2	5	11	21	32	42	53	64	74	85	95	106
	60%	3	6	13	25	38	51	64	76	89	102	114	127
	70%	3	7	15	30	45	59	74	89	104	119	134	148
	80%	3	8	17	34	51	68	85	102	119	136	153	170
	90%	4	10	19	38	57	76	95	114	134	153	172	191
	100%	4	11	21	42	64	85	106	127	148	170	191	212
Three-year mean peak of 212 razorbills in the offshore site in “at-colony” period													
SPA breeding population within mean max foraging range (48.5 km) = 6,110 birds (SMP 2013)													

Table 4.152 Estimated number of razorbills at risk of mortality following displacement from offshore site plus 1 km buffer in "at-colony" period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	1	2	5	7	9	12	14	17	19	21	24
	20%	1	2	5	9	14	19	24	28	33	38	43	47
	30%	1	4	7	14	21	28	36	43	50	57	64	71
	40%	2	5	**9**	19	28	38	47	57	66	76	85	95
	50%	2	6	12	24	36	47	59	71	83	95	107	119
	60%	3	7	14	28	43	57	71	85	100	114	128	142
	70%	3	8	17	33	50	66	83	100	116	133	149	166
	80%	4	9	19	38	57	76	95	114	133	152	171	190
	90%	4	11	21	43	64	85	107	128	149	171	192	213
	100%	5	12	24	47	71	95	119	142	166	190	213	237
Three-year mean peak of 237 razorbills in the offshore site in "at-colony" period													
SPA breeding population within mean max foraging range (48.5 km) = 6,110 birds (SMP 2013)													

Table 4.153 Estimated number of razorbills at risk of mortality following displacement from offshore site plus 2 km buffer in "at-colony" period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	1	3	6	9	12	15	18	21	23	26	29
	20%	1	3	6	12	18	23	29	35	41	47	53	59
	30%	2	4	9	18	26	35	44	53	62	70	79	88
	40%	2	6	**12**	23	35	47	59	70	82	94	105	117
	50%	3	7	15	29	44	59	73	88	103	117	132	147
	60%	4	9	18	35	53	70	88	105	123	141	158	176
	70%	4	10	21	41	62	82	103	123	144	164	185	205
	80%	5	12	23	47	70	94	117	141	164	188	211	234
	90%	5	13	26	53	79	105	132	158	185	211	237	264
	100%	6	15	29	59	88	117	147	176	205	234	264	293
Three-year mean peak of 293 razorbills in the offshore site in "at-colony" period													
SPA breeding population within mean max foraging range (48.5 km) = 6,110 birds (SMP 2013)													

However, it is likely that razorbills can compensate for a moderate amount of displacement by choosing to forage elsewhere. Comparing the distribution of razorbills in the study area from Years 1 to 3 at this time of year shows that there is considerable variation between years. Between April and June of Year 1, razorbills were scattered occasionally at low densities across the offshore site, with low densities also scattered across the buffer area at this time (Figure 4.109). In the same period in Year 2, razorbills were slightly more widespread, with moderate densities recorded in the south-east of the offshore site, and low densities scattered elsewhere in the offshore site (Figure 4.110). Densities in the buffer area were also mostly low at this time, with highest densities in the south-east. In Year 3, low densities were recorded in the offshore site and the buffer area at this time (Figure 4.111). These changes in seasonal distribution between years are most likely influenced by changes in the distribution of prey, and show that razorbills are not regularly relying on the offshore site exclusively at this time of year.

Razorbill GPS tracking data from the Isle of May in June 2010, based on a sample size of 18 individuals (111 foraging trips), found that razorbills showed a strong affinity to coastal regions as well as offshore sand banks, with birds using the Firth of Forth and St Andrews Bay. Mean maximum foraging range from the Isle of May colony was 14 km, with a maximum foraging range of 69 km recorded (Daunt *et al.,* 2011a). The offshore site is within the mean maximum foraging range recorded by this study, as it is approximately 16 km from the Isle of May.

However, analysis of at-sea distributions of razorbills using kernel density estimations found that the offshore site did not overlap to any great extent with the core area used by foraging razorbills (50% kernals). The offshore site was also outside the overall area used by tagged foraging razorbills in 2010 (90% kernels). The core area of use (50% kernels) was estimated to cover an area of 622 km^2, while the overall area of active use (90% kernels) was estimated at 1,947 km^2 (Daunt *et al.,* 2011a).

The 2010 study also examined differences in foraging range between 2010 and previous breeding seasons in 1999 and 2006, and found that there was variation in foraging range between years. The recorded foraging range in 2010 was intermediate across the three study years, with the greatest range recorded in 1999 and the least in 2006.

Other studies have estimated mean maximum foraging range for razorbill as 48.5 km (Thaxter *et al.,* 2012), which suggests that razorbills from the Isle of May may have a more restricted distribution than the broader picture, although this estimate was only based on one breeding season (Daunt *et al.,* 2011a).

The main conclusion that can be drawn from this tagging study is that razorbills are clearly capable of travelling considerable distances during the breeding season (Daunt *et al.,* 2011a). This study also indicates that the offshore site is not a key foraging area for razorbills in the breeding season, although birds from the Isle of May colony do fly through the site on route to their preferred feeding areas.

Based on the above, the impact of displacement on the "at-colony" period 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 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 razorbill population during the colony attendance period of the breeding season are ***not significant*** under the EIA Regulations.

Chicks-at-sea period

Assuming 40% of all razorbills were displaced from the offshore site during the "chicks-at-sea" part of the breeding season (July and August), this would affect an estimated 441 birds (Table 4.154), increasing to 920 birds if displacement is assumed to affect the offshore site and a 2 km buffer (Table 4.156). Assuming that 30% of the population present are immature birds (Wanless *et al.,* 1998), this would mean 309 breeding adults would be displaced from the offshore site, increasing to 644 breeding adults if displacement is assumed to affect the offshore site and a 2 km buffer.

For the purposes of this assessment, it was assumed that 10% of all razorbills displaced from the offshore site during the breeding season (up to 92 birds) would die as a result. Applying the correction for immature birds being present in the breeding season (Wanless *et al.,* 1998), gives an estimated mortality of up to 64 breeding adults. This corresponds to approximately 1.0% of the regional breeding SPA population within mean maximum foraging range of the offshore site (6,110 birds). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

For the remaining 580 breeding adult razorbills displaced from the offshore site and 2 km buffer that survive, there could potentially be a detrimental impact on their breeding success, as a result of having to forage elsewhere. This number corresponds to approximately 9.5% of the regional breeding SPA population within mean maximum foraging range of the offshore site (6,110 birds).

However, for the reasons outlined above, it is likely that razorbills can compensate for a moderate amount of displacement by choosing to forage elsewhere. In addition, as they are no longer tied to the colony, birds are less restricted in where they can forage. Comparing the distribution of razorbills in the study area from Years 1 to 3 at this time of year shows that there is considerable variation between years. In Year 1, razorbills occurred throughout the offshore site at mostly moderate to high densities at this time (Figure 4.112). In the buffer area, razorbills were similarly distributed, although birds were absent from parts of the south-west and north at this time. In July and August of Year 2, razorbills were less widespread in the offshore site, with few area of low density in the north and west of the site. More razorbills were recorded in the east of the offshore site at this time, at moderate to high densities. In the buffer area, highest densities of razorbills were recorded in the north, with fewer birds in the south (Figure 4.113). In July and August of Year 3, razorbills were more widespread at moderate to high densities in the offshore site, compared to the same period in Year 2 (Figure 4.114). These changes in seasonal distribution between years are most likely influenced by changes in the distribution of prey, and show that razorbills are not regularly relying on the offshore site exclusively at this time of year.

Table 4.154 Estimated number of razorbills at risk of mortality following displacement from offshore site in "chicks-at-sea" period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	2	6	11	22	33	44	55	66	77	88	99	110
	20%	4	11	22	44	66	88	110	132	154	176	199	221
	30%	7	17	33	66	99	132	165	199	232	265	298	331
	40%	9	22	**44**	88	132	176	221	265	309	353	397	441
	50%	11	28	55	110	165	221	276	331	386	441	496	552
	60%	13	33	66	132	199	265	331	397	463	529	596	662
	70%	15	39	77	154	232	309	386	463	540	618	695	772
	80%	18	44	88	176	265	353	441	529	618	706	794	882
	90%	20	50	99	199	298	397	496	596	695	794	893	993
	100%	22	55	110	221	331	441	552	662	772	882	993	1,103
Three-year mean peak of 1,103 razorbills in the offshore site in "chicks-at-sea" period													
SPA breeding population within mean max foraging range (48.5 km) = 6,110 birds (SMP 2013)													

Table 4.155 Estimated number of razorbills at risk of mortality following displacement from offshore site plus 1 km buffer in "chicks-at-sea" period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	4	9	18	35	53	71	89	106	124	142	159	177
	20%	7	18	35	71	106	142	177	213	248	283	319	354
	30%	11	27	53	106	159	213	266	319	372	425	478	531
	40%	14	35	**71**	142	213	283	354	425	496	567	638	708
	50%	18	44	89	177	266	354	443	531	620	708	797	886
	60%	21	53	106	213	319	425	531	638	744	850	956	1,063
	70%	25	62	124	248	372	496	620	744	868	992	1,116	1,240
	80%	28	71	142	283	425	567	708	850	992	1,133	1,275	1,417
	90%	32	80	159	319	478	638	797	956	1,116	1,275	1,435	1,594
	100%	35	89	177	354	531	708	886	1,063	1,240	1,417	1,594	1,771
Three-year mean peak of 1,771 razorbills in the offshore site in "chicks-at-sea" period													
SPA breeding population within mean max foraging range (48.5 km) = 6,110 birds (SMP 2013)													

Table 4.156 Estimated number of razorbills at risk of mortality following displacement from offshore site plus 2 km buffer in "chicks-at-sea" period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	5	11	23	46	69	92	115	138	161	184	207	230
	20%	9	23	46	92	138	184	230	276	322	368	414	460
	30%	14	34	69	138	207	276	345	414	483	552	621	690
	40%	18	46	**92**	184	276	368	460	552	644	736	828	920
	50%	23	57	115	230	345	460	575	690	805	920	1,035	1,150
	60%	28	69	138	276	414	552	690	828	966	1,104	1,241	1,379
	70%	32	80	161	322	483	644	805	966	1,127	1,287	1,448	1,609
	80%	37	92	184	368	552	736	920	1,104	1,287	1,471	1,655	1,839
	90%	41	103	207	414	621	828	1,035	1,241	1,448	1,655	1,862	2,069
	100%	46	115	230	460	690	920	1,150	1,379	1,609	1,839	2,069	2,299
Three-year mean peak of 2,299 razorbills in the offshore site in "chicks-at-sea" period													
SPA breeding population within mean max foraging range (48.5 km) = 6,110 birds (SMP 2013)													

Based on the above, the impact of displacement on the "chicks-at-sea" part of the breeding season 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 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 razorbill population during the "chicks-at-sea" period of the breeding season are ***not significant*** under the EIA Regulations.

Post-breeding period

Assuming 40% of all razorbills were displaced from the offshore site during the post-breeding period (September and October), this would affect an estimated 544 birds (Table 4.157), increasing to 1,178 birds if displacement is assumed to affect the offshore site and a 2 km buffer (Table 4.159). For the purposes of this assessment, it was assumed that 2% of all razorbills displaced from the offshore site during the post-breeding period (up to 24 birds) would die as a result. This corresponds to approximately 0.1% of the regional population in the post-breeding period (18,004 birds). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be lower than this. It was concluded that the remaining razorbills displaced from the offshore site and 2 km buffer would move to alternative foraging areas in the post-breeding period.

Comparing the distribution of razorbills in the study area from Years 1 to 3 at this time of year shows that there is considerable variation between years. In September and October of Year 1, razorbills 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 recorded in September and October (Figure 4.115). In Year 2, razorbill densities in the offshore site were lower than in the same period of Year 1. Highest densities of razorbills were recorded in the north and west of the buffer area, with patches of higher densities in the south and east (Figure 4.116). In the Year 3 post-breeding period, razorbill density in the offshore site was lower than in the previous two years, with highest densities recorded in the north of the site (Figure 4.117). These changes in distribution in the post-breeding period between years are most likely influenced by changes in the distribution of prey, and demonstrate that razorbills are not regularly relying on the offshore site exclusively at this time of year.

Table 4.157 Estimated number of razorbills at risk of mortality following displacement from offshore site in post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	3	7	14	27	41	54	68	82	95	109	122	136
	20%	5	14	27	54	82	109	136	163	190	217	245	272
	30%	8	20	41	82	122	163	204	245	285	326	367	408
	40%	**11**	27	54	109	163	217	272	326	381	435	489	544
	50%	14	34	68	136	204	272	340	408	476	544	612	680
	60%	16	41	82	163	245	326	408	489	571	652	734	815
	70%	19	48	95	190	285	381	476	571	666	761	856	951
	80%	22	54	109	217	326	435	544	652	761	870	978	1,087
	90%	24	61	122	245	367	489	612	734	856	978	1,101	1,223
	100%	27	68	136	272	408	544	680	815	951	1,087	1,223	1,359
Three-year mean peak of 1,359 razorbills in the offshore site in post-breeding period													
Regional population in post-breeding season = 18,004 birds (revised from Mitchell *et al.* 2004)													

Table 4.158 Estimated number of razorbills at risk of mortality following displacement from offshore site plus 1 km buffer in post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	4	10	20	39	59	79	98	118	138	157	177	197
	20%	8	20	39	79	118	157	197	236	275	315	354	393
	30%	12	30	59	118	177	236	295	354	413	472	531	590
	40%	**16**	39	79	157	236	315	393	472	551	629	708	787
	50%	20	49	98	197	295	393	492	590	688	787	885	984
	60%	24	59	118	236	354	472	590	708	826	944	1,062	1,180
	70%	28	69	138	275	413	551	688	826	964	1,102	1,239	1,377
	80%	31	79	157	315	472	629	787	944	1,102	1,259	1,416	1,574
	90%	35	89	177	354	531	708	885	1,062	1,239	1,416	1,593	1,770
	100%	39	98	197	393	590	787	984	1,180	1,377	1,574	1,770	1,967
Three-year mean peak of 1,967 razorbills in the offshore site in post-breeding period													
Regional population in post-breeding season = 18,004 birds (revised from Mitchell *et al.* 2004)													

Table 4.159 Estimated number of razorbills at risk of mortality following displacement from offshore site plus 2 km buffer in post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	6	15	29	59	88	118	147	177	206	236	265	295
	20%	12	29	59	118	177	236	295	354	412	471	530	589
	30%	18	44	88	177	265	354	442	530	619	707	795	884
	40%	**24**	59	118	236	354	471	589	707	825	943	1,061	1,178
	50%	29	74	147	295	442	589	737	884	1,031	1,178	1,326	1,473
	60%	35	88	177	354	530	707	884	1,061	1,237	1,414	1,591	1,768
	70%	41	103	206	412	619	825	1,031	1,237	1,444	1,650	1,856	2,062
	80%	47	118	236	471	707	943	1,178	1,414	1,650	1,885	2,121	2,357
	90%	53	133	265	530	795	1,061	1,326	1,591	1,856	2,121	2,386	2,651
	100%	59	147	295	589	884	1,178	1,473	1,768	2,062	2,357	2,651	2,946
Three-year mean peak of 2,946 razorbills in the offshore site in post-breeding period													
Regional population in post-breeding season = 18,004 birds (revised from Mitchell *et al.* 2004)													

The displacement impact in the post-breeding season is therefore 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. It is concluded that the effects of displacement on the regional razorbill population in the post-breeding period are ***not significant*** under the EIA Regulations.

Non-breeding period

Assuming 40% of all razorbills were displaced from the offshore site during the non-breeding period (November to March), this would affect an estimated 91 birds (Table 4.161), increasing to 154 birds if displacement is assumed to affect the offshore site and a 2 km buffer (Table 4.162). For the purposes of this assessment, it was assumed that 2% of all razorbills displaced from the offshore site and 2 km buffer during the non-breeding period (up to three birds) would die as a result. This corresponds to approximately 0.02% of the regional population in the non-breeding period (14,400 birds). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be lower than this. It was concluded that the remaining displaced razorbills would move to alternative foraging areas in the non-breeding period.

Although there was less variation in razorbill distribution in the study area between Years 1 and 3 at this time of year, analysis shows that fewer birds are present in the study area compared to other seasons. Although razorbills were widespread between November and March throughout the study area in all three years, densities in the offshore site at this time were generally lower than in the surrounding buffer area. This suggests that razorbills are not regularly relying on the offshore site at this time of year, and that any displacement effects from this area will be small.

The displacement impact in the non-breeding season is therefore 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. 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.

Table 4.160 Estimated number of razorbills at risk of mortality following displacement from offshore site in non-breeding period

	Mortality (%)											
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	0	1	2	5	7	9	11	14	16	18	20	23
	20%	1	2	5	9	14	18	23	27	32	36	41	45
	30%	1	3	7	14	20	27	34	41	48	54	61	68
	40%	**2**	5	9	18	27	36	45	54	64	73	82	91
	50%	2	6	11	23	34	45	57	68	79	91	102	114
	60%	3	7	14	27	41	54	68	82	95	109	123	136
	70%	3	8	16	32	48	64	79	95	111	127	143	159
	80%	4	9	18	36	54	73	91	109	127	145	163	182
	90%	4	10	20	41	61	82	102	123	143	163	184	204
	100%	5	11	23	45	68	91	114	136	159	182	204	227
Three-year mean peak of 227 razorbills in the offshore site in non-breeding period												
Regional population in the non-breeding season = 14,400 birds (Skov *et al.*, 1995)												

Table 4.161 Estimated number of razorbills at risk of mortality following displacement from offshore site plus 1 km buffer in non-breeding period

	Mortality (%)											
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	2	4	7	11	15	18	22	26	29	33	37
	20%	1	4	7	15	22	29	37	44	51	58	66	73
	30%	2	5	11	22	33	44	55	66	77	88	99	110
	40%	**3**	7	15	29	44	58	73	88	102	117	131	146
	50%	4	9	18	37	55	73	91	110	128	146	164	183
	60%	4	11	22	44	66	88	110	131	153	175	197	219
	70%	5	13	26	51	77	102	128	153	179	204	230	256
	80%	6	15	29	58	88	117	146	175	204	234	263	292
	90%	7	16	33	66	99	131	164	197	230	263	296	329
	100%	7	18	37	73	110	146	183	219	256	292	329	365
Three-year mean peak of 365 razorbills in the offshore site in non-breeding period												
Regional population in the non-breeding season = 14,400 birds (Skov *et al.*, 1995)												

Table 4.162 Estimated number of razorbills at risk of mortality following displacement from offshore site plus 2 km buffer in non -breeding period

	Mortality (%)											
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	2	4	8	12	15	19	23	27	31	35	38
	20%	2	4	8	15	23	31	38	46	54	61	69	77
	30%	2	6	12	23	35	46	58	69	81	92	104	115
	40%	**3**	8	15	31	46	61	77	92	108	123	138	154
	50%	4	10	19	38	58	77	96	115	134	154	173	192
	60%	5	12	23	46	69	92	115	138	161	184	207	230
	70%	5	13	27	54	81	108	134	161	188	215	242	269
	80%	6	15	31	61	92	123	154	184	215	246	276	307
	90%	7	17	35	69	104	138	173	207	242	276	311	346
	100%	8	19	38	77	115	154	192	230	269	307	346	384
Three-year mean peak of 384 razorbills in the offshore site in non-breeding period												
Regional population in the non-breeding season = 14,400 birds (Skov *et al.*, 1995)												

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 3.7 and Table 3.8. 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 3.7). 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 3.8). 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 3.7). 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 3.8). 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 3.7). 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 3.8). 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 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 ***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. 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

CRM estimated the number of potential razorbill collisions per season for four wind turbine designs scenarios (Table 3.6). The minimum height for the turbine blades above the sea surface for all four options is 27.5 m at mean sea level (MSL).

There is no specific SNH guidance on avoidance rates for seabirds, and therefore their default value of 98.0% is presented in this assessment for comparison purposes

CRM estimated no collisions in the “colony attendance” and “chicks at sea” periods of the breeding season combined (April to August) for any of the wind turbine design options under consideration (Table 4.163).

CRM estimated no collisions in the post-breeding and non-breeding periods combined (September to March) for any of the wind turbine design options under consideration (Table 4.163).

Table 4.163 Number of estimated razorbill collisions using avoidance rate of 98% for four wind farm options

	Option 1 (90x5MW)	Option 2 (75x6MW)	Option 3 (73x6.15MW)	Option 4 (73x6.15MW)
Collisions in breeding season (April to August), all ages	0	0	0	0
Collisions in post- and non-breeding period (September to March), all ages	0	0	0	0
Total collisions per year, all ages	0	0	0	0

It is concluded that for all four designs, collision mortality for razorbill is an effect of negligible magnitude, that is temporally long-term and reversible.

It is further concluded that for the wind farm designs examined here, the effects of collision mortality on razorbills from the regional population in the breeding, post-breeding and non-breeding periods are ***not significant*** under the EIA Regulations.

4.4.22.5 Decommissioning Phase

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.

4.4.22.6 Summary of combined effects

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 4.164). It is concluded that the overall impact on the regional population of razorbills in the breeding season is ***not significant*** under the EIA regulations.

Table 4.164 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	Low	Long term	Moderate	***Not Significant***
Barrier Effect, colony-attendance period	Low	Long term	Moderate	***Not Significant***
Vessel disturbance	Negligible	Long term	Minor	***Not significant***
Collision mortality	Negligible	Long term	Low	***Not significant***
All effects combined	Low	Long term	Moderate	***Not Significant***

In combination it is judged that the magnitude of the effects on the regional razorbill population in the non-breeding-period is low (Table 4.165). It is concluded that the overall impact on the regional population of razorbills in the post-breeding and non-breeding-period is ***not significant*** under the EIA regulations.

Table 4.165 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	Low	***Not significant***
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	***Not significant***

4.4.22.7 Cumulative Impact Assessment

Displacement

The potential cumulative displacement risk to razorbills from Neart na Gaoithe and other plans or projects is higher than for Neart na Gaoithe alone.

Projects identified during consultation and the undertaking of the EIA for which there is a potential for a cumulative displacement impact are:

- Inch Cape offshore wind farm;
- Seagreen Project Alpha offshore wind farm; and
- Seagreen Project Bravo offshore wind farm.

Evidence from existing offshore wind farms support the potential for a moderate level of displacement behaviour (e.g. Leopold et al. 2011) and a predicted 40% displacement effect has been used in this assessment.

Breeding season

In the breeding season, peak estimated numbers of razorbills were recorded in the Neart na Gaoithe offshore site during the “chicks-at-sea” period (July and August). Similarly for Seagreen projects Alpha and Bravo, peak estimated numbers were higher in the “chicks-at-sea” period, compared to the colony attendance period (Seagreen 2012). Consequently, cumulative displacement during the "chicks-at-sea" period was considered in this assessment of displacement impacts during the breeding season.

Peak estimated numbers within the Seagreen Project Alpha site boundary occurred in July 2010 (2,102 birds), and 1,535 birds in August 2011. This provides a peak mean across two years, of 1,819 razorbills during the "chicks-at-sea" part of the breeding period (July and August).

Assuming that 30% of the population present are immature birds (Wanless *et al.,* 1998), then of the peak mean of 1,819 individuals at project Alpha, 1,273 may be breeding adults.

Assuming 40% of all razorbills were displaced from the Seagreen project Alpha and a 2 km buffer during the "chicks-at-sea" part of the breeding season, this would affect an estimated 509 breeding adults (Table 4.166).

Assuming that there is the potential for up to 10% rate of mortality during the "chicks-at-sea" part of the breeding period then up to 51 breeding adult razorbills may be impacted by displacement from Seagreen project Alpha and a 2 km buffer (Table 4.166).

Table 4.166 Estimated number of adult razorbills predicted to be at risk of mortality following displacement from Seagreen Project Alpha plus 2 km buffer in the "chicks-at-sea" part of the breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	3	6	13	25	38	51	64	76	89	102	115	127
	20%	5	13	25	51	76	102	127	153	178	204	229	255
	30%	8	19	38	76	115	153	191	229	267	306	344	382
	40%	10	25	**51**	102	153	204	255	306	356	407	458	509
	50%	13	32	64	127	191	255	318	382	446	509	573	637
	60%	15	38	76	153	229	306	382	458	535	611	687	764
	70%	18	45	89	178	267	356	446	535	624	713	802	891
	80%	20	51	102	204	306	407	509	611	713	815	917	1,018
	90%	23	57	115	229	344	458	573	687	802	917	1,031	1,146
	100%	25	64	127	255	382	509	637	764	891	1,018	1,146	1,273
Two-year mean peak of 1,273 adult razorbills in Seagreen Project Alpha & 2 km buffer in "chicks-at-sea" period													
SPA breeding population within mean max foraging range (48.5 km) = 6,110 birds (SMP 2013)													

There are no estimates for the number of razorbills within project Bravo for the "chicks-at-sea" period (July and August), however reading across from presented figures it is estimated that approximately 800 individuals occurred in August 2010 (Seagreen 2012). No peak numbers for 2010 are presented in the ES and so calculating a peak mean is not possible for Project Bravo (Seagreen 2012). Assuming that 30% of the population present are immature birds (Wanless *et al.,* 1998), then of the 800 individuals at project Bravo, 560 birds may be breeding adults.

Assuming 40% of all razorbills were displaced from the Seagreen project Bravo and a 2 km buffer during the "chicks-at-sea" part of the breeding season, this would affect an estimated 224 breeding adults (Table 4.167).

Assuming that there is the potential for up to 10% rate of mortality during the “chicks-at-sea” part of the breeding period then up to 22 breeding adult razorbills may be impacted by displacement from Seagreen project Bravo and a 2 km buffer (Table 4.167).

Table 4.167 Estimated number of adult razorbills predicted to be at risk of mortality following displacement from Seagreen Project Bravo plus 2 km buffer in the “chicks-at-sea” part of the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	3	6	11	17	22	28	34	39	45	50	56
	20%	2	6	11	22	34	45	56	67	78	90	101	112
	30%	3	8	17	34	50	67	84	101	118	134	151	168
	40%	4	11	**22**	45	67	90	112	134	157	179	202	224
	50%	6	14	28	56	84	112	140	168	196	224	252	280
	60%	7	17	34	67	101	134	168	202	235	269	302	336
	70%	8	20	39	78	118	157	196	235	274	314	353	392
	80%	9	22	45	90	134	179	224	269	314	358	403	448
	90%	10	25	50	101	151	202	252	302	353	403	454	504
	100%	11	28	56	112	168	224	280	336	392	448	504	560
One year peak of 560 adult razorbills in Seagreen Project Bravo & 2 km buffer in “chicks-at-sea” period													
SPA breeding population within mean max foraging range (48.5 km) = 6,110 birds (SMP 2013)													

Based on the above, the predicted cumulative displacement impacts on adult razorbills from Neart na Gaoithe and Seagreen projects Alpha and Bravo and a 2 km buffer in the “chicks-at-sea” part of the breeding period involve a predicted 1,377 adult birds, with 10% mortality of 137 adults. This number corresponds to approximately 2.2% of the regional breeding SPA population within mean maximum foraging range of the offshore site (6,110 birds). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

The predicted number of adult razorbills displaced from the Inch Cape site during the breeding season was 295 birds (ICOL, 2013). Assuming 40% of all razorbills were displaced from the Inch Cape site during the breeding season, this would affect an estimated 118 breeding adults.

Assuming that there is the potential for up to 10% rate of mortality during the breeding period then up to 12 breeding adult razorbills may be impacted by displacement mortality from the Inch Cape site.

Based on the above, the predicted cumulative displacement impacts on adult razorbills from Neart na Gaoithe and a 2 km buffer and the Inch Cape site in the breeding period involve a predicted 762 adult birds, with 10% mortality of 76 adults. This number corresponds to approximately 1.2% of the regional breeding SPA population within mean maximum foraging range of the offshore site (6,110 birds). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

Based on the predicted numbers reported for the four proposed development areas a total of 1,495 adult razorbills may be displaced from the development sites and a 2 km buffer during the “chicks-at-sea” part of the breeding period, with 10% mortality of 149 adult birds.

The 10% mortality corresponds to approximately 2.4% of the regional breeding SPA population within mean maximum foraging range of the offshore site (6,110 birds). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

Based on the above, this impact is categorised as having low magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative effects of displacement on the regional razorbill population in the "chicks-at-sea" part of the breeding season are ***not significant*** under the EIA Regulations.

A separate CIA for displacement has not been undertaken for the regional razorbill population in the colony attendance part of the breeding-period (April to June), as data was not complete for these months for the Seagreen or Inch Cape projects. The predicted displacement of razorbills due to Neart na Gaoithe wind farm in the colony attendance part of the breeding period was lower than for the "chicks-at-sea" part of the breeding season, and so it was assumed that any cumulative displacement impact for the regional razorbill population in the colony attendance part of the breeding-period would be lower to that of the "chicks-at-sea" period. This impact was therefore categorised as having low magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative effects of displacement on the regional razorbill population in the colony attendance part of the breeding season (April to June) are ***not significant*** under the EIA Regulations.

Post-breeding period

In the post-breeding period (September and October), peak estimated numbers within the Seagreen Project Alpha site boundary occurred in October 2010. There are no estimates for the number of razorbills at this time, however reading across from presented figures it is estimated that approximately 750 individuals occurred in October 2010 (Seagreen 2012).

Assuming 40% of all razorbills were displaced from the Seagreen project Alpha and a 2 km buffer during the post-breeding period, this would affect an estimated 300 birds (Table 4.168).

Table 4.168 Estimated number of razorbills (all ages) predicted to be at risk of mortality following displacement from Seagreen Project Alpha plus 2 km buffer in the post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	2	4	8	15	23	30	38	45	53	60	68	75
	20%	3	8	15	30	45	60	75	90	105	120	135	150
	30%	5	11	23	45	68	90	113	135	158	180	203	225
	40%	**6**	15	30	60	90	120	150	180	210	240	270	300
	50%	8	19	38	75	113	150	188	225	263	300	338	375
	60%	9	23	45	90	135	180	225	270	315	360	405	450
	70%	11	26	53	105	158	210	263	315	368	420	473	525
	80%	12	30	60	120	180	240	300	360	420	480	540	600
	90%	14	34	68	135	203	270	338	405	473	540	608	675
	100%	15	38	75	150	225	300	375	450	525	600	675	750
One-year peak of 750 razorbills in Seagreen Project Alpha & 2 km buffer in post-breeding period													
Regional population in post-breeding season = 18,004 birds (revised from Mitchell *et al.* 2004)													

Assuming that there is the potential for up to 2% rate of mortality during the post-breeding period then up to six razorbills may be impacted by displacement mortality from Seagreen project Alpha and a 2 km buffer (Table 4.168).

Peak estimated numbers within the Seagreen project Bravo site boundary occurred in September, with 1,293 birds in 2010, and 994 birds in 2011 (Seagreen 2012). This provides a peak mean across two years, of 1,144 razorbills during the post-breeding period (September and October).

Assuming 40% of all razorbills were displaced from the Seagreen project Bravo and a 2 km buffer during the post-breeding season, this would affect an estimated 458 birds (Table 4.169).

Assuming that there is the potential for up to 2% rate of mortality during the post-breeding period then up to nine razorbills may be impacted by displacement mortality from Seagreen project Bravo and a 2 km buffer (Table 4.169).

Table 4.169 Estimated number of razorbills (all ages) predicted to be at risk of mortality following displacement from Seagreen Project Bravo plus 2 km buffer in the post-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	2	6	11	23	34	46	57	69	80	92	103	114
	20%	5	11	23	46	69	92	114	137	160	183	206	229
	30%	7	17	34	69	103	137	172	206	240	275	309	343
	40%	**9**	23	46	92	137	183	229	275	320	366	412	458
	50%	11	29	57	114	172	229	286	343	400	458	515	572
	60%	14	34	69	137	206	275	343	412	480	549	618	686
	70%	16	40	80	160	240	320	400	480	561	641	721	801
	80%	18	46	92	183	275	366	458	549	641	732	824	915
	90%	21	51	103	206	309	412	515	618	721	824	927	1,030
	100%	23	57	114	229	343	458	572	686	801	915	1,030	1,144
Two year peak mean of 1,144 razorbills in Seagreen Project Bravo & 2 km buffer in post-breeding period													
Regional population in post-breeding season = 18,004 birds (revised from Mitchell *et al.* 2004)													

Based on the above, the predicted cumulative displacement impacts on razorbills from Neart na Gaoithe and Seagreen projects Alpha and Bravo and a 2 km buffer in the post-breeding period involve a predicted 1,936 displaced birds, with 2% mortality of 39 birds. This mortality corresponds to approximately 0.2% of the regional population in the post-breeding period (18,004 birds) (revised from Mitchell *et al.,* 2004). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

In the post-breeding period (September and October), peak estimated numbers within the Inch Cape site and a 4 km buffer area occurred in October 2010. There are no estimates for the number of razorbills at this time, however reading across from presented figures it is estimated that approximately 4,000 individuals occurred in October 2010 (ICOL, 2012).

Assuming 40% of all razorbills were displaced from Inch Cape and a 4 km buffer during the post- breeding season, this would affect an estimated 1,600 birds (Table 4.170).

Assuming that there is the potential for up to 2% rate of mortality during the post-breeding period then up to 32 razorbills may be impacted by displacement mortality from Inch Cape and a 4 km buffer (Table 4.170).

Based on the above, the predicted cumulative displacement impacts on razorbills from Neart na Gaoithe and a 2 km buffer and Inch Cape and a 4 km buffer in the post-breeding period involve a predicted 2,778 birds, with 2% mortality of 56 birds. This mortality corresponds to approximately 0.3% of the regional population in the post-breeding period (18,004 birds) (revised from Mitchell *et al.,* 2004). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this. The estimate is also based on a 4 km buffer around Inch Cape, so estimated numbers of displaced razorbills are higher than they would be using a 2 km buffer. This has been taken into account in this assessment.

Table 4.170 Estimated number of razorbills (all ages) predicted to be at risk of mortality following displacement from Inch Cape plus 4 km buffer in the post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	8	20	40	80	120	160	200	240	280	320	360	400
	20%	16	40	80	160	240	320	400	480	560	640	720	800
	30%	24	60	120	240	360	480	600	720	840	960	1,080	1,200
	40%	**32**	80	160	320	480	640	800	960	1,120	1,280	1,440	1,600
	50%	40	100	200	400	600	800	1,000	1,200	1,400	1,600	1,800	2,000
	60%	48	120	240	480	720	960	1,200	1,440	1,680	1,920	2,160	2,400
	70%	56	140	280	560	840	1,120	1,400	1,680	1,960	2,240	2,520	2,800
	80%	64	160	320	640	960	1,280	1,600	1,920	2,240	2,560	2,880	3,200
	90%	72	180	360	720	1,080	1,440	1,800	2,160	2,520	2,880	3,240	3,600
	100%	80	200	400	800	1,200	1,600	2,000	2,400	2,800	3,200	3,600	4,000
One year peak of 4,000 razorbills in Inch Cape & 4 km buffer in post-breeding period													
Regional population in post-breeding season = 18,004 birds (revised from Mitchell *et al.* 2004)													

Based on the predicted numbers reported for the four proposed development areas a total of 3,536 razorbills may be displaced from the development sites and a 2 km buffer (4 km for Inch Cape) during the post-breeding period, with 2% mortality of 71 birds. The 2% mortality corresponds to approximately 0.4% of the regional population in the post-breeding period (18,004 birds) (revised from Mitchell *et al.,* 2004). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this. The estimate is also based on a 4 km buffer around Inch Cape, so estimated numbers of displaced razorbills are higher than they would be using a 2 km buffer. This has been taken into account in this assessment.

Based on the above, this impact is categorised as having negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative effects of displacement on the regional razorbill population in the post-breeding season are ***not significant*** under the EIA Regulations.

A separate CIA for displacement has not been undertaken for the regional razorbill population in the non-breeding period (November to March), as data was not available for these months for the Seagreen or Inch Cape projects. The predicted displacement of razorbill 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 considered to be low. Therefore it is not plausible that the Neart na Gaoithe development could contribute to a significant cumulative impact for the regional populations during this period.

Collision mortality

No significant impacts were predicted to arise from collision mortality caused by Neart na Gaoithe for razorbills in the breeding, post-breeding or non-breeding periods. This was based on zero bird collisions predicted by collision risk modelling, indicating very low levels of collision mortality for this species. It is therefore not plausible that Neart na Gaoithe could contribute to a significant cumulative collision mortality impact for the regional population of razorbills in the breeding, post-breeding or non-breeding periods. As a result, no further cumulative impact assessment for collision mortality was undertaken for this species.

4.4.22.8 Mitigation measures

The following mitigation measures are suggested for razorbill:

1 - Minimise disturbance by reduced vessel usage during peak periods of razorbill 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;

4.4.23 Atlantic puffin *Fratercula arctica*

4.4.23.1 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 offshore site and 8 km buffer 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).

4.4.23.2 Offshore site and 8 km buffer area

Puffin was one of the most frequently recorded seabirds on surveys in the Neart na Gaoithe study area during the baseline surveys, with a total of 11,199 birds recorded in Year 1, 6,622 birds in Year 2 and 5,983 birds in Year 3 (raw numbers, all sea states). The majority of guillemots were recorded in the buffer area (Table 4.2).

During the Year 1 breeding season (April to August), the mean estimated number of puffins in the offshore site was 1,178 birds, with a peak of 3,507 birds in August (Table 4.171). In the same period of Year 2, the mean estimated number of puffins in the offshore site was 1,396 birds, with a peak of 2,481 birds in July. In Year 3, the mean estimated number of puffins during the breeding period was 2,012 birds, with a peak of 3,812 birds in April.

Table 4.171 Estimated numbers of puffins in the offshore site (and 1, 2 & 8 km buffer) in Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 8 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	0	0	0	0	0	39	39	79
Yr1 Dec	0	0	0	0	0	0	0	162
Yr1 Jan	0	0	0	0	0	0	0	0
Yr1 Feb	0	0	0	0	0	0	0	0
Yr1 Mar	241	167	347	0	241	289	422	1,136
Yr1 Apr	1,387	876	2,193	245	1,632	2,084	2,496	5,342
Yr1 May	188	100	354	20	208	373	625	3,428
Yr1 Jun	217	135	348	41	258	358	564	2,487
Yr1 Jul	242	161	364	41	284	544	620	2,565
Yr1 Aug	3,391	1,410	8,158	116	3,507	6,717	15,016	37,677
Yr1 Sep	465	300	720	0	465	719	794	1,513
Yr1 Oct	1,881	1,286	2,750	0	1,881	2,900	4,109	12,168
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	42	12	140	0	42	42	42	83
Yr2 Jan	0	0	0	0	0	0	17	137
Yr2 Feb	53	39	71	7	60	67	102	331
Yr2 Mar	27	15	48	0	27	27	53	332
Yr2 Apr	1,721	1,102	2,687	48	1,769	2,745	3,442	7,197
Yr2 May	1,734	1,199	2,509	108	1,842	2,479	3,002	9,720
Yr2 Jun	263	144	479	129	391	532	662	3,013
Yr2 Jul	2,279	1,300	3,995	202	2,481	2,831	3,288	9,199
Yr2 Aug	442	292	668	55	496	624	684	1,738
Yr2 Sep	336	206	550	0	336	537	874	3,541
Yr2 Oct	1,821	1,429	2,320	0	1,821	2,935	4,994	17,089
Yr3 Nov	112	73	173	0	112	168	243	1,067
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	72	45	114	0	72	90	126	377
Yr3 Feb	69	38	125	0	69	139	156	278
Yr3 Mar	904	628	1,302	7	911	1,363	1,864	6,228
Yr3 Apr	3,792	2,953	4,870	21	3,812	5,474	7,568	14,500
Yr3 May	1,244	820	1,888	69	1,313	1,726	2,158	6,167
Yr3 Jun	613	427	879	76	689	1,021	1,201	3,065
Yr3 Jul	3,526	2,449	5,078	193	3,719	4,899	6,175	19,018
Yr3 Aug	519	367	736	7	526	691	1,147	5,051
Yr3 Sep	832	530	1,304	0	832	1,104	1,739	4,188
Yr3 Oct	498	362	685	27	525	710	950	2,579

In the Year 1 post-breeding period (September and October), the mean estimated number of puffins in the offshore site was 1,173 birds, with a peak of 1,881 birds in October (Table 4.171). In the same period of Year 2, the mean estimated number of puffins in the offshore site was 1,079 birds, with a peak of 1,821 birds in October. In Year 3, the mean estimated number of puffins during the post-breeding period was 679 birds, with a peak of 832 birds in September.

In the Year 1 non-breeding period (November to March), the mean estimated number of puffins in the offshore site was 48 birds, with a peak of 241 birds in March (Table 4.171). In the same period of Year 2, the mean estimated number of puffins in the offshore site was 32 birds, with a peak of 60 birds in February, although there was no November survey. In Year 3, the mean estimated number of puffins during the non-breeding period was 291 birds, with a peak of 911 birds in March, although there was no December survey.

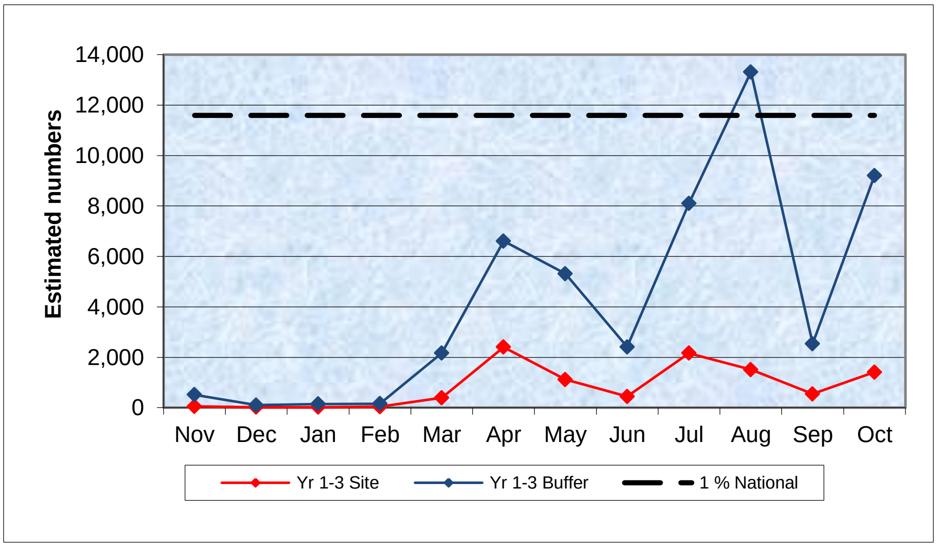


Figure 4.120 Mean monthly estimated numbers of puffins in the offshore site and buffer area in Years 1 to 3 (Three-year mean)

Mean estimated numbers of puffins in the offshore site were low between November and March, increasing to a peak in April (2,404 birds) at the start of the breeding season, and again in July (2,161 birds), towards the end of the breeding season (Figure 4.120). A similar pattern was recorded in the buffer area, although mean estimated numbers were generally higher, with peaks in April, August and October. Mean estimated numbers of puffins in August (13,312 birds, three-year mean) exceeded the 1% threshold of national importance (11,590 birds) (Mitchell, *et al.,*2004).

In the Year 1 non-breeding season (November to March), 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 4.121).

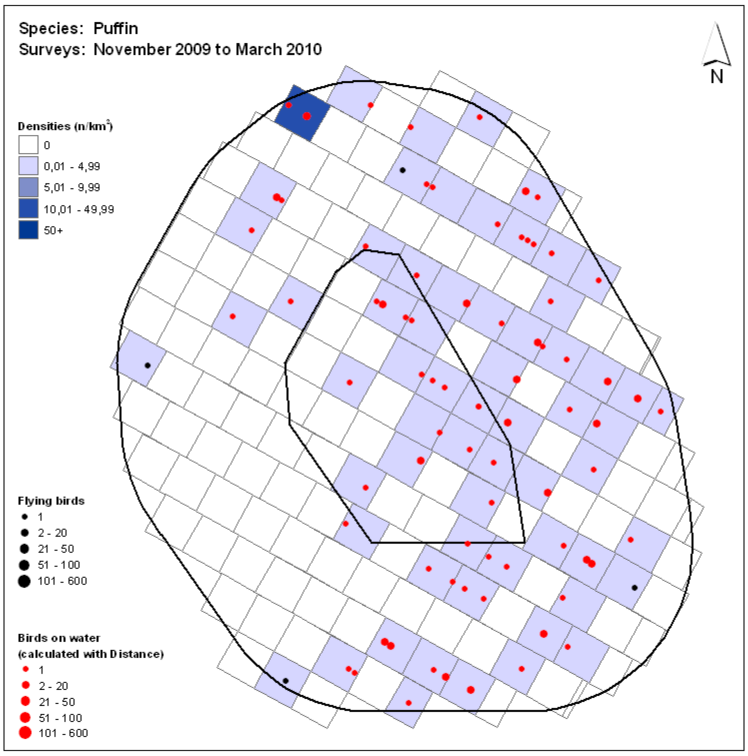


Figure 4.121 Puffin density between November and March, Year 1

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 4.122). Puffins were slightly less widespread in the offshore site at this time in Year 2, compared to the same period of Year 1.

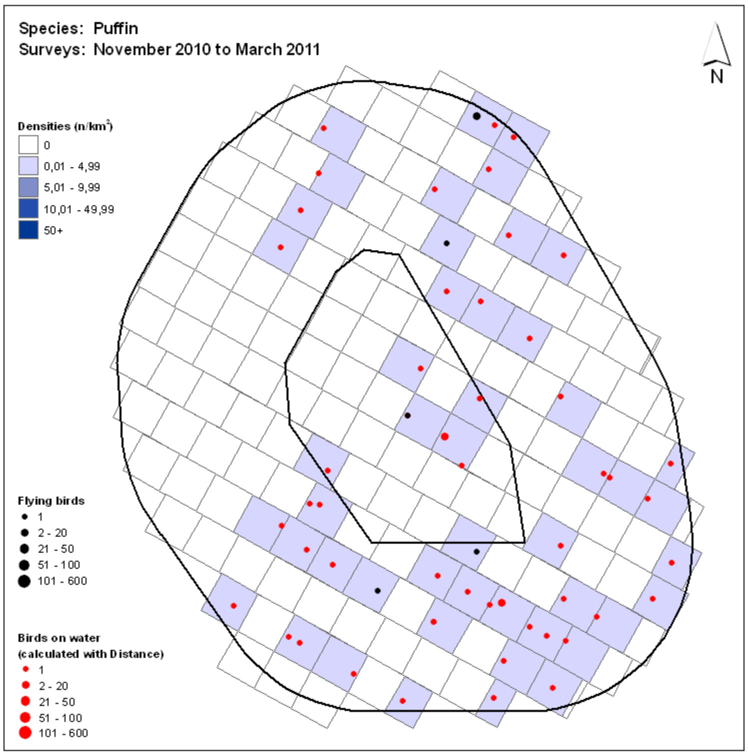


Figure 4.122 Puffin density between November and March, Year 2

Puffin distribution in the offshore site in the Year 3 non-breeding season was similar to previous years, with few birds recorded at low densities (Figure 4.123). Puffins were most widespread in the south-east of the buffer area, although desbities were low at this time.

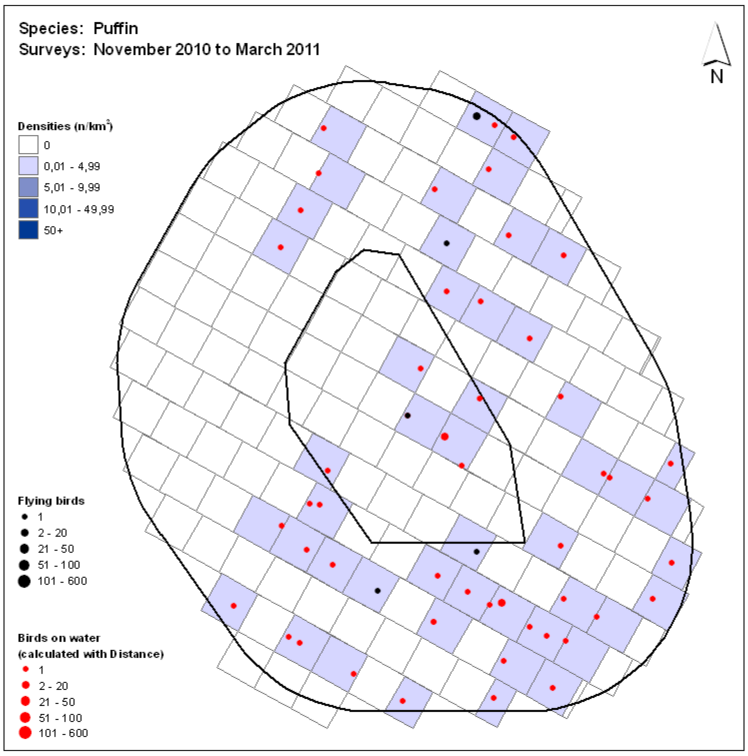


Figure 4.123 Puffin density between November and March, Year 3

During the Year 1 breeding season (April to August), highest densities of puffins were recorded in the southern half of the offshore site and the buffer area, with lower densities recorded in the north (Figure 4.124).

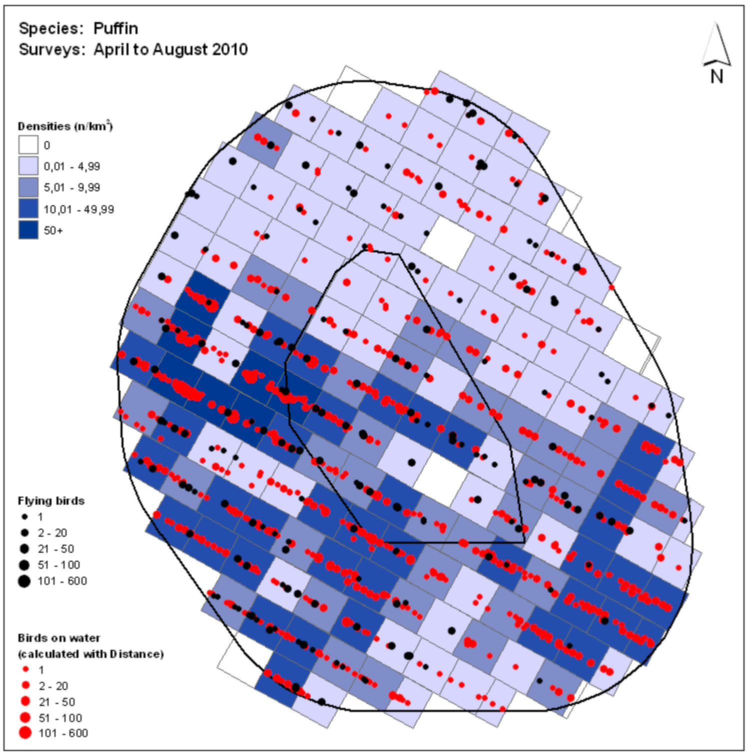


Figure 4.124 Puffin density between April and August, Year 1

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 offshore site and buffer area at this time (Figure 4.125).

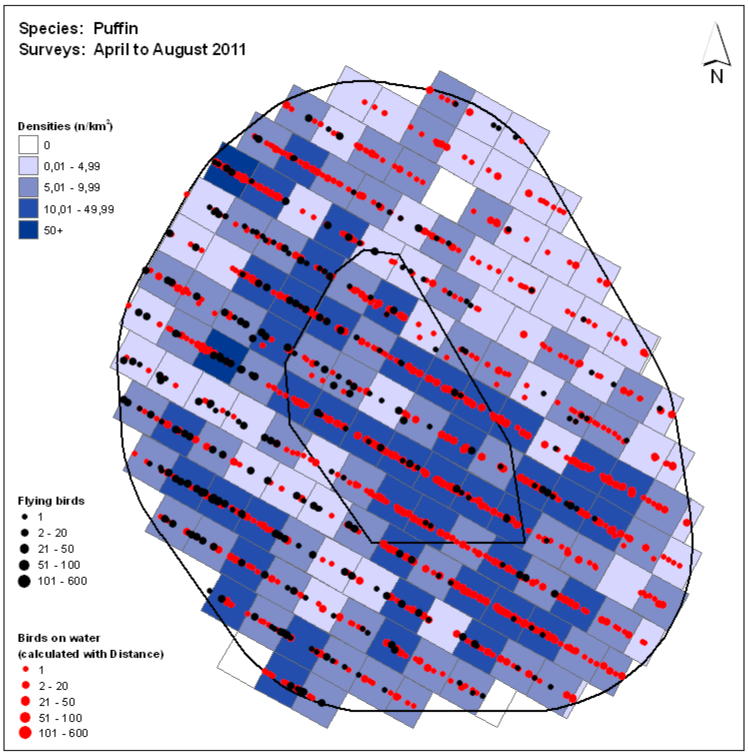


Figure 4.125 Puffin density between April and August, Year 2

In the Year 3 breeding season, puffins were more widespread than in previous years, with moderate to high densities recorded across the offshore site and the buffer area (Figure 4.126).

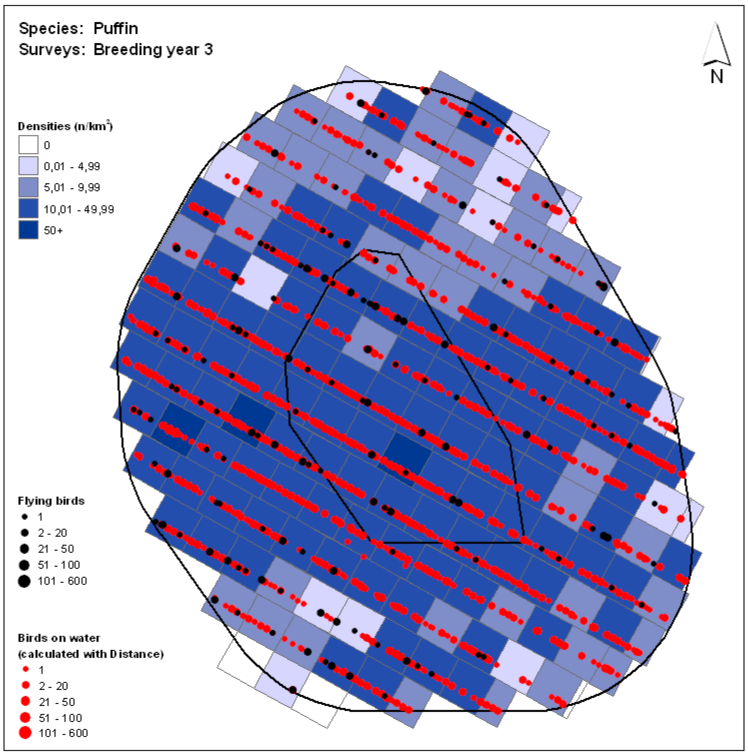


Figure 4.126 Puffin density between April and August, Year 3

In the Year 1 post-breeding period (September and October), 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 4.127).

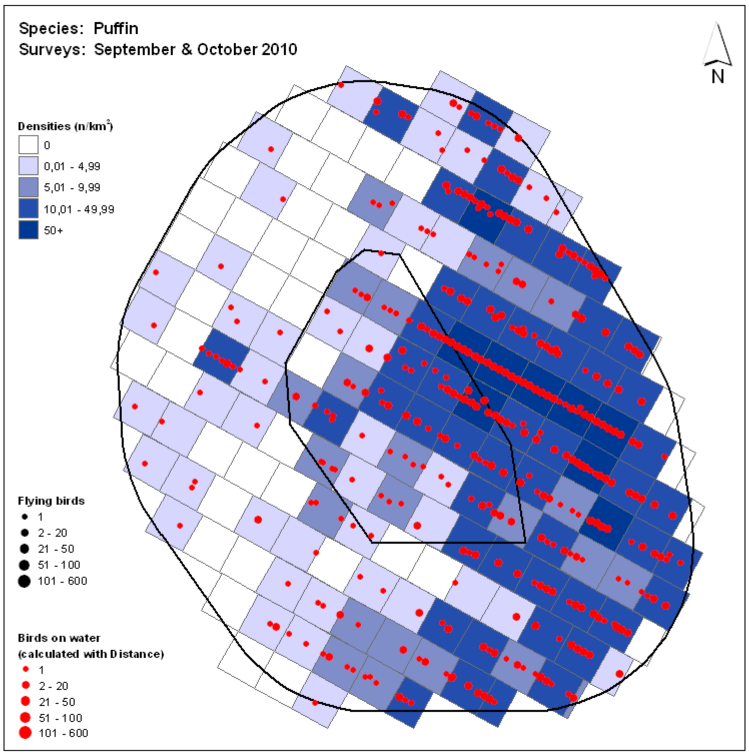


Figure 4.127 Puffin density in September and October, Year 1

In contrast, puffins remained widespread at mostly high densities in September and October of Year 2 across the offshore site and buffer area, with lower densities in the east of the buffer area at this time (Figure 4.128).

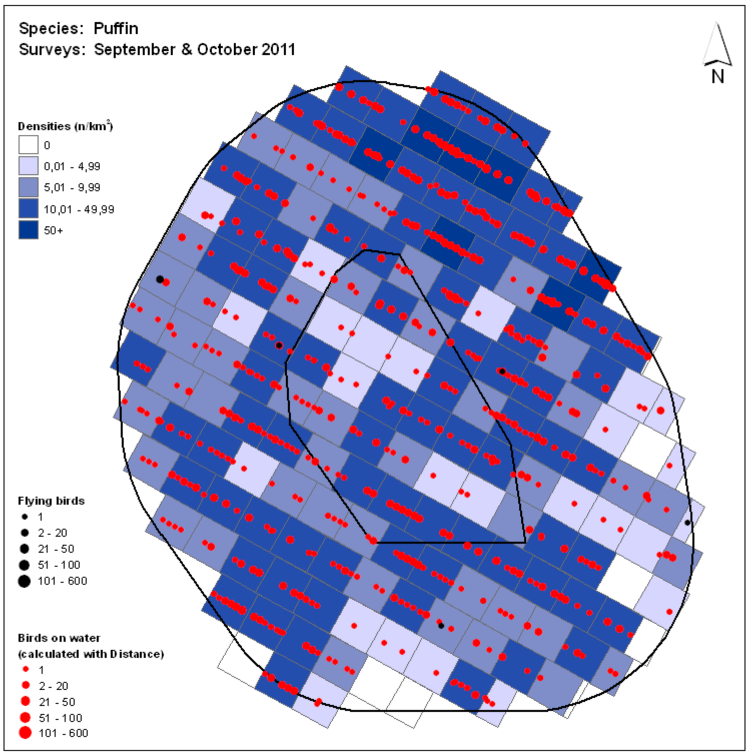


Figure 4.128 Puffin density in September and October, Year 2

In the Year 3 post-breeding period, puffins remained widespread in the offshore site and buffer area, although high density concentrations were more scattered than in Year 2 (Figure 4.129).

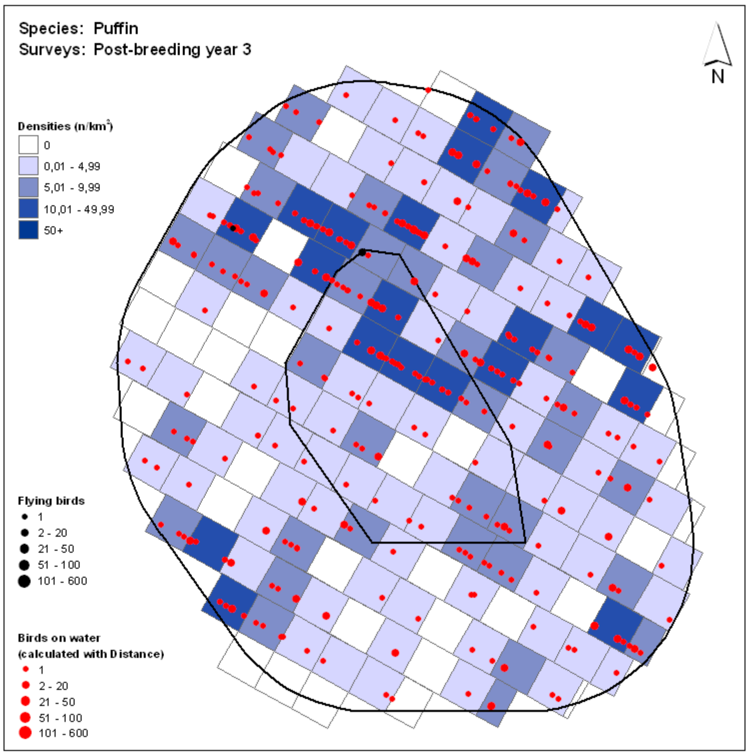
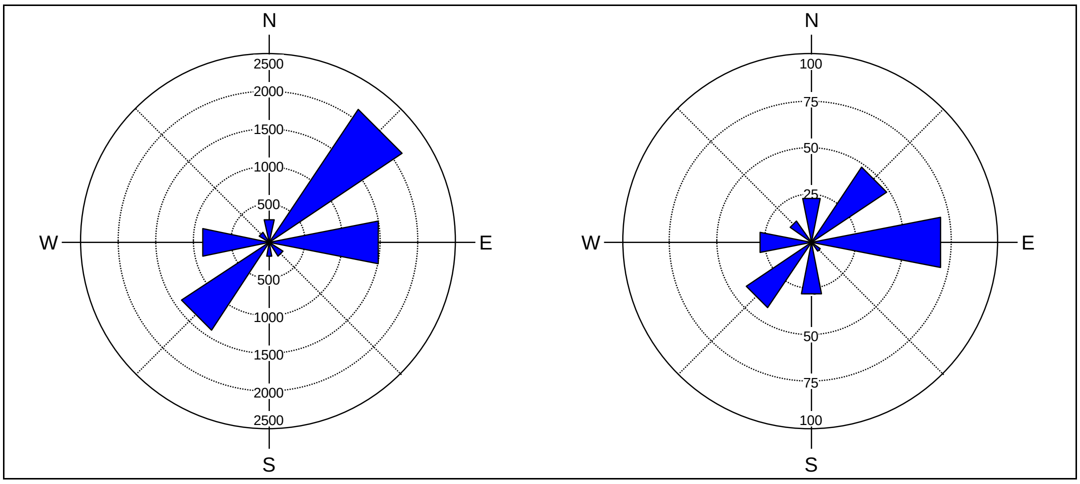


Figure 4.129Puffin density in September and October, Year 3

A total of 7,049 puffins were recorded in flight on baseline surveys, with almost all birds recorded flying below 27.5 m in height, and 98.8% of birds recorded flying below 7.5 m in height (Table 4.3). Three birds (0.04%) were recorded flying above 27.5 m, i.e. within the rotor swept zone, at an estimated height of 30 m.

Flight direction was recorded for 6,781 puffins in the breeding season (April to August), with direction recorded for 260 puffins in the post-breeding and non-breeding periods (September to March) (Figure 4.130).

In the breeding season, just over half of all birds recorded were flying north-east (31.4%) and east (21.8%), with just over one third of birds flying south-west (20.6%), and west (13.4%). In the non-breeding season, just under a half of all birds recorded were flying east (26.9%) and north-east (18.5%), with just over a quarter of birds flying south-west (16.2%) or west (10.8%).



April to August (n=6,781 birds) September to March (n=260 birds)
Numbers shown on figures are number of birds recorded

Figure 4.130 Flight direction of puffins in the offshore site and 8 km buffer area in Years 1 to 3

Foraging behaviour was recorded for 152 puffins in the offshore site and 8 km buffer area on baseline surveys, with four types of foraging behaviour recorded (Table 4.172). The majority of all foraging birds were recorded holding fish (63.8%). Prey identification was only recorded in four instances (all sandeels), with the remaining sightings being "unidentified fish".

Table 4.172 Puffin foraging behaviour in the offshore site and 8 km buffer area in Years 1 to 3

Behaviour	Number of birds
Actively searching	18
Pursuit diving	26
Holding fish	97
Pursuit plunging	11
Total	**152**

4.4.23.3 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).

A recent JNCC statistical analysis of ESAS data investigating possible marine SPAs, identified waters around the offshore site and 8 km buffer area as an important area for puffins during the breeding season (Kober *et al.,* 2010).

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 4.173). These SPAs held 14.9% of the UK breeding population and 7.5% of the biogeographic population at the time of designation (JNCC, 2013). 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 4.131.

Table 4.173 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 (pairs) [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 (2013) – SPA online species accounts. 2 SMP (2013) – 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).

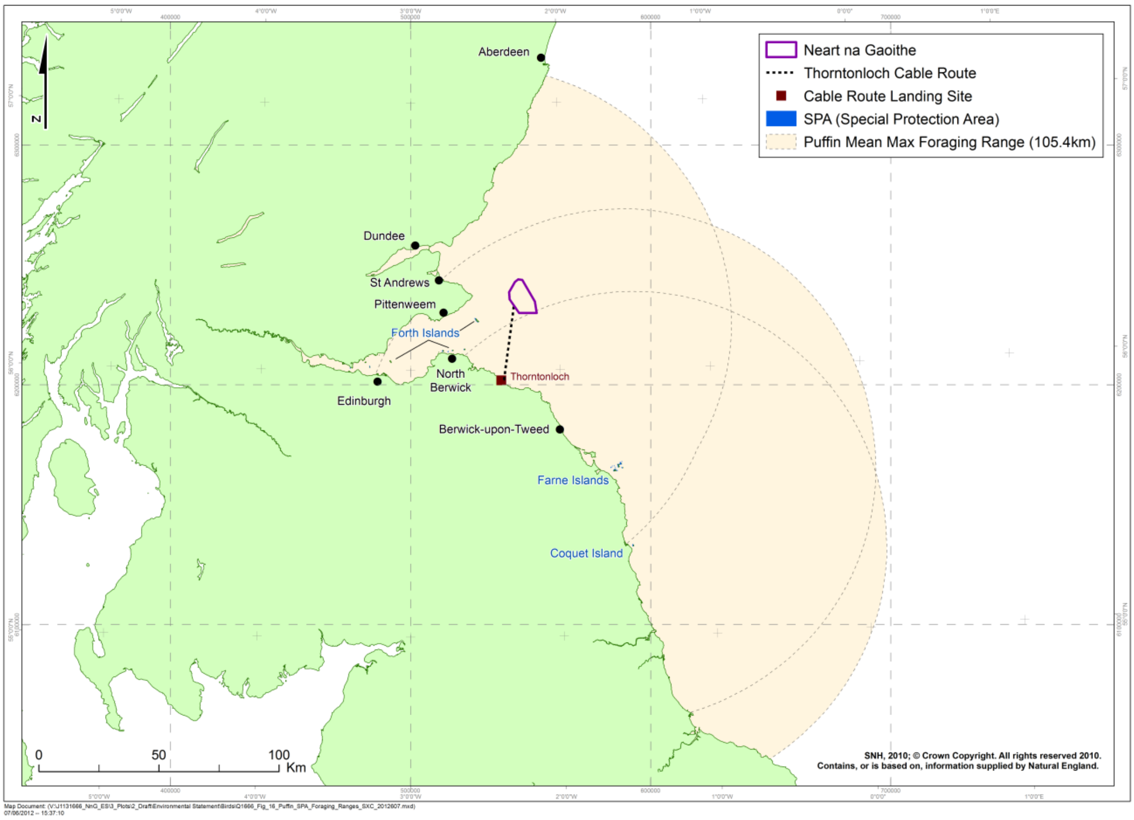


Figure 4.131 Puffin mean maximum foraging range from breeding SPAs in relation to the Development

4.4.23.4 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).

Populations

Puffin is the second commonest seabird species breeding in the region. Most recent available counts show that the breeding population of puffins on the Isle of May has increased since Seabird 2000, while colonies on Coquet Island and the Farne Islands have undergone recent declines (SMP, 2013) (Table 4.174).

Table 4.174 Recent counts & Seabird 2000 counts at main colonies for breeding puffin between Peterhead and Blyth

Colony	Distance to site (km)	Seabird 2000 count	Recent count (birds at colony)	Year	Percentage change since Seabird 2000
Isle of May	16	42,000 [2]	56,867 [1]	2009	+35.4%
Coquet Island	106	17,208 [2]	15,812 [1]	2009	-8.1%
Farne Islands	72	55,674 [2]	36,835 [1]	2008	-33.8%
				Mean percentage change	**-6.5%**

Sources: 1 SMP (2013) – Seabird Monitoring Programme Online Database. 2 Mitchell *et al.*, 2004.

Assuming that the sub-set of colonies that have been recently counted is representative, the regional decline since the Seabird 2000 counts amounts to -6.5%. 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 (146,670 pairs) has been adjusted downwards by 6.5%. On this basis, the regional breeding population is assumed to be 137,136 pairs, or 274,272 breeding adults.

The SPA breeding population within mean maximum foraging range of the offshore site was estimated to be 99,002 pairs, based on most recent available counts (Table 4.173).

The size of the post-breeding-period regional puffin population is estimated to be the same as the regional population between Peterhead and Blyth (274,272 birds).

The size of the regional puffin population in the non-breeding-period is assumed to be 23,633 birds. This is the mean of locality 1 in October to January (20,900 birds) and localities 2 and 3 between February to March (50,000) estimates given in Skov *et al*. (1995).

Nature conservation importance

The nature conservation importance of puffin using the offshore site is rated as High during the breeding season and post-breeding period, 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 three-year mean peak estimated number of puffins present in the offshore site between April and August (the breeding season) (3,267 birds) is 1.6% of the SPA breeding population within mean maximum foraging range of the offshore site (99,002 pairs).

Offshore wind farm studies

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 concluded, “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.

Construction Phase

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.

Operational Phase

In order to assess the potential impacts from displacement, the approach presented in the interim guidance note on displacement issued by the JNCC and NE has been followed (JNCC & NE 2012).

Peak estimated numbers of puffins in the offshore site in the breeding season (April to August), the post-breeding season (September and October) and the non-breeding season (November to March) for Years 1 to 3 were averaged to get the three-year mean peak per season. This was repeated for a 1 km and 2 km buffer around the offshore site (Table 4.175).

The three-year mean peak estimated number of puffins was then used to predict the estimated number of birds at potential risk of mortality following displacement in the at-colony period of the breeding season in the offshore site (plus 1 km and 2 km buffers), as recommended in the draft guidance note on displacement (JNCC & NE, 2012). This was repeated for the post-breeding period and the non-breeding period.

Table 4.175 Seasonal three-year mean peak estimated numbers of puffins in the offshore site (plus 1 & 2 km buffer)

Year	Offshore site		
	Breeding	Post-breeding	Non-breeding
Year 1	3,507	1,881	241
Year 2	2,481	1,821	60
Year 3	3,812	832	911
3-year mean peak	3,267	1,511	404
Year	Offshore site + 1 km		
	Breeding	Post-breeding	Non-breeding
Year 1	6,717	2,900	289
Year 2	2,831	2,935	67
Year 3	5,474	1,104	1,363
3-year mean peak	5,007	2,313	573
Year	Offshore site + 2 km		
	Breeding	Post-breeding	Non-breeding
Year 1	15,016	4,109	422
Year 2	3,288	4,994	102
Year 3	7,568	1,739	1,864
3-year mean peak	8,624	3,614	796

Likely impacts of displacement

Breeding season

For this assessment, it was assumed that there will be 40% displacement of puffins from the offshore site in all seasons. Additional scenarios considering 40% displacement out to a 1 km and a 2 km buffer are also presented.

Assuming 40% of all puffins were displaced from the offshore site during the "at-colony" part of the breeding season (April to June), this would affect an estimated 1,307 birds (Table 4.176) increasing to 3,450 birds if displacement is assumed to affect the offshore site and a 2 km buffer (Table 4.178). Assuming that 30% of the population present are immature birds (Wanless *et al.,* 1998), this would mean 915 breeding adults would be displaced from the offshore site, increasing to 2,415 breeding adults if displacement is assumed to affect the offshore site and a 2 km buffer.

For the purposes of this assessment, it was assumed that 10% of all puffins displaced from the offshore site during the breeding season (up to 345 birds) would die as a result. Applying the correction for immature birds being present in the breeding season (Wanless *et al.,* 1998), gives an estimated mortality of up to 241 breeding adults. This corresponds to approximately 0.1% of the regional breeding SPA population within mean maximum foraging range of the offshore site (99,002 pairs). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

Table 4.176 Estimated number of puffins at risk of mortality following displacement from offshore site in breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	7	16	33	65	98	131	163	196	229	261	294	327
	20%	13	33	65	131	196	261	327	392	457	523	588	653
	30%	20	49	98	196	294	392	490	588	686	784	882	980
	40%	26	65	**131**	261	392	523	653	784	915	1,045	1,176	1,307
	50%	33	82	163	327	490	653	817	980	1,143	1,307	1,470	1,634
	60%	39	98	196	392	588	784	980	1,176	1,372	1,568	1,764	1,960
	70%	46	114	229	457	686	915	1,143	1,372	1,601	1,830	2,058	2,287
	80%	52	131	261	523	784	1,045	1,307	1,568	1,830	2,091	2,352	2,614
	90%	59	147	294	588	882	1,176	1,470	1,764	2,058	2,352	2,646	2,940
	100%	65	163	327	653	980	1,307	1,634	1,960	2,287	2,614	2,940	3,267
Three-year mean peak of 3,267 puffins in the offshore site in "at-colony" period													
SPA breeding population within mean max foraging range (105.4 km) = 99,002 pairs (SMP 2013)													

Table 4.177 Estimated number of puffins at risk of mortality following displacement from offshore site plus 1 km buffer in breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	10	25	50	100	150	200	250	300	350	401	451	501
	20%	20	50	100	200	300	401	501	601	701	801	901	1,001
	30%	30	75	150	300	451	601	751	901	1,051	1,202	1,352	1,502
	40%	40	100	**200**	401	601	801	1,001	1,202	1,402	1,602	1,803	2,003
	50%	50	125	250	501	751	1,001	1,252	1,502	1,752	2,003	2,253	2,504
	60%	60	150	300	601	901	1,202	1,502	1,803	2,103	2,403	2,704	3,004
	70%	70	175	350	701	1,051	1,402	1,752	2,103	2,453	2,804	3,154	3,505
	80%	80	200	401	801	1,202	1,602	2,003	2,403	2,804	3,204	3,605	4,006
	90%	90	225	451	901	1,352	1,803	2,253	2,704	3,154	3,605	4,056	4,506
	100%	100	250	501	1,001	1,502	2,003	2,504	3,004	3,505	4,006	4,506	5,007
Three-year mean peak of 5,007 puffins in the offshore site in "at-colony" period													
SPA breeding population within mean max foraging range (105.4 km) = 99,002 pairs (SMP 2013)													

Table 4.178 Estimated number of puffins at risk of mortality following displacement from offshore site plus 2 km buffer in breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	17	43	86	172	259	345	431	517	604	690	776	862
	20%	34	86	172	345	517	690	862	1,035	1,207	1,380	1,552	1,725
	30%	52	129	259	517	776	1,035	1,294	1,552	1,811	2,070	2,328	2,587
	40%	69	172	**345**	690	1,035	1,380	1,725	2,070	2,415	2,760	3,105	3,450
	50%	86	216	431	862	1,294	1,725	2,156	2,587	3,018	3,450	3,881	4,312
	60%	103	259	517	1,035	1,552	2,070	2,587	3,105	3,622	4,140	4,657	5,174
	70%	121	302	604	1,207	1,811	2,415	3,018	3,622	4,226	4,829	5,433	6,037
	80%	138	345	690	1,380	2,070	2,760	3,450	4,140	4,829	5,519	6,209	6,899
	90%	155	388	776	1,552	2,328	3,105	3,881	4,657	5,433	6,209	6,985	7,762
	100%	172	431	862	1,725	2,587	3,450	4,312	5,174	6,037	6,899	7,762	8,624
Three-year mean peak of 8,624 puffins in the offshore site in "at-colony" period													
SPA breeding population within mean max foraging range (105.4 km) = 99,002 pairs (SMP 2013)													

For the remaining 1,904 puffins displaced from the offshore site and 2 km buffer that survived, there could potentially be a detrimental impact on their breeding success, as a result of having to forage elsewhere. This number corresponds to approximately 1.0% of the regional breeding SPA population within mean maximum foraging range of the offshore site (99,002 pairs).

However, it is likely that puffins can compensate for a moderate amount of displacement by choosing to forage elsewhere. Comparing the distribution of puffins in the offshore site and buffer area from Years 1 to 3 at this time of year shows that puffins were recorded in both the offshore site and the surrounding buffer area in mostly high densities. During the Year 1 breeding season, highest densities of puffins were recorded in the southern half of the offshore site and 8 km buffer, with lower densities recorded in the north (Figure 4.124). A broadly similar distribution pattern was recorded in Year 2, with predominantly high densities in the offshore site and most of the 8 km buffer, and lower densities of puffins in the north-east of the buffer area (Figure 4.125). In the Year 3 breeding season, puffins were more widespread than in previous years, with moderate to high densities recorded across the offshore site and the buffer area (Figure 4.126). This indicates that puffins are not regularly relying on the offshore site exclusively at this time of year.

Limited puffin GPS tracking data was available from the Isle of May in June 2010, based on a sample size of seven individuals (11 foraging trips) (CEH 2010). Although the sample size was small, these GPS tracks showed that some birds did use the offshore site, however birds also travelled considerably beyond the offshore site. This supports the conclusions from the distribution maps that puffins are not regularly relying on the offshore site exclusively at this time of year.

The main conclusion that can be drawn from this limited tagging data is that puffins are clearly capable of travelling considerable distances during the breeding season. Baseline surveys also indicate that the offshore site is not a key foraging area for puffins in the breeding season.

Based on the above, the impact of displacement on the breeding period 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 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 puffin population during the breeding season are ***not significant*** under the EIA Regulations.

Post-breeding period

Assuming 40% of all puffins were displaced from the offshore site during the post-breeding period (September and October), this would affect an estimated 604 birds (Table 4.179), increasing to 1,446 birds if displacement is assumed to affect the offshore site and a 2 km buffer (Table 4.181). For the purposes of this assessment, it was assumed that 2% of all puffins displaced from the offshore site during the post-breeding period (up to 29 birds) would die as a result. This corresponds to approximately 0.01% of the regional population in the post-breeding period (274,272 birds). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be lower than this.

Table 4.179 Estimated number of puffins at risk of mortality following displacement from offshore site in the post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	3	8	15	30	45	60	76	91	106	121	136	151
	20%	6	15	30	60	91	121	151	181	212	242	272	302
	30%	9	23	45	91	136	181	227	272	317	363	408	453
	40%	**12**	30	60	121	181	242	302	363	423	484	544	604
	50%	15	38	76	151	227	302	378	453	529	604	680	756
	60%	18	45	91	181	272	363	453	544	635	725	816	907
	70%	21	53	106	212	317	423	529	635	740	846	952	1,058
	80%	24	60	121	242	363	484	604	725	846	967	1,088	1,209
	90%	27	68	136	272	408	544	680	816	952	1,088	1,224	1,360
	100%	30	76	151	302	453	604	756	907	1,058	1,209	1,360	1,511
Three-year mean peak of 1,511 puffins in the offshore site in post-breeding period													
Regional population in post-breeding season = 274,272 birds (revised from Mitchell *et al.* 2004)													

Table 4.180 Estimated number of puffins at risk of mortality following displacement from offshore site plus 1 km buffer in post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	5	12	23	46	69	93	116	139	162	185	208	231
	20%	9	23	46	93	139	185	231	278	324	370	416	463
	30%	14	35	69	139	208	278	347	416	486	555	625	694
	40%	**19**	46	93	185	278	370	463	555	648	740	833	925
	50%	23	58	116	231	347	463	578	694	810	925	1,041	1,157
	60%	28	69	139	278	416	555	694	833	971	1,110	1,249	1,388
	70%	32	81	162	324	486	648	810	971	1,133	1,295	1,457	1,619
	80%	37	93	185	370	555	740	925	1,110	1,295	1,480	1,665	1,850
	90%	42	104	208	416	625	833	1,041	1,249	1,457	1,665	1,874	2,082
	100%	46	116	231	463	694	925	1,157	1,388	1,619	1,850	2,082	2,313
Three-year mean peak of 2,313 puffins in the offshore site in post-breeding period													
Regional population in post-breeding season = 274,272 birds (revised from Mitchell *et al.* 2004)													

Table 4.181 Estimated number of puffins at risk of mortality following displacement from offshore site plus 2 km buffer in post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	7	18	36	72	108	145	181	217	253	289	325	361
	20%	14	36	72	145	217	289	361	434	506	578	651	723
	30%	22	54	108	217	325	434	542	651	759	867	976	1,084
	40%	**29**	72	145	289	434	578	723	867	1,012	1,156	1,301	1,446
	50%	36	90	181	361	542	723	904	1,084	1,265	1,446	1,626	1,807
	60%	43	108	217	434	651	867	1,084	1,301	1,518	1,735	1,952	2,168
	70%	51	126	253	506	759	1,012	1,265	1,518	1,771	2,024	2,277	2,530
	80%	58	145	289	578	867	1,156	1,446	1,735	2,024	2,313	2,602	2,891
	90%	65	163	325	651	976	1,301	1,626	1,952	2,277	2,602	2,927	3,253
	100%	72	181	361	723	1,084	1,446	1,807	2,168	2,530	2,891	3,253	3,614
Three-year mean peak of 3,614 puffins in the offshore site in post-breeding period													
Regional population in post-breeding season = 274,272 birds (revised from Mitchell *et al.* 2004)													

It was concluded that the remaining 1,417 puffins displaced from the offshore site and 2 km buffer would move to alternative foraging areas in the post-breeding period. This number corresponds to approximately 0.5% of the regional population in the post-breeding period (274,272 birds).

Comparing the distribution of puffins in the study area from Years 1 to 3 at this time of year shows that there is considerable variation between years. 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 4.127). In contrast, puffins remained widespread across the offshore site and buffer area 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 4.128). In the Year 3 post-breeding period, puffins remained widespread in the offshore site and buffer area, although high density concentrations were more scattered than in Year 2 (Figure 4.129). These changes in distribution in the post-breeding period between years are most likely influenced by changes in the distribution of prey, and demonstrate that puffins are not regularly relying on the offshore site exclusively at this time of year.

The displacement impact in the post-breeding season is therefore 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. 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

Assuming 40% of all puffins were displaced from the offshore site during the non-breeding period (November to March), this would affect an estimated 162 birds (Table 4.182), increasing to 318 birds if displacement is assumed to affect the offshore site and a 2 km buffer (Table 4.184). For the purposes of this assessment, it was assumed that 2% of all puffins displaced from the offshore site and 2 km buffer during the non-breeding period (up to six birds) would die as a result. This corresponds to approximately 0.04% of the estimated regional population in the non-breeding period (14,400 birds). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be lower than this.

It was concluded that the remaining 312 puffins displaced from the offshore site and 2 km buffer would move to alternative foraging areas in the non-breeding period. This number corresponds to approximately 2.2% of the estimated regional population in the non-breeding period (14,400 birds).

Comparing the distribution of puffins in the offshore area from Years 1 to 3 showed that between November and March of all years, few birds were recorded in the offshore site, at low densities. This suggests that puffins are not regularly relying on the offshore site at this time of year, and that any displacement effects from this area will be small.

Table 4.182 Estimated number of puffins at risk of mortality following displacement from offshore site in non-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	2	4	8	12	16	20	24	28	32	36	40
	20%	2	4	8	16	24	32	40	48	57	65	73	81
	30%	2	6	12	24	36	48	61	73	85	97	109	121
	40%	**3**	8	16	32	48	65	81	97	113	129	145	162
	50%	4	10	20	40	61	81	101	121	141	162	182	202
	60%	5	12	24	48	73	97	121	145	170	194	218	242
	70%	6	14	28	57	85	113	141	170	198	226	255	283
	80%	6	16	32	65	97	129	162	194	226	259	291	323
	90%	7	18	36	73	109	145	182	218	255	291	327	364
	100%	8	20	40	81	121	162	202	242	283	323	364	404
Three-year mean peak of 404 puffins in the offshore site in non-breeding period													
Regional population in the non-breeding season = 23,633 birds (Skov *et al*., 1995)													

Table 4.183 Estimated number of puffins at risk of mortality following displacement from offshore site plus 1 km buffer in non-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	1	3	6	11	17	23	29	34	40	46	52	57
	20%	2	6	11	23	34	46	57	69	80	92	103	115
	30%	3	9	17	34	52	69	86	103	120	138	155	172
	40%	**5**	11	23	46	69	92	115	138	160	183	206	229
	50%	6	14	29	57	86	115	143	172	201	229	258	287
	60%	7	17	34	69	103	138	172	206	241	275	309	344
	70%	8	20	40	80	120	160	201	241	281	321	361	401
	80%	9	23	46	92	138	183	229	275	321	367	413	458
	90%	10	26	52	103	155	206	258	309	361	413	464	516
	100%	11	29	57	115	172	229	287	344	401	458	516	573
Three-year mean peak of 573 puffins in the offshore site in non-breeding period													
Regional population in the non-breeding season = 23,633 birds (Skov *et al*., 1995)													

Table 4.184 Estimated number of puffins at risk of mortality following displacement from offshore site plus 2 km buffer in non -breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	2	4	8	16	24	32	40	48	56	64	72	80
	20%	3	8	16	32	48	64	80	96	111	127	143	159
	30%	5	12	24	48	72	96	119	143	167	191	215	239
	40%	**6**	16	32	64	96	127	159	191	223	255	287	318
	50%	8	20	40	80	119	159	199	239	279	318	358	398
	60%	10	24	48	96	143	191	239	287	334	382	430	478
	70%	11	28	56	111	167	223	279	334	390	446	501	557
	80%	13	32	64	127	191	255	318	382	446	509	573	637
	90%	14	36	72	143	215	287	358	430	501	573	645	716
	100%	16	40	80	159	239	318	398	478	557	637	716	796
Three-year mean peak of 796 puffins in the offshore site in non-breeding period													
Regional population in the non-breeding season = 23,633 birds (Skov *et al*., 1995)													

The displacement impact in the non-breeding season is therefore 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. 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 3.7 and Table 3.8. 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 3.7). 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 (F. Daunt pers. comm.). 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 3.8). 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 3.7). 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 (F. 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 3.8). 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 (56,867 pairs in 2012) and Craigleith (12,100 pairs in 2003) (SMP 2013) together hold approximately 50% of the regional breeding puffin population (137,136 pairs) (based on Mitchell *et al,* 2004). 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, individual puffins 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

CRM estimated the number of potential puffin collisions per season for four wind turbine designs scenarios (Table 3.6). The minimum height for the turbine blades above the sea surface for all four options is 27.5 m at mean sea level (MSL).

There is no specific SNH guidance on avoidance rates for seabirds, and therefore their default value of 98.0% is presented in this assessment for comparison purposes

CRM predicted no collisions in the breeding season combined (April to August) for any of the wind turbine design options under consideration (Table 4.185).

CRM predicted no collisions in the post-breeding and non-breeding periods combined (September to March) for any of the wind turbine design options under consideration (Table 4.185).

Table 4.185 Number of predicted puffin collisions using avoidance rate of 98% for four wind farm options

	Option 1 (90x5MW)	Option 2 (75x6MW)	Option 3 (73x6.15MW)	Option 4 (73x6.15MW)
Collisions in breeding season (April to August), all ages	0	0	0	0
Collisions in post- and non-breeding period (September to March), all ages	0	0	0	0
Total collisions per year, all ages	0	0	0	0

It is concluded that for all four designs, collision mortality for puffin is an effect of negligible magnitude, that is temporally long-term and reversible.

It is further concluded that for the wind farm designs examined here, the effects of collision mortality on puffins from the regional population in the breeding, post-breeding and non-breeding periods are ***not significant*** under the EIA Regulations.

Decommissioning Phase

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 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.

Summary of combined effects

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 is low. It is concluded that the overall impact on the regional population of puffins in the breeding period is ***minor significant*** under the EIA regulations (Table 4.186).

Table 4.186 Summary of effects on the regional population of puffins during the 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, breeding period	Low	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***

In combination it is judged that the magnitude of the effects on the regional puffin population during the post-breeding and non-breeding periods is low. It is concluded that the overall impact on the regional population of puffins in the post-breeding and non-breeding periods is ***not significant*** under the EIA regulations (Table 4.187).

Table 4.187 Summary of effects on the regional population of puffins in the post-breeding and non-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, post-breeding period	Negligible	Long term	Moderate	**Not Significant**
Displacement from foraging habitat, non-breeding period	Low	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**

4.4.23.5 Cumulative Impact Assessment

Displacement

The potential cumulative displacement risk to puffins from Neart na Gaoithe and other plans or projects is higher than for Neart na Gaoithe alone.

Projects identified during consultation and the undertaking of the EIA for which there is a potential for a cumulative displacement impact are:

- Inch Cape offshore wind farm;
- Seagreen Project Alpha offshore wind farm; and
- Seagreen Project Bravo offshore wind farm.

Evidence from existing offshore wind farms support the potential for a moderate level of displacement behaviour (e.g. Leopold et al. 2011) and a predicted 40% displacement effect has been used in this assessment.

Breeding season

Site-specific surveys undertaken for Seagreen project Alpha recorded peak estimated numbers of puffins during the breeding period with a peak of 2,787 individuals, of which 1,967 birds were aged as adults (Seagreen 2013).

Assuming 40% of all adult puffins were displaced from the Seagreen project Alpha and a 2 km buffer during the breeding period, this would affect an estimated 787 breeding adults (Table 4.188).

Assuming that there is the potential for up to 10% rate of mortality during the breeding period then up to 79 breeding adult puffins may be impacted by displacement mortality from Seagreen project Alpha and a 2 km buffer (Table 4.188).

Table 4.188 Estimated number of adult puffins predicted to be at risk of mortality following displacement from Seagreen Project Alpha plus 2 km buffer in the breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	4	10	20	39	59	79	98	118	138	157	177	197
	20%	8	20	39	79	118	157	197	236	275	315	354	393
	30%	12	30	59	118	177	236	295	354	413	472	531	590
	40%	16	39	**79**	157	236	315	393	472	551	629	708	787
	50%	20	49	98	197	295	393	492	590	688	787	885	984
	60%	24	59	118	236	354	472	590	708	826	944	1,062	1,180
	70%	28	69	138	275	413	551	688	826	964	1,102	1,239	1,377
	80%	31	79	157	315	472	629	787	944	1,102	1,259	1,416	1,574
	90%	35	89	177	354	531	708	885	1,062	1,239	1,416	1,593	1,770
	100%	39	98	197	393	590	787	984	1,180	1,377	1,574	1,770	1,967
Two-year mean peak of 1,967 adult puffins in Seagreen Project Alpha & 2 km buffer in breeding period													
SPA breeding population within mean max foraging range (105.4 km) = 99,002 pairs (SMP 2013)													

Peak numbers of puffins within the Project Bravo study area also occurred during the breeding period but in higher numbers compared to Project Alpha with 5,439 individuals, of which 3,411 were considered to be adults (Seagreen 2013).

Assuming 40% of all adult puffins were displaced from the Seagreen project Bravo and a 2 km buffer during the breeding period, this would affect an estimated 1,364 breeding adults (Table 4.189).

Table 4.189 Estimated number of adult puffins predicted to be at risk of mortality following displacement from Seagreen Project Bravo plus 2 km buffer in the breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	7	17	34	68	102	136	171	205	239	273	307	341
	20%	14	34	68	136	205	273	341	409	478	546	614	682
	30%	20	51	102	205	307	409	512	614	716	819	921	1,023
	40%	27	68	**136**	273	409	546	682	819	955	1,092	1,228	1,364
	50%	34	85	171	341	512	682	853	1,023	1,194	1,364	1,535	1,706
	60%	41	102	205	409	614	819	1,023	1,228	1,433	1,637	1,842	2,047
	70%	48	119	239	478	716	955	1,194	1,433	1,671	1,910	2,149	2,388
	80%	55	136	273	546	819	1,092	1,364	1,637	1,910	2,183	2,456	2,729
	90%	61	153	307	614	921	1,228	1,535	1,842	2,149	2,456	2,763	3,070
	100%	68	171	341	682	1,023	1,364	1,706	2,047	2,388	2,729	3,070	3,411
Two year mean peak of 3,411 adult puffins in Seagreen Project Bravo & 2 km buffer in "chicks-at-sea" period													
SPA breeding population within mean max foraging range (105.4 km) = 99,002 pairs (SMP 2013)													

Assuming that there is the potential for up to 10% rate of mortality during the breeding period then up to 136 breeding adult puffins may be impacted by displacement mortality from Seagreen project Bravo and a 2 km buffer (Table 4.189).

Based on the above, the predicted cumulative displacement impacts on adult puffins from Neart na Gaoithe and Seagreen projects Alpha and Bravo and a 2 km buffer in the breeding period involve a predicted 4,566 adult birds, with 10% mortality of 456 adults. This corresponds to approximately 0.07% of the regional breeding SPA population within mean maximum foraging range of the offshore site (99,002 pairs). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

Inch Cape have not yet submitted their application however, they have provided provisional information (ICOL, 2013). The results indicate that up to 1,292 adult puffins may be displaced. Assuming 40% of all adult puffins were displaced from Inch Cape during the breeding season, this would affect an estimated 517 breeding adults (Table 4.190).

Table 4.190 Estimated number of adult puffins predicted to be at risk of mortality following displacement from Inch Cape in the breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	3	6	13	26	39	52	65	78	90	103	116	129
	20%	5	13	26	52	78	103	129	155	181	207	233	258
	30%	8	19	39	78	116	155	194	233	271	310	349	388
	40%	10	26	**52**	103	155	207	258	310	362	413	465	517
	50%	13	32	65	129	194	258	323	388	452	517	581	646
	60%	16	39	78	155	233	310	388	465	543	620	698	775
	70%	18	45	90	181	271	362	452	543	633	724	814	904
	80%	21	52	103	207	310	413	517	620	724	827	930	1,034
	90%	23	58	116	233	349	465	581	698	814	930	1,047	1,163
	100%	26	65	129	258	388	517	646	775	904	1,034	1,163	1,292
One year peak of 1,292 adult puffins in Inch Cape site in breeding period													
SPA breeding population within mean max foraging range (105.4 km) = 99,002 pairs (SMP 2013)													

Assuming that there is the potential for up to 10% rate of mortality during the breeding period then up to 52 breeding adult puffins may be impacted by displacement mortality from Inch Cape (Table 4.190).

Based on the above, the predicted cumulative displacement impacts on adult puffins from Neart na Gaoithe and a 2 km buffer and Inch Cape and a 4 km buffer in the breeding period involve a predicted 2,932 adult birds, with 10% mortality of 293 adults. This corresponds to approximately 0.1% of the regional breeding SPA population within mean maximum foraging range of the offshore site (99,002 pairs). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

Based on the predicted numbers reported for the four proposed development areas a total of 5,083 adult puffins may be displaced from the development sites and a 2 km buffer during the breeding period, with 10% mortality of 508 adult birds. The 10% mortality corresponds to approximately 0.3% of the regional breeding SPA population within mean maximum foraging range of the offshore site (99,002 pairs). It was considered that 10% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

Based on the above, this impact is categorised as having negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative effects of displacement on the regional puffin population in the breeding season are ***not significant*** under the EIA Regulations.

Post-breeding period

Peak estimated numbers within the Seagreen project Alpha site boundary occurred in September, with 1,420 birds in 2010, and 1,481 birds in 2011 (Seagreen 2012). This provides a peak mean across two years, of 1,451 puffins during the post-breeding period (September and October).

Assuming 40% of all puffins were displaced from the Seagreen project Alpha and a 2 km buffer during the post-breeding period, this would affect an estimated 580 birds (Table 4.191).

Assuming that there is the potential for up to 2% rate of mortality during the post-breeding period then up to 12 puffins may be impacted by displacement mortality from Seagreen project Alpha and a 2 km buffer (Table 4.191).

Table 4.191 Estimated number of puffins (all ages) predicted to be at risk of mortality following displacement from Seagreen Project Alpha plus 2 km buffer in the post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	3	7	15	29	44	58	73	87	102	116	131	145
	20%	6	15	29	58	87	116	145	174	203	232	261	290
	30%	9	22	44	87	131	174	218	261	305	348	392	435
	40%	**12**	29	58	116	174	232	290	348	406	464	522	580
	50%	15	36	73	145	218	290	363	435	508	580	653	726
	60%	17	44	87	174	261	348	435	522	609	696	784	871
	70%	20	51	102	203	305	406	508	609	711	813	914	1,016
	80%	23	58	116	232	348	464	580	696	813	929	1,045	1,161
	90%	26	65	131	261	392	522	653	784	914	1,045	1,175	1,306
	100%	29	73	145	290	435	580	726	871	1,016	1,161	1,306	1,451
Two-year peak of 1,451 puffins in Seagreen Project Alpha & 2 km buffer in post-breeding period													
Regional population in post-breeding season = 274,272 birds (revised from Mitchell *et al.* 2004)													

In the post-breeding period (September and October), peak estimated numbers within the Seagreen Project Bravo site boundary occurred in September 2011, when 5,370 birds were estimated (Seagreen 2012). There are no corresponding estimates available for the same period in 2010, therefore this figure was used in the assessment.

Assuming 40% of all puffins were displaced from the Seagreen project Bravo and a 2 km buffer during the post-breeding season, this would affect an estimated 2,148 birds (Table 4.192).

Table 4.192 Estimated number of puffins (all ages) predicted to be at risk of mortality following displacement from Seagreen Project Bravo plus 2 km buffer in the post-breeding season

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	11	27	54	107	161	215	269	322	376	430	483	537
	20%	21	54	107	215	322	430	537	644	752	859	967	1,074
	30%	32	81	161	322	483	644	806	967	1,128	1,289	1,450	1,611
	40%	**43**	107	215	430	644	859	1,074	1,289	1,504	1,718	1,933	2,148
	50%	54	134	269	537	806	1,074	1,343	1,611	1,880	2,148	2,417	2,685
	60%	64	161	322	644	967	1,289	1,611	1,933	2,255	2,578	2,900	3,222
	70%	75	188	376	752	1,128	1,504	1,880	2,255	2,631	3,007	3,383	3,759
	80%	86	215	430	859	1,289	1,718	2,148	2,578	3,007	3,437	3,866	4,296
	90%	97	242	483	967	1,450	1,933	2,417	2,900	3,383	3,866	4,350	4,833
	100%	107	269	537	1,074	1,611	2,148	2,685	3,222	3,759	4,296	4,833	5,370
One year peak of 5,370 puffins in Seagreen Project Bravo & 2 km buffer in post-breeding period													
Regional population in post-breeding season = 274,272 birds (revised from Mitchell *et al.* 2004)													

Assuming that there is the potential for up to 2% rate of mortality during the post-breeding period then up to 43 puffins may be impacted by displacement mortality from Seagreen project Bravo and a 2 km buffer (Table 4.192).

Based on the above, the predicted cumulative displacement impacts on puffins from Neart na Gaoithe and Seagreen projects Alpha and Bravo and a 2 km buffer in the post-breeding period involve a predicted 4,174 displaced birds, with 2% mortality of 84 birds. This figure corresponds to approximately 0.03% of the regional population in the post-breeding period (274,272 birds) (revised from Mitchell *et al.,* 2004). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this.

In the post-breeding period (September and October), peak estimated numbers of puffins within the Inch Cape site and a 4 km buffer area occurred in September 2010 (ICOL, 2012). There are no estimates for the number of puffins at this time, however reading across from presented figures it is estimated that approximately 900 individuals occurred in September 2010 (ICOL, 2012). There are no corresponding estimates available for the same period in 2011, therefore this figure was used in the assessment.

Assuming 40% of all puffins were displaced from Inch Cape and a 4 km buffer during the post- breeding season, this would affect an estimated 360 birds (Table 4.193).

Table 4.193 Estimated number of puffins (all ages) predicted to be at risk of mortality following displacement from Inch Cape plus 4 km buffer in the post-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	2	5	9	18	27	36	45	54	63	72	81	90
	20%	4	9	18	36	54	72	90	108	126	144	162	180
	30%	5	14	27	54	81	108	135	162	189	216	243	270
	40%	**7**	18	36	72	108	144	180	216	252	288	324	360
	50%	9	23	45	90	135	180	225	270	315	360	405	450
	60%	11	27	54	108	162	216	270	324	378	432	486	540

	70%	13	32	63	126	189	252	315	378	441	504	567	630
	80%	14	36	72	144	216	288	360	432	504	576	648	720
	90%	16	41	81	162	243	324	405	486	567	648	729	810
	100%	18	45	90	180	270	360	450	540	630	720	810	900
One year peak of 900 puffins in Inch Cape & 4 km buffer in post-breeding period													
Regional population in post-breeding season = 274,272 birds (revised from Mitchell *et al.* 2004)													

Assuming that there is the potential for up to 2% rate of mortality during the post-breeding period then up to seven puffins may be impacted by displacement mortality from Inch Cape and a 4 km buffer (Table 4.193).

Based on the above, the predicted cumulative displacement impacts on razorbills from Neart na Gaoithe and a 2 km buffer and Inch Cape and a 4 km buffer in the post-breeding period involve a predicted 1,806 birds, with 2% mortality of 36 birds. This mortality corresponds to approximately 0.01% of the regional population in the post-breeding period (274,272 birds) (revised from Mitchell *et al.,* 2004). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this. The estimate is also based on a 4 km buffer around Inch Cape, so estimated numbers of displaced razorbills are higher than they would be using a 2 km buffer. This has been taken into account in this assessment.

Based on the predicted numbers reported for the four proposed development areas a total of 4,531 puffins may be displaced from the development sites and a 2 km buffer (4 km for Inch Cape) during the post-breeding period, with 2% mortality of 91 birds. The 2% mortality corresponds to approximately 0.03% of the regional population in the post-breeding period (274,272 birds) (revised from Mitchell *et al.,* 2004). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be much lower than this. The estimate is also based on a 4 km buffer around Inch Cape, so estimated numbers of displaced razorbills are higher than they would be using a 2 km buffer. This has been taken into account in this assessment.

Based on the above, this impact is categorised as having negligible magnitude, temporally long-term and reversible (Table 3.1 and Table 3.2). It is concluded that the cumulative effects of displacement on the regional puffin population in the post-breeding season are ***not significant*** under the EIA Regulations.

A separate CIA for displacement has not been undertaken for the regional puffin population in the non-breeding period (November to March), as data was not available for these months for the Seagreen or Inch Cape projects. The predicted displacement of puffin 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 considered to be low. Therefore it is not plausible that the Neart na Gaoithe development could contribute to a significant cumulative impact for the regional populations during this period.

Collision mortality

No significant impacts were predicted to arise from collision mortality caused by Neart na Gaoithe for puffins in the breeding, post-breeding or non-breeding periods. This was based on zero bird collisions predicted by collision risk modelling, indicating very low levels of collision mortality for this species. It is therefore not plausible that Neart na Gaoithe could contribute to a significant cumulative collision mortality impact for the regional population of puffins in the breeding, post-breeding or non-breeding periods. As a result, no further cumulative impact assessment for collision mortality was undertaken for this species.

4.4.23.6 Mitigation measures

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

4.4.24 Little auk *Alle alle*

4.4.24.1 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).

4.4.24.2 Offshore site and 8 km buffer area

Little auks were only recorded in the offshore site and 8 km buffer area between November and April. Between November and February of Year 1, 135 little auks were recorded in the offshore site and 8 km buffer area, with 26 birds seen in the offshore site (Table 4.2) (raw numbers, all sea states). Numbers recorded on surveys between December and February of Year 2 were similar (113 birds), with 16 birds seen in the offshore site. In Year 3, a total of 2,710 little auks were recorded on surveys between November and April, with 415 birds seen in the offshore site.

In the Year 1 non-breeding season (November to February), the mean estimated number of little auks in the offshore site was 141 birds, with a peak of 425 birds in November (Table 4.194). In the same period in Year 2, the mean estimated number of little auks was 47 birds, with a peak of 96 birds in December, although there was no November survey. In Year 3, the mean estimated number of little auks in the offshore site was 2,503 birds, with a peak of 5,844 birds in January.

Table 4.194 Estimated numbers of little auks in the offshore site (and 1, 2 & 8 km buffer) between November and March of Years 1 to 3

Month	Offshore Site					Estimated total offshore site + 1 km	Estimated total offshore site + 2 km	Estimated total offshore site + 8 km
	Estimated nos on water	Lower 95 % C.L.	Upper 95 % C.L.	Estimated nos flying	Estimated total			
Yr1 Nov	412	255	664	14	425	592	798	1,397
Yr1 Dec	0	0	0	0	0	0	0	161
Yr1 Jan	0	0	0	0	0	0	0	0
Yr1 Feb	137	74	255	0	137	251	274	1,166
Yr1 Mar	0	0	0	0	0	0	0	0
Yr1 Apr	0	0	0	0	0	0	0	0
Yr1 May	0	0	0	0	0	0	0	0
Yr1 Jun	0	0	0	0	0	0	0	0
Yr1 Jul	0	0	0	0	0	0	0	0
Yr1 Aug	0	0	0	0	0	0	0	0
Yr1 Sep	0	0	0	0	0	0	0	0
Yr1 Oct	0	0	0	0	0	0	0	0
Yr2 Nov	-	-	-	-	-	-	-	-
Yr2 Dec	96	47	197	0	96	224	272	635
Yr2 Jan	46	21	104	0	46	53	76	513
Yr2 Feb	0	0	0	0	0	0	0	0
Yr2 Mar	0	0	0	0	0	0	0	0
Yr2 Apr	0	0	0	0	0	0	0	0
Yr2 May	0	0	0	0	0	0	0	0
Yr2 Jun	0	0	0	0	0	0	0	0
Yr2 Jul	0	0	0	0	0	0	0	0
Yr2 Aug	0	0	0	0	0	0	0	0
Yr2 Sep	0	0	0	0	0	0	0	0
Yr2 Oct	0	0	0	0	0	0	0	0
Yr3 Nov	1,487	1,024	2,160	21	1,508	2,287	2,951	10,120
Yr3 Dec	-	-	-	-	-	-	-	-
Yr3 Jan	5,715	4,130	7,908	130	5,844	8,792	12,603	36,195
Yr3 Feb	158	89	280	0	158	158	203	474
Yr 3 Mar	143	87	235	7	150	198	221	610
Yr3 Apr	0	0	0	0	0	0	0	0
Yr3 May	0	0	0	0	0	0	0	0
Yr3 Jun	0	0	0	0	0	0	0	0
Yr3 Jul	0	0	0	0	0	0	0	0
Yr3 Aug	0	0	0	0	0	0	0	0
Yr3 Sep	0	0	0	0	0	0	0	0
Yr3 Oct	0	0	0	0	0	0	0	0

Mean estimated numbers of little auks in the offshore site were highest in November (1,440 birds, three-year mean) and January (2,948 birds, three-year mean), with lower numbers in December, February and March (Figure 4.132). A similar pattern was recorded in the buffer area, although estimated numbers were considerably higher in November (4,792 birds, three-year mean) and particularly in January (10,273 birds, three-year mean).

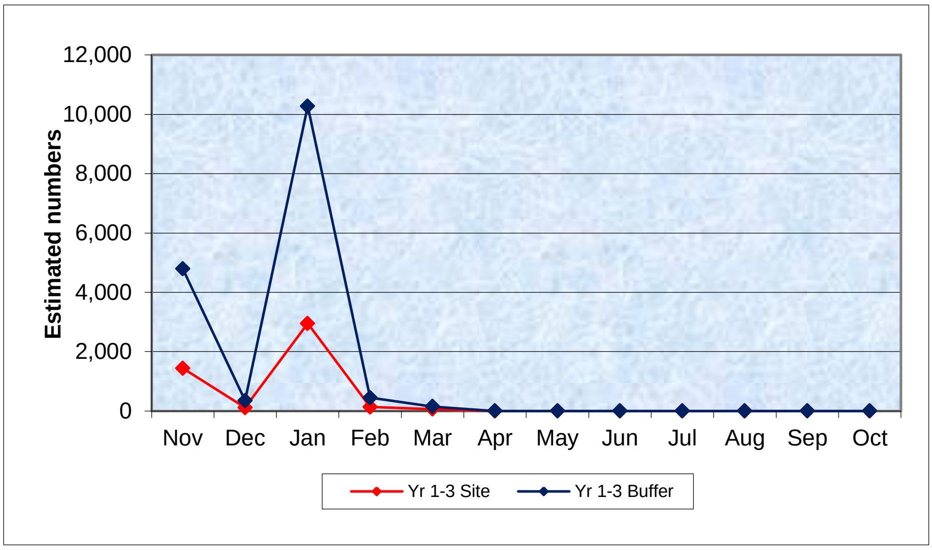


Figure 4.132 Mean monthly estimated numbers of little auks in the offshore site & 8 km buffer area in Years 1 to 3

Between November and February of Year 1, little auks were relatively 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 offshore site and buffer area (Figure 4.133).

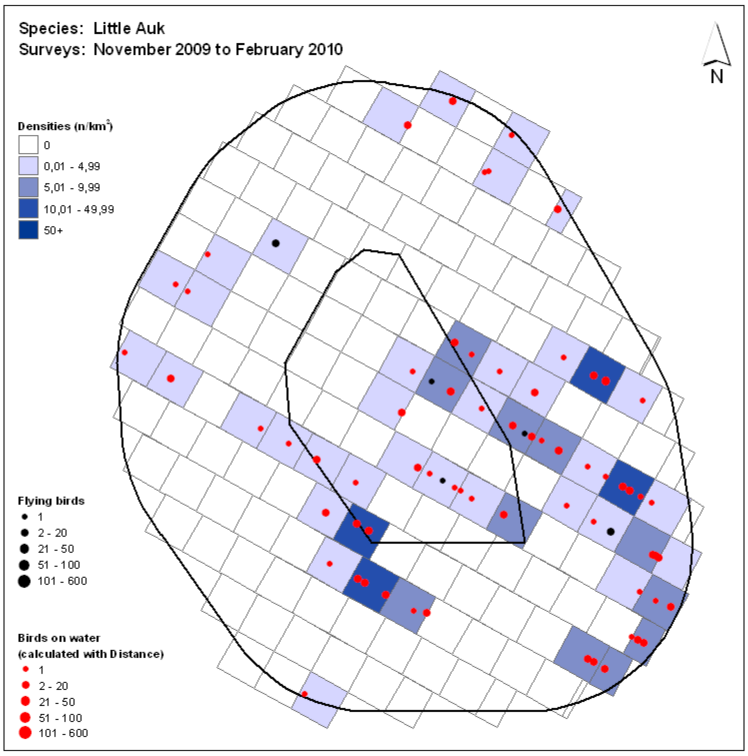


Figure 4.133 Little auk density between November and February, Year 1

Little auks were less widespread in the offshore site over the same period in Year 2 (Figure 4.134). Birds were scattered across the buffer area at low to moderate densities at this time.

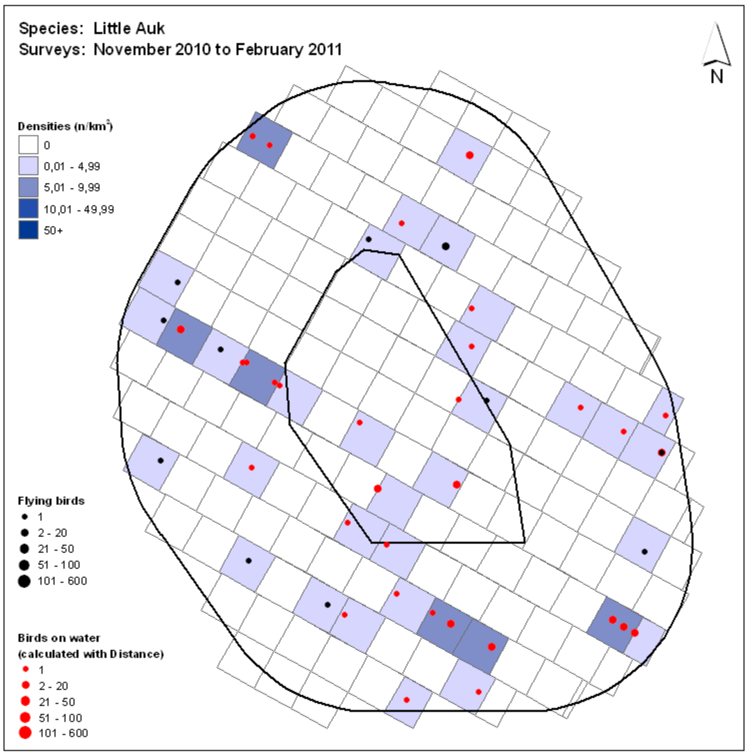


Figure 4.134Little Auk density between November and February, Year 2

Little auks were widespread at mostly high densities in the offshore site and buffer area over the same period in Year 3 (Figure 4.135).

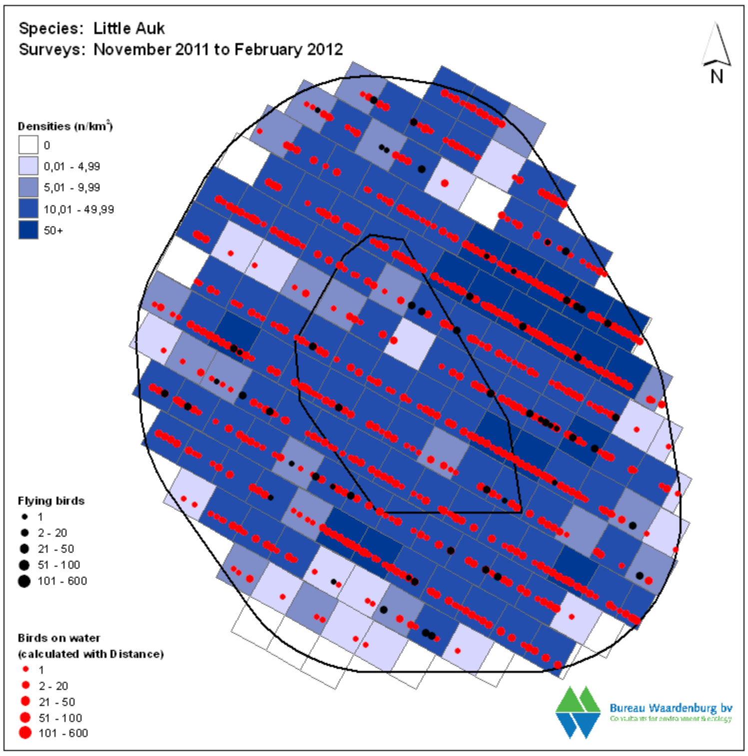


Figure 4.135Little Auk density between November and February, Year 3

A total of 631 little auks were recorded in flight on baseline surveys, with all birds below 27.5 m in height (Table 4.3).

4.4.24.3 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).

4.4.24.4 Assessment

Definition of seasons

Little auks were mainly recorded in the offshore site between November and February and therefore this was the period considered for this assessment.

Populations

Little auks are a winter visitor to the seas around the UK from Arctic breeding grounds. Numbers of little auks recorded off the east coast of Britain in the winter months varies considerably between years, with very high numbers recorded in some winters. These fluctuations are largely dependant on the weather (Pollock *et al.,* 1996). 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.

The peak winter counts (raw numbers) from three proposed wind farm survey areas (Neart na Gaoithe: 2,710 birds; Inch Cape: 126 birds in Year 1; and Seagreen R3: approximately 2,358 birds in Year 1) suggests a minimum regional population of 5,194 birds. However, surveys did not cover the whole area around these wind farm developments. The three-year mean estimated number for the offshore site and 8 km buffer in the non-breeding period was 12,742 birds. However, this figure was largely influenced by the very large number of little auks estimated in the offshore site and buffer area in January 2012 (36,195 birds). No survey data for Inch Cape or Seagreen R3 for the 2012 winter period was available for comparison. For the purposes of this assessment, such large numbers were considered unusual, and a regional winter population of 7,500 birds was assumed.

Nature conservation importance

For EIA assessment purposes, the nature conservation importance of little auk using the offshore site is rated as low during the non-breeding season. This species is not subject to any special legislative protection, nor is it on any conservation priority lists.

Offshore wind farm studies

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.

Construction Phase assessment

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.

Operational Phase

In order to assess the potential impacts from displacement, the approach presented in the interim guidance note on displacement issued by the JNCC and NE has been followed (JNCC & NE 2012).

Peak estimated numbers of little auks in the offshore site in the non-breeding period (November to February) for Years 1 to 3 were averaged to get the three-year mean peak per season. This was repeated for a 1 km and 2 km buffer around the offshore site (Table 4.195).

Table 4.195 Seasonal three-year mean peak estimated numbers of little auks in the offshore site (plus 1 & 2 km buffer)

Year	Site	Site + 1 km	Site + 2 km
Year 1	425	592	798
Year 2	96	224	272
Year 3	5,844	8,792	12,603
3-year mean peak	2,122	3,203	4,558

The three-year mean peak estimated number of little auks was then used to predict the estimated number of birds at potential risk of mortality following displacement in the non-breeding period in the offshore site (plus 1 km and 2 km buffers), as recommended in the draft guidance note on displacement (JNCC & NE, 2012).

Likely impacts of displacement

Non-breeding period

For this assessment, it was assumed that there will be 40% displacement of little auks from the offshore site in the non-breeding period. Additional scenarios considering 40% displacement out to a 1 km and a 2 km buffer are also presented.

Assuming 40% of all little auks were displaced from the offshore site during the non-breeding period (November to February), this would affect an estimated 849 birds (Table 4.196), increasing to 1,823 birds if displacement is assumed to affect the offshore site and a 2 km buffer (Table 4.198). For the purposes of this assessment, it was assumed that 2% of all little auks displaced from the offshore site and 2 km buffer during the non-breeding period (up to 36 birds) would die as a result. This corresponds to approximately 0.5% of the estimated regional population in the non-breeding period (7,500 birds). It was considered that 2% mortality was a precautionary estimate, and that the actual mortality rate as a direct result of displacement would be lower than this.

It was concluded that the remaining little auks displaced from the offshore site and 2 km buffer would move to alternative foraging areas in the non-breeding period.

Comparing the distribution of little auks in the offshore site from Years 1 to 3 showed that between November and February of Year 1, little auks were scattered at mostly low densities in the south and east of the offshore site and buffer area, with fewer birds in the west and north (Figure 4.133). Little auks were less widespread in the offshore site in the same period in Year 2 (Figure 4.134). In the same period in Year 3, little auks were widespread at mostly high desnities throughout the offshore site and buffer area (Figure 4.135). This demonstrates that little auk distribution is very variable between winters, and also suggests that little auks are not regularly relying solely on the offshore site at this time of year. It is concluded that any displacement effects from this area will be small.

Table 4.196 Estimated number of little auks at risk of mortality following displacement from offshore site in non-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	4	11	21	42	64	85	106	127	149	170	191	212
	20%	8	21	42	85	127	170	212	255	297	340	382	424
	30%	13	32	64	127	191	255	318	382	446	509	573	637
	40%	**17**	42	85	170	255	340	424	509	594	679	764	849
	50%	21	53	106	212	318	424	531	637	743	849	955	1,061
	60%	25	64	127	255	382	509	637	764	891	1,019	1,146	1,273
	70%	30	74	149	297	446	594	743	891	1,040	1,188	1,337	1,485
	80%	34	85	170	340	509	679	849	1,019	1,188	1,358	1,528	1,698
	90%	38	95	191	382	573	764	955	1,146	1,337	1,528	1,719	1,910
	100%	42	106	212	424	637	849	1,061	1,273	1,485	1,698	1,910	2,122
Three-year mean peak of 2,122 little auks in the offshore site in non-breeding period													
Regional population in the non-breeding season = 7,500 birds													

Table 4.197 Estimated number of little auks at risk of mortality following displacement from offshore site plus 1 km buffer in non-breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	6	16	32	64	96	128	160	192	224	256	288	320
	20%	13	32	64	128	192	256	320	384	448	512	577	641
	30%	19	48	96	192	288	384	480	577	673	769	865	961
	40%	**26**	64	128	256	384	512	641	769	897	1,025	1,153	1,281
	50%	32	80	160	320	480	641	801	961	1,121	1,281	1,441	1,602
	60%	38	96	192	384	577	769	961	1,153	1,345	1,537	1,730	1,922
	70%	45	112	224	448	673	897	1,121	1,345	1,569	1,794	2,018	2,242
	80%	51	128	256	512	769	1,025	1,281	1,537	1,794	2,050	2,306	2,562
	90%	58	144	288	577	865	1,153	1,441	1,730	2,018	2,306	2,594	2,883
	100%	64	160	320	641	961	1,281	1,602	1,922	2,242	2,562	2,883	3,203
Three-year mean peak of 3,203 little auks in the offshore site in non-breeding period													
Regional population in the non-breeding season = 23,633 birds (Skov *et al*., 1995)													

Table 4.198 Estimated number of little auks at risk of mortality following displacement from offshore site plus 2 km buffer in non -breeding period

	Mortality (%)												
Displacement level (%)	0%	2%	5%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
	10%	9	23	46	91	137	182	228	273	319	365	410	456
	20%	18	46	91	182	273	365	456	547	638	729	820	912
	30%	27	68	137	273	410	547	684	820	957	1,094	1,231	1,367
	40%	**36**	91	182	365	547	729	912	1,094	1,276	1,459	1,641	1,823
	50%	46	114	228	456	684	912	1,140	1,367	1,595	1,823	2,051	2,279
	60%	55	137	273	547	820	1,094	1,367	1,641	1,914	2,188	2,461	2,735
	70%	64	160	319	638	957	1,276	1,595	1,914	2,233	2,552	2,872	3,191
	80%	73	182	365	729	1,094	1,459	1,823	2,188	2,552	2,917	3,282	3,646
	90%	82	205	410	820	1,231	1,641	2,051	2,461	2,872	3,282	3,692	4,102
	100%	91	228	456	912	1,367	1,823	2,279	2,735	3,191	3,646	4,102	4,558
Three-year mean peak of 796 puffins in the offshore site in non-breeding period													
Regional population in the non-breeding season = 23,633 birds (Skov *et al*., 1995)													

The displacement impact in the non-breeding season is therefore 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. It is concluded that the effects of displacement on the regional little auk 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.

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 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.

Decommissioning Phase

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.

Summary of combined effects

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 4.199).

Table 4.199 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***

4.4.24.5 Cumulative Impact Assessment

There were no significant impacts predicted for the regional population of little auks in the breeding or non-breeding periods from the proposed Neart na Gaoithe development. The predicted effects of the development on the regional population of little auks in the breeding or non-breeding periods arising from construction, operation and decommissioning are very close to no effect.

Based on these findings, it was considered that it is not plausible that Neart na Gaoithe could contribute to a significant cumulative impact for this population, and therefore no further cumulative impact assessment was undertaken for this species.

4.4.24.6 Mitigation measures

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

4.4.25 Non-seabirds

A total of 1,209 birds of 19 species of non-seabird were recorded in the offshore site and 8 km buffer area in Year 1. In Year 2, 424 birds of 22 species were recorded on baseline surveys. In Year 3, 64 birds of 13 species were recorded on baseline surveys.

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. In Year 3, the three most frequently recorded species were pink-footed goose, sanderling and teal, accounting for 65.6% of all non-seabirds recorded.

In all three 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, 3.5% recorded in the offshore site in Year 2, and 6.3% recorded in the offshore site in Year 3.

A brief summary of the non-seabird species recorded on baseline surveys is given below.

4.4.25.1 Grey Heron

Grey herons are typically found on lakes, rivers and estuaries and would be unlikely to occur regularly offshore. One grey heron was recorded in the buffer area, flying below 7.5 m in August of Year 3.

4.4.25.2 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.

4.4.25.3 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 offshore site and 8 km buffer area in Year 1, with 90 birds in November 2010, 34 in February 2010, 1 in March 2010 and 91 in September 2010. 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. In Year 3, 28 were recorded flying through the buffer area in September.

Less than half of all pink-footed geese (45.4%) were recorded flying above 27.5 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.

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 4.200).

4.4.25.1 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 offshore site and 8 km buffer area on 12th October of Year 1. 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 Years 2 and 3.

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.

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 4.200). 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 4.200 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	
Teal	Loch of Strathbeg	160	1,898	1.4	0.5
	Loch Leven	65	2,483	1.8	0.6
Shoveler	Loch Leven	65	520	5.2	1.3
Species	**SPA site**	**Distance to site (km)**	**Site total**	**% of national popn**	**% of biogeo graphic popn**
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 (2013) (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)

4.4.25.2 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 offshore site and 8 km buffer 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.

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 4.200).

4.4.25.3 Teal

Teal are a common and widespread passage migrant and winter visitor to Scotland, with a small breeding population (Forrester *et al.,* 2007).

A flock of six teal were recorded flying below 7.5 m in height in the buffer area in September of Year 3.

Teal 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 4.200).

4.4.25.4 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.

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 4.200).

4.4.25.5 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.

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 4.200).

4.4.25.6 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.

4.4.25.7 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.

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 4.200).

4.4.25.8 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.

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 4.200).

4.4.25.9 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.

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 4.200).

4.4.25.10 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.

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 4.200).

4.4.25.11 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.

4.4.25.12 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.

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 4.200).

4.4.25.13 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. 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 4.200).

4.4.25.14 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.

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 4.200).

4.4.25.15 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.

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 4.200).

4.4.25.16 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.

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 4.200).

4.4.25.17 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.

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 4.200).

4.4.25.18 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.

4.4.25.19 Passerines

Six species of passerines (or perching birds) were recorded in the offshore site and 8 km buffer area in Year 1, with three species seen in the offshore site; one meadow pipit, one blackbird and three skylarks. 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. 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.

In Year 3, no passerines were recorded in the offshore site in Year 3. Five species were recorded in the buffer area; 4 swallows, 3 house martins, 1 meadow pipit, 2 blackbirds and 2 starlings.

Overall, meadow pipit was the only species of land bird with a sample size greater than 20 birds on baseline surveys. Of the 58 meadow pipits recorded, 57 (98.3%) were flying below 27.5 m in height (Table 4.3).

4.4.25.20 Assessment

The majority of the non-seabird species recorded on baseline surveys in the offshore site and buffer 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 4.201. Further details are presented in Ornithology Appendix 2: Collision Rate Estimates of Seabirds at Neart na Gaoithe.

Table 4.201 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 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.

4.4.25.21 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;

Table A1 **Sea state during surveys in the offshore site and 8 km buffer area on baseline surveys**

Month	Year 1			Year 2			Year 3		
	Sea State	Km	%	Sea State	Km	%	Sea State	Km	%
November	1	102.9	31.4	1	-	-	1	4.6	1.5
	2	224.6	68.6	2			2	107.9	35.4
	3	0	0	3	-	-	3	82.6	27.1
	4	0	0	4	-	-	4	109.5	35.9
December	0	18.4	5.7	0	0	0	-	-	-
	1	113.8	35.0	1	0	0	-	-	-
	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	1	0	0	1	0	0	1	13.0	4.2
	2	112.4	34.6	2	42.2	13.0	2	96.6	31.6
	3	121.0	37.3	3	231.7	71.4	3	131.6	43.0
	4	78.9	24.3	4	50.6	15.6	4	64.8	21.2
	5	12.4	3.8	5	0	0	5	0	0
February	1	0	0	1	47.2	14.2	1	97.9	31.9
	2	129.7	39.6	2	99.9	30.1	2	67.7	22.1
	3	150.1	45.8	3	150.6	45.4	3	88.9	29.0
	4	35.7	10.9	4	34.2	10.3	4	51.9	16.9
	5	12.2	3.7	5	0	0	5	0	0
March	0	135.4	41.0	0	61.3	18.3	0	0	0
	1	60.0	18.2	1	101.1	30.2	1	35.8	11.7
	2	134.7	40.8	2	64.8	19.3	2	103.9	33.8
	3	0	0	3	93.5	27.9	3	132.7	43.2
	4	0	0	4	13.1	3.9	4	34.5	11.2
	5	0	0	5	1.2	0.4	5	0	0
April	0	54.5	16.7	0	45.9	13.9	0	79.1	25.7
	1	174.3	53.3	1	52.8	16.1	1	44.8	14.6
	2	98.5	30.1	2	103.3	31.4	2	105.5	34.3
	3	0	0	3	77.4	23.5	3	57.3	18.6
	4	0	0	4	45.4	13.8	4	20.3	6.6
	5	0	0	5	3.8	1.2	5	0.8	0.2

Table A1 Sea state during surveys in the offshore site and 8 km buffer area on baseline surveys (continued)

Month	Year 1			Year 2			Year 3		
	Sea State	Km	%	Sea State	Km	%	Sea State	Km	%
May	0	0	0	0	0	0	0	9.7	3.2
	1	37.3	11.5	1	13.2	3.8	1	37.5	12.2
	2	198.6	60.6	2	166.7	48.6	2	72.8	23.7
	3	76.4	23.3	3	72.2	21.0	3	116.5	37.9
	4	15.4	4.7	4	78.3	22.8	4	70.9	23.1
	5	0	0	5	0.3	0.1	5	0	0
June	0	13.7	4.2	0	0	0	0	0	0
	1	37.0	11.3	1	13.1	4.0	1	0	0
	2	50.6	15.5	2	78.1	23.7	2	98.0	31.9
	3	198.1	60.6	3	235.6	71.6	3	167.8	54.7
	4	27.4	8.4	4	2.1	0.6	4	40.9	13.3
July	1	0	0	1	64.6	19.6	1	0	0
	2	67.8	20.7	2	176.8	53.6	2	0	0
	3	146.5	44.6	3	88.5	26.8	3	163.4	53.0
	4	106.5	32.4	4	0	0	4	90.4	29.3
	5	7.8	2.4	5	0	0	5	54.6	17.7
August	0	16.8	5.1	0	29.0	9.3	0	0	0
	1	55.2	16.8	1	191.5	61.6	1	102.3	31.7
	2	136.3	41.3	2	90.6	29.1	2	96.1	29.8
	3	121.6	36.9	3	0	0	3	124.3	38.5
September	0	21.5	6.6	0	4.5	1.4	0	0	0
	1	9.8	3.0	1	108.5	34.5	1	22.0	14.5
	2	158.8	48.4	2	148.3	47.1	2	84.3	55.4
	3	105.8	32.3	3	41.9	13.3	3	45.8	30.1
	4	32.2	9.8	4	11.4	3.6	4	0	0
October	0	57.0	17.3	0	9.3	3.0	0	0	0
	1	164.5	49.9	1	37.3	12.0	1	3.1	1.0
	2	95.5	29.0	2	140.0	45.0	2	75.6	24.5
	3	12.5	3.8	3	90.2	29.0	3	100.8	32.7
	4	0	0	4	34.1	11.0	4	122.2	39.6
	5	0	0	5	0	0	5	6.7	2.2

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 offshore site in Year 3 (Raw numbers, all sea states)

Species	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Total
Fulmar	0	-	11	9	10	7	2	3	13	9	20	3	87
Sooty shearwater	0	-	0	0	0	0	0	0	1	0	0	0	1
Manx shearwater	0	-	0	0	0	0	0	0	7	20	0	0	27
Gannet	47	-	8	23	123	81	393	465	219	487	168	120	2,134
Pomarine skua	2	-	0	0	0	0	0	0	1	0	0	0	3
Great skua	1	-	1	0	0	0	0	0	0	0	0	0	2
Little gull	0	-	0	0	0	0	0	0	0	0	39	4	43
Common gull	1	-	0	1	0	0	0	0	0	0	0	0	2
Lesser black-backed gull	0	-	0	0	0	0	20	3	8	6	0	0	37
Herring gull	3	-	3	5	0	3	15	19	3	3	0	0	54
Great black-backed gull	5	-	6	1	1	0	0	2	0	0	2	0	17
Large gull species	0	-	1	0	0	0	0	1	80	0	0	0	82
Kittiwake	65	-	21	8	20	6	31	115	68	479	8	17	838
Arctic tern	0	-	0	0	0	0	0	0	6	84	0	0	90
Guillemot	141	-	89	115	85	171	193	106	87	530	168	84	1,769
Razorbill	16	-	9	3	3	8	28	6	22	125	45	13	278
Little auk	84	-	311	7	13	0	0	0	0	0	0	0	415
Puffin	6	-	4	4	100	334	131	104	367	60	55	31	1,196
Guillemot/razorbill	35	-	36	29	6	0	1	0	25	30	39	12	213
Unidentified auk species	3	-	0	2	0	1	0	0	0	0	1	0	7
Total numbers	409	-	500	207	361	611	814	824	907	1,833	545	284	**7,295**

Table A5 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 A6 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 A7 Numbers of seabirds recorded in the Neart na Gaoithe buffer area in Year 3 (Raw numbers, all sea states)

Species	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Total
Fulmar	7	-	55	46	40	23	18	25	38	56	83	13	404
Sooty shearwater	0	-	0	0	0	0	0	0	1	5	5	0	11
Manx shearwater	1	-	0	1	0	0	0	3	22	124	1	0	152
Balearic shearwater	0	-	0	0	0	0	0	0	0	0	0	1	1
Storm petrel	0	-	0	0	0	0	0	1	0	0	0	0	1
Gannet	438	-	174	195	861	883	1,599	3,083	1,805	1,879	1,169	605	12,691
Cormorant	0	-	0	3	0	0	0	0	0	0	0	0	3
Shag	0	-	0	0	2	0	1	0	0	0	0	0	3
Eider	3	-	3	1	0	0	0	0	0	0	0	0	7
Common scoter	0	-	0	0	0	0	0	0	0	0	0	1	1
Grey phalarope	3	-	0	0	0	0	0	0	0	0	0	0	3
Pomarine skua	19	-	0	0	0	0	0	0	0	0	0	0	19
Arctic skua	3	-	0	0	0	0	0	2	0	0	0	0	5
Great skua	10	-	0	0	0	0	0	0	1	1	3	3	18
Unidentified skua species	1	-	0	0	0	0	0	0	0	0	0	0	1
Little gull	0	-	0	2	0	0	0	0	0	3	352	22	379
Black-headed gull	1	-	0	0	0	0	0	0	0	0	0	0	1
Common gull	9	-	1	2	0	0	0	1	0	3	1	3	20
Lesser black-backed gull	0	-	0	0	1	9	15	28	43	37	0	1	134
Herring gull	153	-	136	176	63	9	45	75	61	21	1	5	745
Great black-backed gull	99	-	28	26	16	2	9	3	4	5	8	8	208
Large gull species	324	-	3	141	124	0	2	10	27	0	0	3	634
Kittiwake	663	-	152	47	115	115	272	271	346	1,067	206	208	3,462
Sandwich tern	0	-	0	0	0	0	1	1	0	0	0	0	2
Common tern	0	-	0	0	0	0	0	0	0	1	0	0	1
Arctic tern	0	-	0	0	0	0	1	0	16	442	0	0	459
Common/Arctic tern	0	-	0	0	0	0	0	1	2	3	0	0	6
Guillemot	852	-	628	553	602	1,603	813	620	275	2,696	577	569	9,788
Razorbill	55	-	40	20	48	72	90	21	86	731	355	119	1,637
Little auk	458	-	1,766	16	54	1	0	0	0	0	0	0	2,295
Puffin	52	-	20	12	422	964	579	543	1,398	460	225	112	4,787
Guillemot/razorbill	45	-	103	58	42	11	23	52	10	1,017	106	87	1,554
Unidentified auk species	21	-	4	13	6	5	18	72	21	1	16	2	179
Total numbers	3,217	-	3,113	1,312	2,396	3,697	3,486	4,812	4,156	8,552	3,108	1,762	39,611

Table A8 Numbers of non-seabird species recorded in the offshore site and 8 km buffer 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 A9 Numbers of non-seabird species recorded in the offshore site and 8 km buffer 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**

Table A10 Numbers of non-seabird species recorded in the offshore site and 8 km buffer area in Year 3

Species	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Total
Grey heron	0	0	0	0	0	0	0	0	0	1	0	0	1
Pink-footed goose	0	0	0	0	0	0	0	0	0	0	28	0	28
Teal	0	0	0	0	0	0	0	0	0	0	6	0	6
Ringed plover	0	0	0	0	0	0	1	0	0	1	0	0	2
Sanderling	0	0	0	0	0	0	8	0	0	0	0	0	8
Dunlin	0	0	0	3	0	0	0	0	0	0	0	0	3
Curlew	0	0	0	0	0	0	0	0	0	0	0	1	1
Swallow	0	0	0	0	0	0	3	2	0	0	0	0	5
House martin	0	0	0	0	0	0	3	0	0	0	0	0	3
Meadow pipit	1	0	0	0	0	0	0	0	0	0	1	0	2
Blackbird	2	0	0	0	0	0	0	0	0	0	0	0	2
Starling	1	0	0	0	2	0	0	0	0	0	0	0	3
Total numbers	4	0	0	3	2	0	15	2	0	2	35	1	64