From: Gemma Keenan
To: Tracey Williams
Cc: [redacted]
Subject: Norfolk Vanguard - Email 4 of 18 Deadline 1 Submissions
Date: 16 January 2019 14:53:35
Attachments: ExA;WQApp01.1;10.D1.3 Norfolk Vanguard WQ Appendix 1.1 Horlock Rules.pdf
ExA;WQApp03.01;10.D1.3 Norfolk Vanguard WQ Appendix 3.1 Red-Throated Diver Displacement.pdf
ExA;WQApp03.02;10.D1.3 Norfolk Vanguard WQ Appendix 3.2 CRM update and clarification.pdf
ExA;WQApp03.03;10.D1.3 Norfolk Vanguard WQ Appendix 3.3 Auk gannet displacement.pdf
ExA;WQApp03.04;10.D1.3 Norfolk Vanguard WQ Appendix 3.4 Furness et al. 2013.pdf

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Dear Tracey

This is email 4 of 18 of the Applicant's submission for Norfolk Vanguard Examination Deadline 1.

We enclose the following documents:

Appendices to Written Questions:

- Appendix 1.1 Horlock Rules
- Appendix 3.1 Red-throated diver displacement
- Appendix 3.2 Collision Risk Modelling: update and clarification
- Appendix 3.3 Operational Auk and Gannet Displacement: update and clarification
- Appendix 3.4 Furness, B, MacArthur, D., Trinder, M. & MacArthur, K. (2013) Evidence review to support the identification of potential conservation measures for selected species of seabirds. Report to DEFRA.

Please could you kindly confirm receipt.

Best Regards

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Norfolk Vanguard Offshore Wind Farm

The Applicant Responses to First Written Questions

Appendix 1.1 – Horlock Rules
(Q 1.4)

Applicant: Norfolk Vanguard Limited
Document Reference: ExA;WQApp1.1;10.D1.3
Deadline 1

Date: January 2019

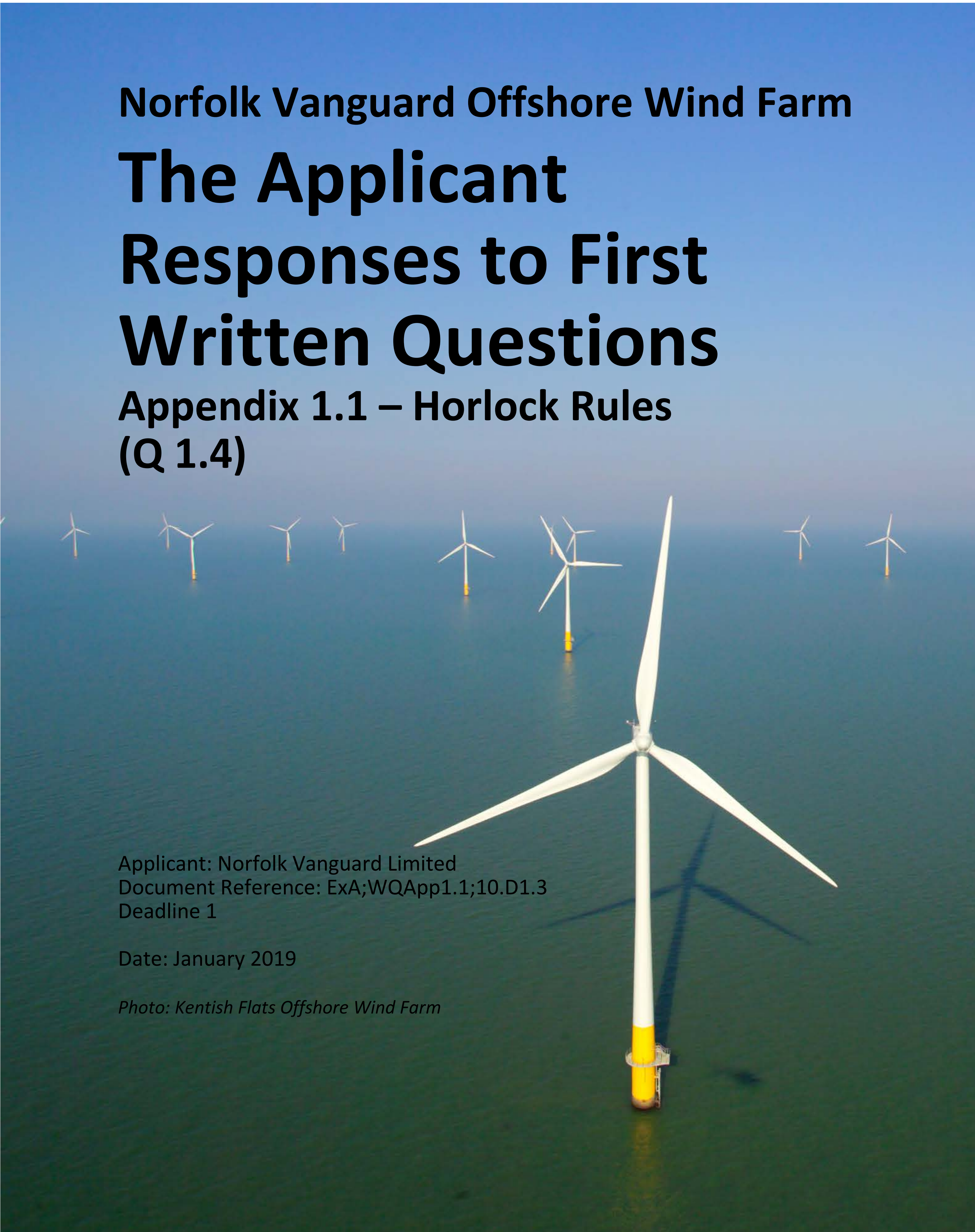

Photo: Kentish Flats Offshore Wind Farm

THE NATIONAL GRID COMPANY plc

NGC SUBSTATIONS AND THE ENVIRONMENT: GUIDELINES ON SITING AND DESIGN

Section 1 INTRODUCTION

1 The National Grid Company plc's (NGC's) policy statement on the environment recognises the importance of giving due regard to protecting and enhancing the environment and taking into account the environmental effects of the Company's actions. The Company has statutory duties in relation to preservation of amenity under Schedule 9 of the Electricity Act 1989, and has published a Schedule 9 Statement setting out the manner in which it proposes to meet these duties.

2 NGC has a statutory duty under the Act to develop and maintain an efficient, co-ordinated and economical transmission system of electricity for England and Wales. New transmission lines, new substations, sealing end compounds, line entries, additions and extensions to existing substations may be required to provide new connections for customers or reinforcement of the national grid system arising from changes in the demand for and generation of electricity.

3 This document explains the approach NGC takes towards such developments (Section II) and contains Guidelines (Section III) to assist those responsible for siting and designing substations to mitigate the environmental effects of such developments and so meet the Company's policy. The document complements the Company's Holford Rules guidelines on the routeing of high voltage transmission lines and when appropriate should be used in conjunction with them.

4 The guidelines are to be used by NGC staff, their consultants, and contractors in the siting and design of new substations and extensions to substations. They reflect the criteria the company requires its staff, consultants and contractors to satisfy.

5 As recognised in its Schedule 9 Statement NGC places importance on consultation with statutory planning and amenity bodies over its proposals for new developments. NGC believes that the availability of these guidelines will assist in such discussions by referring to the main considerations relevant to substation siting, and will thereby assist in achieving the most appropriate siting and design solutions.

Section II NGC'S APPROACH TO DESIGN AND SITING OF SUBSTATIONS

Approach to the Environment

6 NGC's environmental policy recognises the importance of giving due regard to protecting and enhancing the environment and taking into account the effect on the environment of all the Company's actions. Following the principle of integrating environmental considerations into all its activities, NGC seeks to keep known adverse effects on the environment to a reasonably practicable minimum and, in accordance with its duties under Schedule 9 of the Electricity Act, the Company gives due regard to the preservation of amenity and takes reasonable steps to mitigate the effects of its relevant proposals. To achieve these aims the Company therefore has to balance technical, economic and environmental considerations to reach reasonably practicable development proposals.

7 The guidelines (Section III) deal with the amenity issues associated with the siting and design of new substations and major extensions or major modifications to existing substations. They cover a range of key issues from the time options are initially considered to final design, including form, silhouette and colour of the entire development in relation to the surrounding area, and also related issues such as overhead line entries, since these are dominant features in any substation.

Environmental Report

8 In order to achieve these objectives, the environmental effects of new substations and extensions or modifications to existing substations will be assessed and where appropriate an environmental report prepared describing the effects and mitigative measures. Items to be considered are summarised in Appendix A.

Integrating Environmental Considerations into Power System Planning

9 The nature of transmission system planning is such that scheme proposals and options may go through various stages before it is finally decided to proceed with construction.

10 The purpose of each proposal for substation, sealing end compound or line entry development should be set out in a brief, and a range of system and siting options should be evaluated and documentated as part of the selection of the preferred solution. In each case the effects of the overall development on the environment should be assessed, prior to a commitment to a particular site or design.

11 When it is clear a project is likely to proceed, an assessment should be made of any additional skills required to deal effectively with the range of environmental, land use, planning and design issues. Consideration should also be given to consultation as soon as reasonably possible with appropriate statutory planning and amenity bodies.

Liaison with other Electricity Companies

12 NGC will encourage and recommend other parties such as power generators or regional electricity companies to adopt these guidelines when working with NGC on proposals for substations, sealing end compounds or line entries.

Post Construction Review

13 Following completion of the project, a review should be undertaken to check that the necessary measures identified in the environmental report have been implemented.

Section III GUIDELINES

Overall System Options and Site Selection

1 **In the development of system options including new substations, consideration must be given to environmental issues from the earliest stage to balance the technical benefits and capital cost requirements for new developments against the consequential environmental effects in order to keep adverse effects to a reasonably practicable minimum.**

Amenity, Cultural or Scientific Value of Sites

2 **The siting of new NGC substations, sealing end compounds and line entries should as far as reasonably practicable seek to avoid altogether internationally and nationally designated areas of the highest amenity, cultural or scientific value by the overall planning of the system connections.**

- ***Notes:***

  1 *Internationally and nationally designated areas of highest amenity, cultural or scientific value are:*

  National Parks;
  Areas of Outstanding Natural Beauty;
  Heritage Coasts;
  World Heritage Sites;
  Ramsar Sites;
  Sites of Special Scientific Interest;
  National Nature Reserves;
  Special Protection Areas;
  Special Areas of Conservation.

  2 *Care should be taken in relation to all historic sites with statutory protection eg Ancient Monuments, Battlefields and Listed Buildings.*

  3 *Account should be taken of Government Planning Policy Guidance and established codes of practice.*

  4 *Account should be taken of any development plan policies relevant to the siting or design of substations.*

3 **Areas of local amenity value, important existing habitats and landscape features including ancient woodland, historic hedgerows, surface and ground water sources and nature conservation areas**

should be protected as far as reasonably practicable.

Local Context, Land Use and Site Planning

4 **The siting of substations, extensions and associated proposals should take advantage of the screening provided by land form and existing features and the potential use of site layout and levels to keep intrusion into surrounding areas to a reasonably practicable minimum.**

- ***Notes:***

  1 *A preliminary study should be undertaken to identify the extent of land required to meet both operational and environmental needs.*
  2 *In some instances it may be possible to site a substation partially or fully enclosed by existing woodlands.*
  3 *Topographical information should be obtained at an early stage. In some cases a geotechnical survey may be required.*

5 **The proposals should keep the visual, noise and other environmental effects to a reasonably practicable minimum.**

- ***Notes:***

  1 *Allow sufficient space for screening of views by mounding or planting.*
  2 *Consider appropriate noise attenuation measures where necessary.*
  3 *Use security measures which minimise visual intrusion from lighting.*
  4 *Consider appropriate on-site water pollution prevention measures.*
  5 *Consider adjoining uses and the amenity of local inhabitants.*

6 **The land use effects of the proposal should be considered when planning the siting of substations or extensions.**

- ***Notes:***

  1 *Issues for consideration include potential sterilisation of nationally important land, eg Grade 1 agricultural land and sites of nationally scarce minerals.*
  2 *Effects on land drainage.*

Design

7 **In the design of new substations or line entries, early consideration should be given to the options available for terminal towers, equipment, buildings and ancillary development appropriate to individual locations, seeking to keep effects to a reasonably practicable minimum.**

- ***Notes:***

  1 *With outdoor equipment, a preference should be given normally to a low profile design with low height structures and silhouettes*

appropriate to the background.

2 *Use lightweight narrow section materials for taller structures especially for gantries over about 6 metres in height.*

3 *Commission exterior design and colours appropriate to the surroundings.*

4 *Materials and colours for buildings, equipment and fencing should be chosen to harmonise with local surroundings.*

5 *Where possible avoid the use of prominent insulators by consideration of available colours appropriate to the background.*

6 *Where possible site buildings to act as visual screens for switchgear.*

7 *Ensure that the design of high voltage and low voltage substations is co-ordinated by early consultation between NGC and its customers.*

8 *Where there are particular technical or environmental constraints, it may be appropriate to consider the use of Gas Insulated Switchgear (GIS) equipment which occupies less space and is usually enclosed within a building.*

9 *Early consideration should be given to the routeing of utility service connections.*

8 **Space should be used effectively to limit the area required for development consistent with appropriate mitigation measures and to minimise the adverse effects on existing land use and rights of way, whilst also having regard to future extension of the substation.**

- ***Notes:***

1 *Assess the benefit of removing redundant substation equipment from existing sites where this would improve their appearance.*

9 **The design of access roads, perimeter fencing, earthshaping, planting and ancillary development should form an integral part of the site layout and design to fit in with the surroundings.**

<u>Line Entries</u>

10 **In open landscape especially, high voltage line entries should be kept, as far as possible, visually separate from low voltage lines and other overhead lines so as to avoid a confusing appearance.**

11 **The inter-relationship between towers and substation structures and background and foreground features should be studied to reduce the prominence of structures from main viewpoints. Where practicable the exposure of terminal towers on prominent ridges should be minimised by siting towers against a background of trees rather than open skylines.**

END

APPENDIX A

NGC SUBSTATIONS – ENVIRONMENTAL REPORT

Introduction

All proposals for significant extensions of existing substations or for new substations and associated development should be the subject of an environmental appraisal and an environmental report should be produced. The project manager will be responsible for ensuring that an appropriate appraisal is undertaken and report prepared, with due regard to expert advice available to the team.

For a major development a scoping exercise should be undertaken with the contribution of appropriate skills to establish the range and depth of the appraisal. It will generally be appropriate at this stage to consider consultation with the local planning authority.

A clear distinction should be drawn between the preparation of an environmental report which will be undertaken in most cases and a full environmental statement (ES) which may on occasion be required under UK environmental assessment legislation, for example where the substation forms part of a major new power station for which an ES may be needed.

Recommended Content of Environmental Reports for Substations

Section 1

Information describing the project during construction, when operational and on de-commissioning including:-

1.1 Purpose and physical characteristics of the project, including details of access and transport arrangements and employment.

1.2 Land use requirements and other physical features of the project.

1.3 Operational features of the project and relevant measurements of emissions such as noise, vibration, light, heat and electric and magnetic fields.

1.4 Main alternative sites considered and reasons for final choice.

Section 2

Information describing the site and its environment including:-

2.1 Physical features such as

-Flora and fauna
-Soil: agricultural quality, geology
-Water courses including land drainage generally
-Climatic factors

-Historic heritage and archaeological sites
-Landscape and topography
-Local recreational uses
-Proximity of population and any other relevant environmental features.

2.2 The policy framework

The policy framework including all relevant statutory designations such as national nature reserves, sites of special scientific interest, national parks, areas of outstanding natural beauty, heritage coasts, special protection areas, special areas of conservation, regional parks, country parks, national forest parks, local nature reserves, areas affected by tree preservation orders, water protection zones, minerals protection zones, nitrate sensitive areas, conservation areas, listed buildings, scheduled ancient monuments, and designated areas of archaeological importance. It should also include references to Structure, Unitary and Local plan policies applying to the site and the surrounding area which are relevant to the proposed development as well as to any international designations.

Section 3

Assessment of effects on the surrounding area and landscape including:-

3.1 Visual effects, emissions during normal operation, noise, light, impact on local roads and transport.

3.2 Effects of the development on buildings, the architectural and historic heritage and archaeological features.

3.3 Loss of, and damage to flora, fauna and geology.

3.4 Land use/resource effects such as
- quality and quantity of agricultural land to be taken
- sterilisation of mineral resources and alternative uses of the site.

3.5 Changes to hydrographic characteristics.

3.6 Air and Climate

3.7 Indirect matters such as
- traffic (road, rail, air, water) related to the development,
- development associated with the project, eg new roads, sewers, power lines, pipelines, telecommunications etc.

Section 4

Mitigation measures

4.1 Where significant adverse effects are identified, a description of the measures to be taken to avoid, reduce or remedy those effects, eg

a) site planning;

b) technical measures eg equipment selection, recycling of waste or redundant parts, pollution control and treatment, containment (eg shielding of transformers and bunding)

c) aesthetic and ecological measures eg
   - mounding, design, colour, landscaping, tree planting
   - measures to preserve particular habitats or create alternative habitats
   - recording of archaeological sites
   - measures to safeguard historic buildings or sites.

END

Norfolk Vanguard Offshore Wind Farm

The Applicant Responses to First Written Questions

Appendix 3.1 - Red-throated diver displacement

Applicant: Norfolk Vanguard Limited
Document Reference: ExA;WQApp3.1;10.D1.3
Revision: Version 1

Date: January 2019
Author: MacArthur Green

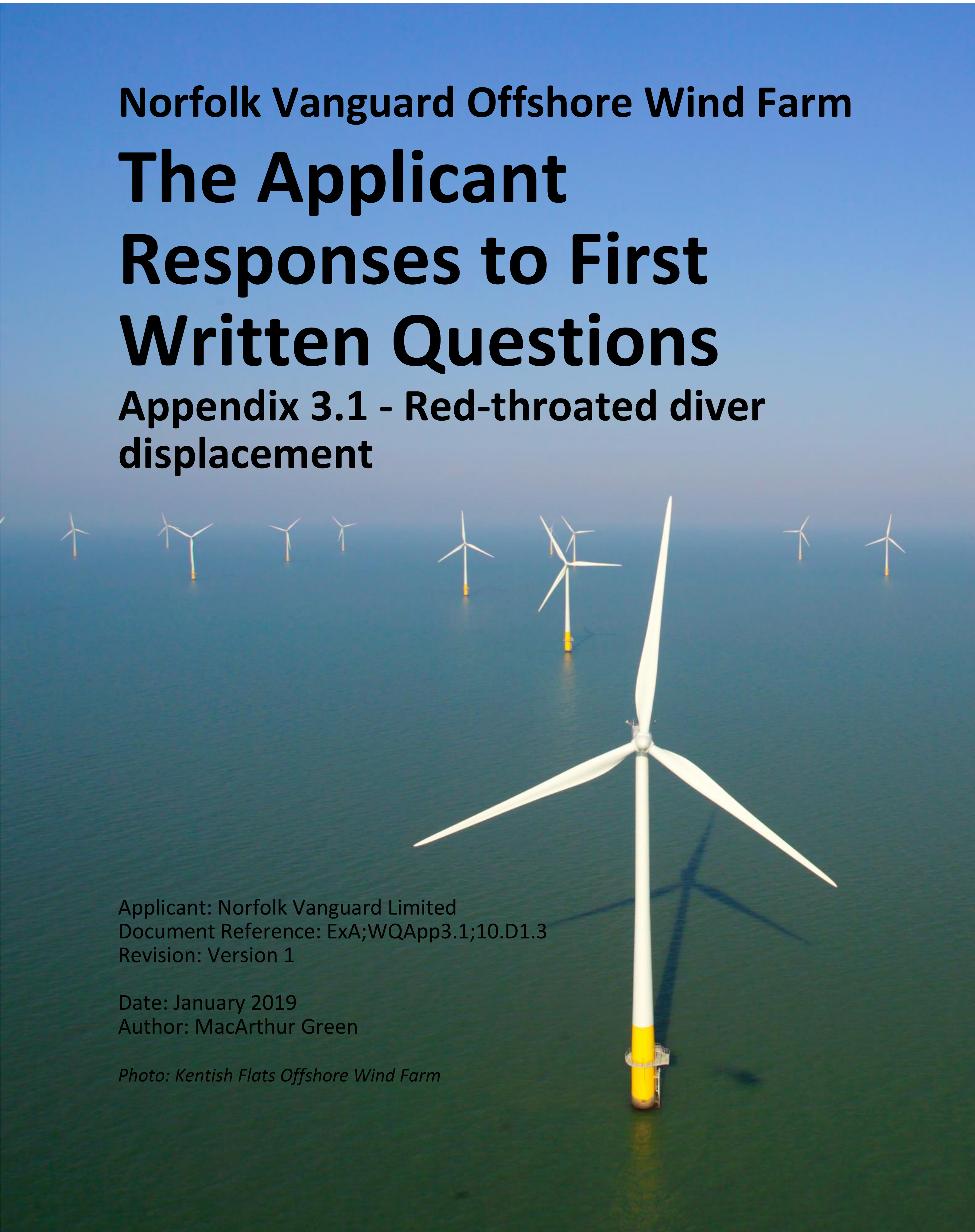

Photo: Kentish Flats Offshore Wind Farm

Date	Issue No.	Remarks / Reason for Issue	Author	Checked	Approved
23/11/2018	01D	First draft for Norfolk Vanguard Ltd review	MT	JKL	EV
28/11/2018	02D	Second draft for Norfolk Vanguard Ltd review	MT	RWF	EV
18/12/2018	03D	Final	MT	RWF	EV

EXECUTIVE SUMMARY

This report provides an updated assessment of potential displacement impacts on red-throated divers in relation to the Norfolk Vanguard Offshore Wind Farm. This has been produced taking into account comments provided by Natural England in their Relevant Representation for the project.

Natural England recommended use of a displacement rate of 100% and consequent mortality of 10% including all birds within 4 km of the wind farm boundary, as per Statutory Nature Conservation Body (SNCB) guidelines. A review of studies conducted at offshore wind farms and other relevant literature has been conducted to further inform this assessment. Following this review, Norfolk Vanguard Ltd proposes an evidence-based displacement rate of 90% and a consequent mortality rate of 1%, including birds within 2km of the wind farm boundary. The assessment presented in this report provides both the precautionary rates recommended by Natural England and the evidence-based ones.

This report provides an update of the conclusions of the original assessment following the revised displacement predictions. Most impacts remain of minor significance, as concluded in the Environmental Statement (ES), however minor to moderate significant effects are now predicted for Norfolk Vanguard West in mid-winter and annually, for both NV East and West combined and cumulative. However, it should be noted that these results only apply when the highly precautionary approach recommended by the Statutory Nature Conservation Bodies (SNCBs) is used.

Taking all of the evidence into account (as presented in the review appended to this report), displacement of red-throated divers, from Norfolk Vanguard alone and cumulatively, assessed for the complete south west North Sea red-throated diver Biologically Defined Minimum Population Scale (BDMPS), are still considered to be of minor significance, as concluded in the ES.

Table of Contents

Tables

Glossary

AIS	Automatic Identification System
BACI	Before-After-Control-Impact
BDMPS	Biologically Defined Minimum Population Scale
EIA	Environmental Impact Assessment
ES	Environmental Statement
HRA	Habitats Regulations Assessment
NE	Natural England
NV	Norfolk Vanguard
RTD	Red-throated diver
SNCB	Statutory Nature Conservation Body
SPA / pSPA	Special Protection Area / proposed Special Protection Area

1 INTRODUCTION

1. This report provides an updated assessment of potential red-throated displacement (RTD) from Norfolk Vanguard (NV) alone and cumulatively during both the construction and operational phases of the project with respect to the Environmental Impact Assessment (EIA). Assessment with regards the designated populations of the Greater Wash Special Protection Area (SPA) and Outer Thames proposed SPA (pSPA) are provided in a separate update to the Habitats Regulations Assessment (HRA). This assessment has been produced to address comments provided by Natural England in their Relevant Representation (Natural England 2018), which have been reproduced in Table 1.1.

Table 1.1 Natural England (2018) comments with respect to the assessment of potential impact of displacement on red-throated diver and sections in this report where they have been addressed.

NE Relevant Representation paragraph no.	Comment	Section where addressed
4.2.5	As advised in our Section 42 (Preliminary Environmental Information Report) response, NE require that the variability (uncertainty) in the underlying population estimates (i.e. through consideration of appropriately calculated upper and lower confidence intervals) is considered in the displacement assessments. Whilst the upper and lower confidence limits around the bird abundance estimates are presented in the tables in Annex 1 of Appendix 13.01, these have not been considered in the impact assessments for construction or operational displacement within the ES, with only the mean peak seasonal abundances considered. This approach needs to be revisited for all relevant species.	This has been provided in Table 1.8. Note that in the interests of clarity, the updated displacement matrices only present the mean population estimates, but Table 1.8 and the discussion provides additional consideration of the impacts predicted for the upper and lower confidence interval estimates.

NE Relevant Representation paragraph no.	Comment	Section where addressed
4.2.6	Natural England has a number of concerns with the assessment of displacement impacts on red-throated diver (RTD), including the following key points: a) The mean peak seasonal abundances used by the Applicant in the operational displacement assessments and matrices for Vanguard West appear to be too low, seemingly because only data for birds on the water have been used. The joint Statutory Nature Conservation Bodies (SNCB) interim displacement advice note (SNCBs 2017) advises that displacement assessments should use bird data for birds sitting on the water and birds in flight, as is the case with the Vanguard East assessment. b) In particular, Natural England does not consider the 80% displacement and 5% mortality rate used by the Applicant to be appropriate for assessing disturbance and displacement impacts to RTD from offshore wind farms. We note that this does not follow SNCB guidance (SNCBs 2017). Natural England considers that there is no clear justification to change our current advice of a 4 km buffer and 100% displacement across this buffer, and continue to advise that assessments of operational disturbance and displacement for RTD for offshore wind farm assessments are based on a constant displacement rate across the offshore wind farm site and a 4 km buffer and suggest that a range of displacement rates up to 100% and a mortality rate of up to 10% are considered. These values should also be used in the assessment of construction disturbance and displacement for RTD for both EIA and for the HRA assessment for RTD at the Greater Wash SPA. c) Impacts from the operational phase of the development through vessel movements etc., and how these impacts might be mitigated have not been given sufficient consideration. This applies to the Greater Wash SPA and potentially also the Outer Thames Estuary SPA red-throated diver population, as this also could be affected by vessels transiting the site (as the operations and maintenance port is yet to be confirmed).	a) abundance estimates for NV West have been updated to include birds on the water and in flight throughout. b) The SNCB recommended rates of 100% displacement and 10% mortality within the 4 km buffer have been discussed throughout and used for assessing construction and operational effects. We have also conducted a thorough evidence review which is included in Annex 1. This review concludes that 90% displacement and 1% consequent mortality is a more appropriately precautionary combination for this species and that this effect extends to 1.5 km from the wind farm boundary. The impact using these values is also discussed. Note that effects in relation to the SPA populations (i.e. HRA) will be considered in a separate document. c) Extra consideration for this impact has been provided in Section 1.1.2.

NE Relevant Representation paragraph no.	Comment	Section where addressed
4.2.15	The cumulative RTD displacement mortality has been conducted by the Applicant using the same magnitudes of displacement (80%) and mortality (5%) applied to all birds within the 4 km wind farm buffer. As with the assessment of operational displacement for Vanguard alone, Natural England does not consider this to be precautionary and advises that a worst case scenario of 100% displacement and 10% mortality is used.	The SNCB recommended rates of 100% displacement and 10% mortality including the 4 km buffer have been discussed throughout. We have also conducted a thorough evidence review which is included as an annex. This review concludes that 90% displacement and 1% consequent mortality is a more appropriately precautionary combination for this species, extending within a 1.5 km buffer (although a 2km buffer has been used in this assessment). The impact using these values is also discussed.
4.2.16	The Applicant has considered that all wind farms at which turbines were installed before or during 2012 form part of the Norfolk Vanguard baseline. Natural England does not agree that these wind farms should be considered part of the baseline. This is because, although some of the wind farms included in the Applicant's list have been operational for over 10 years, the RTD population data used in Furness (2015) pre-date the installations. We suggest that a similar approach to that undertaken for the auk cumulative displacement assessments is undertaken for RTD, i.e. to sum the bird abundance estimates for each relevant offshore wind farm and put this total through a displacement matrix, and then assess with a worst case scenario of 100% displacement and 10% mortality. The assessment should include all offshore wind farms located within the south-west North Sea RTD BDMPS.	The red-throated diver assessment has been recalculated including all wind farms in the south west North Sea and using rates recommended by the SNCBs and also those identified by the evidence review (Annex 1).

1.1 Assessment of potential impacts

1.1.1 Potential impacts during operation

1.1.1.1 Project alone: Disturbance and displacement from offshore infrastructure

2. The Norfolk Vanguard ES (Vattenfall 2018) assessed red-throated diver displacement effects using a wind farm displacement rate of 80% and a consequent increase in mortality of 5%, applied to all birds within 4 km of the wind farm boundary.

3. Natural England (2018) states that they do not agree these are appropriate rates and note that these rates are not in line with those recommended which are 100% displacement (up to 4 km from the wind farm) and mortality of 10% (SNCBs 2017). Natural England also identified an error in the populations assessed for NV West, which omitted birds in flight.

4. Following receipt of Natural England (2018), a comprehensive review of the most up to date evidence on red-throated diver disturbance was conducted (Annex 1). This review reaches the following evidence-based conclusions on appropriate rates for displacement and consequent mortality:

   - Displacement from offshore wind farms is expected to be 90%;
   - The extent of displacement beyond the wind farm boundary is variable, but a figure of 90% within 1.5 km is suitable precautionary; and
   - The consequence of displacement is currently unknown (in terms of potential increases in mortality), however considering the ecology of red-throated divers and evidence for displacement effects across a wide range of species, it is likely that any increase in mortality will be close to 0% and is highly unlikely to exceed a precautionary increase of 1%.

5. Combining all of the above, the following sections provide the following revised assessment for red-throated divers:

   - Operational displacement matrices for NV East and NV West which highlight both NE’s recommended rates (100% & 10% including the 4 km buffer) and those identified in the evidence review (90% and 1% including the 4km buffer) and discussion are provided for effects within the 2 km buffer, in line with the conclusions of the evidence review. Note that the review provides evidence for effects only within 1.5 km, but effects have been assessed using the existing 2 km densities, thereby including precaution on this aspect.
   - For both NV East (section 1.1.1.1.1) and NV West (section 1.1.1.1.2) the populations at risk include birds recorded on the water and in flight (correcting the omission of birds in flight for NV West).
   - An additional table (Table 1.8) presenting displacement mortality using the upper and lower confidence intervals of the seasonal abundance has been included.

6. The displacement matrices have been populated with data for red-throated diver during the autumn migration, mid-winter and spring migration periods within the site and 4 km buffer. These tables present displacement from 0 – 100% at 10%

increments and mortality from 0 – 100%, at 1% increments up to 10%, and larger gaps thereafter. Shading has been used to highlight the 90-100% displacement and 1-10% mortality ranges defined above. Additional discussion of the potential impact for the evidence-based effect operating within 2 km has also been provided.

1.1.1.1.1 Norfolk Vanguard East

Autumn migration

7. Using the seasonal mean peak autumn migration abundance on NV East and 4 km buffer of 50, the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated as:

- Five individuals at the SNCB guidance rates of 100% displacement and 10% mortality (Table 1.2);
- No individuals at the evidence-based rates of 90% displacement and 1% mortality (Table 1.2).

8. The BDMPS for red-throated diver in autumn is 13,277 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of five individuals to this would increase the mortality rate by 0.16%, while the evidence-based prediction (0) results in no increase in mortality.

9. It should also be noted that the mean abundance within NV East and the 2 km buffer was 45 individuals (compared with 50 within the 4 km buffer) which would slightly reduce the magnitude of impact (i.e. by 10%).

10. Even the maximum magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the autumn migration period, the magnitude of effect is assessed as negligible even on the basis of the SNCB's highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance remains **minor adverse** as presented in the ES.

Table 1.2 Displacement matrix presenting the number of red-throated divers in Norfolk Vanguard East (and 4 km buffer) during the autumn migration season that may be subject to mortality. Highlighted cells bracket the range from the evidence-based combination (90%-1%) to the SNCB guidance combination (100%-10%).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	0	0	0	0	0	0	0	0	1
2	0	0	0	0	1	1	1	1	1	1
3	0	0	0	1	1	1	1	1	1	2
4	0	0	1	1	1	1	1	2	2	2
5	0	1	1	1	1	2	2	2	2	3
6	0	1	1	1	2	2	2	2	3	3
7	0	1	1	1	2	2	2	3	3	4
8	0	1	1	2	2	2	3	3	4	4
9	0	1	1	2	2	3	3	4	4	5
10	1	1	2	2	3	3	4	4	5	5
20	1	2	3	4	5	6	7	8	9	10
30	2	3	5	6	8	9	11	12	14	15
50	3	5	8	10	13	15	18	20	23	25
75	4	8	11	15	19	23	26	30	34	38
100	5	10	15	20	25	30	35	40	45	50

Mid-winter

11. Using the seasonal mean peak winter abundance on NV East and 4 km buffer of 25, the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated as:

    - Three individuals at the SNCB guidance rates of 100% displacement and 10% mortality (Table 1.3);
    - No individuals at the evidence-based rates of 90% displacement and 1% mortality (Table 1.3).

12. The BDMPS for red-throated diver in winter is 10,177 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number of individuals expected to die is 2,320 (10,177 x 0.228). The addition of three individuals to this would increase the mortality rate by 0.13%, while the evidence-based prediction (0) results in no increase in mortality.

13. It should be noted that the mean abundance within NV East and the 2 km buffer was 13 individuals (compared with 25 within the 4 km buffer) which would halve the magnitude of predicted impact to 2 individuals using the SNCB recommended rates.

14. Even the maximum magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the winter period, the magnitude of effect is assessed as negligible even on the basis of the SNCB's highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance remains **minor adverse** as presented in the ES.

Table 1.3 Displacement matrix presenting the number of red-throated divers in Norfolk Vanguard East (and 4km buffer) during the winter that may be subject to mortality. Highlighted cells bracket the range from the evidence-based combination (90%-1%) to the SNCB guidance combination (100%-10%).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	0	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0	0	1
3	0	0	0	0	0	0	1	1	1	1
4	0	0	0	0	1	1	1	1	1	1
5	0	0	0	1	1	1	1	1	1	1
6	0	0	0	1	1	1	1	1	1	2
7	0	0	1	1	1	1	1	1	2	2
8	0	0	1	1	1	1	1	2	2	2
9	0	0	1	1	1	1	2	2	2	2
10	0	1	1	1	1	2	2	2	2	3
20	1	1	2	2	3	3	4	4	5	5
30	1	2	2	3	4	5	5	6	7	8
50	1	3	4	5	6	8	9	10	11	13
75	2	4	6	8	9	11	13	15	17	19
100	3	5	8	10	13	15	18	20	23	25

Spring migration

15. Using the seasonal mean peak autumn migration abundance on NV East and 4 km buffer of 119, the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated as:

- 12 individuals at the SNCB guidance rates of 100% displacement and 10% mortality (Table 1.4);
- One individual at the evidence-based rates of 90% displacement and 1% mortality (Table 1.4).

16. The BDMPS for red-throated diver in spring is 13,277 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of 12

individuals to this would increase the mortality rate by 0.4%, while the evidence-based prediction (1) results in an increase in mortality of 0.03%.

17. It should be noted that the mean abundance within NV East and the 2 km buffer was 90 individuals (compared with 119 within the 4 km buffer) which would reduce the magnitude of predicted impact by 25%, to 8 individuals using the SNCB recommended rates.

18. Even the maximum magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the spring migration period, the magnitude of effect is assessed as negligible even on the basis of the SNCB's highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance remains **minor adverse** as presented in the ES.

Table 1.4 Displacement matrix presenting the number of red-throated divers in Norfolk Vanguard East (and 4 km buffer) during the spring migration period that may be subject to mortality. Highlighted cells bracket the range from the evidence-based combination (90%-1%) to the SNCB guidance combination (100%-10%).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	0	0	0	1	1	1	1	1	1
2	0	0	1	1	1	1	2	2	2	2
3	0	1	1	1	2	2	2	3	3	4
4	0	1	1	2	2	3	3	4	4	5
5	1	1	2	2	3	4	4	5	5	6
6	1	1	2	3	4	4	5	6	6	7
7	1	2	2	3	4	5	6	7	7	8
8	1	2	3	4	5	6	7	8	9	10
9	1	2	3	4	5	6	7	9	10	11
10	1	2	4	5	6	7	8	10	11	12
20	2	5	7	10	12	14	17	19	21	24
30	4	7	11	14	18	21	25	29	32	36
50	6	12	18	24	30	36	42	48	54	60
75	9	18	27	36	45	54	62	71	80	89
100	12	24	36	48	60	71	83	95	107	119

Complete nonbreeding season

19. The summed NV East displacement mortality for autumn, mid-winter and spring using the SNCB recommended rates is 20 individuals (at 100% displaced and 10% mortality within the 4 km buffer) while the evidence-based mortality is 2 individuals (at 90% displacement and 1% mortality within the 4 km buffer, allowing for

rounding), although these figures include an unknown degree of double counting due to overlaps in the populations in each period.

20. It should be noted that the summed mortality for NV East within the 2 km buffer was 15 individuals (compared with 20 within the 4 km buffer) which further reduces the magnitude of predicted impact.

21. This additional mortality would increase the background mortality by a maximum of 0.66% which would be undetectable, although it is more likely there would be no effect at all. Therefore, during the entire nonbreeding period, the magnitude of effect is assessed as negligible even on the basis of the highly precautionary assessment approach recommended by the SNCBs, and the additional precaution due to the potential for double counting. As the species is of high sensitivity to disturbance, the impact significance remains **minor adverse** as presented in the ES.

1.1.1.1.2 Norfolk Vanguard West

Autumn migration

22. Using the seasonal mean peak autumn migration abundance on NV West and 4 km buffer of 25, the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated as:

- Three individuals at the SNCB guidance rates of 100% displacement and 10% mortality (Table 1.5);
- No individuals at the evidence-based rates of 90% displacement and 1% mortality (Table 1.5).

23. The BDMPS for red-throated diver in autumn is 13,277 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of three individuals to this would increase the mortality rate by 0.01%, while the evidence-based prediction (0) results in no increase in mortality.

24. It should also be noted that the mean abundance within NV West and the 2 km buffer was 18 individuals (compared with 25 within the 4 km buffer) which would reduce the magnitude of impact by almost 30% to 2 individuals.

25. Even the maximum magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the autumn migration period, the magnitude of effect is assessed as negligible even on the basis of the SNCB’s highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance remains **minor adverse** as presented in the ES.

Table 1.5 Displacement matrix presenting the number of red-throated divers in Norfolk Vanguard West (and 4 km buffer) during the autumn migration season that may be subject to mortality. Highlighted cells bracket the range from the evidence-based combination (90%-1%) to the SNCB guidance combination (100%-10%).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	0	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0	0	1
3	0	0	0	0	0	0	1	1	1	1
4	0	0	0	0	1	1	1	1	1	1
5	0	0	0	1	1	1	1	1	1	1
6	0	0	0	1	1	1	1	1	1	2
7	0	0	1	1	1	1	1	1	2	2
8	0	0	1	1	1	1	1	2	2	2
9	0	0	1	1	1	1	2	2	2	2
10	0	1	1	1	1	2	2	2	2	3
20	1	1	2	2	3	3	4	4	5	5
30	1	2	2	3	4	5	5	6	7	8
50	1	3	4	5	6	8	9	10	11	13
75	2	4	6	8	9	11	13	15	17	19
100	3	5	8	10	13	15	18	20	23	25

Mid-winter

26. Using the seasonal mean peak winter abundance on NV West and 4 km buffer of 356, the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated as:

    - 36 individuals at the SNCB guidance rates of 100% displacement and 10% mortality (Table 1.6);
    - Three individuals at the evidence-based rates of 90% displacement and 1% mortality (Table 1.6).

27. The BDMPS for red-throated diver in winter is 10,177 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number of individuals expected to die is 2,320 (10,177 x 0.228). The addition of 36 individuals to this would increase the mortality rate by 1.5%, while the evidence-based prediction of 3 would increase the mortality rate by 0.13%.

28. It should be noted that the mean abundance within NV West and the 2 km buffer was 235 individuals (compared with 356 within the 4 km buffer) which would reduce the magnitude of predicted impact by 33% to 23 individuals using the SNCB recommended rates.

29. Using the highly precautionary SNCB approach this indicates the potential for a low magnitude of effect, while the evidence-based approach indicates a negligible magnitude of effect.

30. Therefore, during the winter period, since the species is of high sensitivity to disturbance, on the basis of the evidence based displacement and mortality rates the impact significance remains **minor adverse** as presented in the ES and **minor to moderate adverse** using the precautionary SNCB methods.

Table 1.6 Displacement matrix presenting the number of red-throated divers in Norfolk Vanguard West (and 4 km buffer) during the winter period that may be subject to mortality. Highlighted cells bracket the range from the evidence-based combination (90%-1%) to the SNCB guidance combination (100%-10%).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	1	1	1	2	2	2	3	3	4
2	1	1	2	3	4	4	5	6	6	7
3	1	2	3	4	5	6	7	9	10	11
4	1	3	4	6	7	9	10	11	13	14
5	2	4	5	7	9	11	12	14	16	18
6	2	4	6	9	11	13	15	17	19	21
7	2	5	7	10	12	15	17	20	22	25
8	3	6	9	11	14	17	20	23	26	28
9	3	6	10	13	16	19	22	26	29	32
10	4	7	11	14	18	21	25	28	32	36
20	7	14	21	28	36	43	50	57	64	71
30	11	21	32	43	53	64	75	85	96	107
50	18	36	53	71	89	107	125	142	160	178
75	27	53	80	107	134	160	187	214	240	267
100	36	71	107	142	178	214	249	285	320	356

Spring migration

31. Using the seasonal mean peak autumn migration abundance on NV West and 4 km buffer of 197, the predicted number of individual red-throated divers which could potentially suffer mortality as a consequence of displacement has been estimated as:

- 20 individuals at the SNCB guidance rates of 100% displacement and 10% mortality (Table 1.7);
- Two individuals at the evidence-based rates of 90% displacement and 1% mortality (Table 1.7).

32. The BDMPS for red-throated diver in spring is 13,277 (Furness 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number

of individuals expected to die is 3,027 (13,277 x 0.228). The addition of 20 individuals to this would increase the mortality rate by 0.66%, while the evidence-based prediction (2) results in an increase in mortality of 0.07%.

33. It should be noted that the mean abundance within NV West and the 2 km buffer was 127 individuals (compared with 197 within the 4 km buffer) which would reduce the magnitude of predicted impact by 35%, to 13 individuals using the SNCB recommended rates.

34. Even the maximum magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the spring migration period, the magnitude of effect is assessed as negligible even on the basis of the SNCB's highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance remains **minor adverse** as presented in the ES.

Table 1.7 Displacement matrix presenting the number of red-throated divers in Norfolk Vanguard West (and 4 km buffer) during the spring migration period that may be subject to mortality. Highlighted cells bracket the range from the evidence-based combination (90%-1%) to the SNCB guidance combination (100%-10%).

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	0	0	1	1	1	1	1	2	2	2
2	0	1	1	2	2	2	3	3	4	4
3	1	1	2	2	3	4	4	5	5	6
4	1	2	2	3	4	5	6	6	7	8
5	1	2	3	4	5	6	7	8	9	10
6	1	2	4	5	6	7	8	9	11	12
7	1	3	4	6	7	8	10	11	12	14
8	2	3	5	6	8	9	11	13	14	16
9	2	4	5	7	9	11	12	14	16	18
10	2	4	6	8	10	12	14	16	18	20
20	4	8	12	16	20	24	28	32	35	39
30	6	12	18	24	30	35	41	47	53	59
50	10	20	30	39	49	59	69	79	89	99
75	15	30	44	59	74	89	103	118	133	148
100	20	39	59	79	99	118	138	158	177	197

Complete nonbreeding season

35. The summed NV West displacement mortality for autumn, mid-winter and spring using the SNCB recommended rates is 59 individuals (at 100% displaced and 10% mortality within the 4 km buffer) while the evidence-based mortality is 5 individuals (at 90% displacement and 1% mortality within the 4 km buffer), although these

figures include an unknown degree of double counting due to overlaps in the populations in each period.

36. It should be noted that the summed mortality for NV West within the 2 km buffer was 38 individuals (compared with 59 within the 4 km buffer) which further reduces the magnitude of predicted impact.

37. The maximum additional mortality would increase the background mortality rate by a maximum of 1.95%. Using the highly precautionary SNCB approach this indicates the potential for a low magnitude of effect, while the evidence-based approach indicates a negligible magnitude of effect.

38. Therefore, during the entire nonbreeding period, the magnitude of effect is assessed as low to negligible and as the species is of high sensitivity to disturbance, on the basis of the evidence based displacement and mortality rates the impact significance remains **minor adverse** as presented in the ES and **minor to moderate adverse** using the precautionary SNCB methods.

1.1.1.1.3 Norfolk Vanguard East and Norfolk Vanguard West

39. The worst case displacement impact has been assessed on the basis that both NV East and NV West would be completely developed, although this is highly precautionary since even if each site contains half the total number of turbines it is very unlikely they would be distributed across the entirety of both sites (i.e. the actual developed area would be less than the sum of the total area of both assumed by this approach).

40. The combined NV east and NV west assessment (Table 1.8) also presents the range of predicted outcomes with inclusion of uncertainty in the estimated population sizes as requested by Natural England. This uses the upper and lower 95% confidence intervals of abundance for the wind farm areas plus 4 km buffers (as presented in Vattenfall 2018, Offshore Ornithology Technical Appendix 13.1).

Table 1.8 Red-throated diver displacement mortality across all seasons for NV East, NV West (inc. 4 km buffers) and both combined, with full consideration for uncertainty in abundance estimates and assessed using the SNCB recommended rates and the evidence-based rates.

Period	Site	Displacement rate (%)	Mortality rate (%)	Displacement mortality including uncertainty in population size		
				Lower confidence interval	Mean abundance	Upper confidence interval
NV East	Autumn	100	10	0	5	11
		90	1	0	0	1
	Mid-winter	100	10	0	3	8
		90	1	0	0	1

Period	Site	Displacement rate (%)	Mortality rate (%)	Displacement mortality including uncertainty in population size		
				Lower confidence interval	Mean abundance	Upper confidence interval
	Spring	100	10	0	12	38
		90	1	0	1	3
	Total	100	10	0	20	59
		90	1	0	1	5
NV west	Autumn	100	10	0	3	8
		90	1	0	0	1
	Mid-winter	100	10	7	36	71
		90	1	1	3	6
	Spring	100	10	11	20	29
		90	1	1	2	3
	Total	100	10	18	59	108
		90	1	2	5	10
NV East & NV West	Total	100	10	18	79	167
		90	1	2	6	15

41. The combined displacement mortality across both NV East and NV West (inc. 4 km buffers) for the complete nonbreeding period using the mean abundance estimates would be 79 individuals (at 100% displaced and 10% mortality) while the evidence-based mortality is 6 individuals (at 90% displacement and 1% mortality). Both these sets of figures include a very large degree of precaution: in the approach to estimating numbers affected, the assumption that both sites will be fully developed and double counting across overlapping seasons. Furthermore, the evidence review has found that displacement is very likely to extend no further than 1.5 km beyond the wind farm. As discussed above this would reduce the number affected by around 33% (i.e. the upper mean estimate would be around 52, compared with 79). Given these existing sources of precaution, it is considered highly unrealistic to add further precaution to this assessment through the incorporation of predictions derived using the upper confidence intervals of abundance. Furthermore, any consideration of the upper estimates should be balanced against the lower confidence estimates which suggest a maximum mortality of 18 using the highly precautionary SNCB methods.

42. The various assessment approaches generate the following predicted increases in mortality for NV East and NV West combined:

- 2.6% (4 km buffer, 100% displacement, 10% mortality);
- 1.7% (2 km buffer, 100% displacement, 10% mortality);
- 0.2% (4 km buffer, 90% displacement, 1% mortality);
- 0.1% (2 km buffer, 90% displacement, 1% mortality).

43. Using the highly precautionary SNCB approach (detailed above), combined displacement across both NV East and NV West (4 km buffer, 100% displaced, 10% consequent mortality) would increase the background mortality by 2.6%, while the evidence-based approach indicates a maximum increase in background mortality of 0.2% (4 km buffer) and a more realistic increase of 0.1% (within the 2 km buffer).

44. Therefore, during the entire nonbreeding period across both NV East and NV West these therefore predict a low to negligible magnitude of effect and as the species is of high sensitivity to disturbance, on the basis of the evidence based displacement and mortality rates the impact significance remains **minor adverse** as presented in the ES and **minor to moderate adverse** using the precautionary SNCB methods.

1.1.1.2 Cumulative: Disturbance and displacement from offshore infrastructure

45. Cumulative red-throated diver displacement mortality has been estimated for wind farms in the south-west North Sea BDMPS (Furness 2015) which have the potential to contribute to a cumulative effect. This has been conducted using the precautionary rates of displacement and mortality recommended by the SNCBs (100% displacement and 10% mortality within the 4 km buffer) as well as those derived from the review of evidence for this species, reported in Annex 1 (90% displacement and 1% mortality).

46. The original cumulative assessment (Vattenfall 2018) focused on the core area for this species, as highlighted by the species' inclusion in the Greater Wash SPA and Outer Thames SPA. This included wind farms located between Triton Knoll and Thanet on the basis that the coastal waters in this region cover the primary over-wintering areas. In their review of the assessment, Natural England (2018) requested inclusion of all wind farms in the south-west North Sea BDMPS for red-throated diver (Furness 2015). Natural England (2018) also stated that none of the wind farms included in the cumulative assessment should be considered as part of the baseline (i.e. as if their effects were already accounted for in the Norfolk Vanguard survey data) because the BDMPS population estimates pre-dated all the wind farm construction dates. Hence, no distinction between wind farms has been made on the basis of date of commission.

47. A review of the impact assessments for the additional wind farms in the relevant BDMPS identified three categories with respect to red-throated divers: wind farms with no population estimates presented (Dogger Bank sites and Blyth demonstrator), coastal wind farms with low numbers of over-wintering birds reported (Teesside, Humber Gateway and Westermost Rough) and wind farms with sightings made during months considered to belong to the breeding season (Hornsea projects). A summary of the data for all these sites is presented along with those assessed in the ES (Table 1.9).

Table 1.9 Summary of red-throated diver assessments for older wind farms in south west North Sea BDMPS with potential to contribute to a cumulative operational displacement impact. Displacement estimates presented as the evidence-based combination (90%-1%) and SNCB guidance combination (100%-10%).

Wind farm	Red-throated diver displacement assessed?	Estimated no. of red-throated diver mortalities due to displacement	Conclusion for NV cumulative assessment
Scroby Sands	Not assessed	No number presented	NA
Kentish Flats	Yes: qualitative	No number presented	NA
Lynn & Inner Dowsing	Yes: qualitative	No number presented	NA
Gunfleet Sands	Yes: qualitative	‘very small'	NA
Thanet	Yes: quantitative	<1 - 2	Included
Sheringham Shoal	Not assessed	No number presented	NA
Greater Gabbard	Yes: quantitative	4 - 40	Included
London Array	Yes: qualitative	No number presented	NA
Lincs	Yes: qualitative	No number presented	NA
Kentish Flats Extension	Yes: qualitative	No number presented	NA
Galloper	Yes: quantitative	1 - 14	Included
Dudgeon	Not assessed	No number presented	NA
Race Bank	Not assessed	No number presented	NA
Triton Knoll	Not assessed	No number presented	NA
Dogger Bank Creyke Beck A & B	Not assessed	No number presented	NA
Dogger Bank Teesside A / Sofia	Not assessed	No number presented	NA
Blyth Demonstrator	Not assessed	No number presented	NA
Teesside	Not assessed	No number presented	NA
Westermost Rough	Not assessed	No number presented	NA
Humber Gateway	Not assessed	No number presented	NA
Hornsea 1	Not assessed	No number presented	NA
Hornsea 2	Not assessed	No number presented	NA
Hornsea 3	Not assessed	No number presented	NA

48. Although several of the additional wind farms recorded red-throated diver in low numbers in their surveys, none undertook assessment of displacement impacts. Thus, the inclusion of the additional wind farms has not altered the number of individuals assessed as at risk of displacement (although the tables present revised rates of displacement and consequent mortality). This is not wholly surprising since most of these additional wind farms are located in areas typically considered to be less preferred by species (i.e. farther offshore and in depths of >20 m). In total, for the wind farms outside the former East Anglia zone the displacement assessments indicated that between 6 and 56 individuals would be at risk of mortality. This total has been included in the cumulative assessment, together with the former East Anglia zone wind farms. Note that the latter also includes revised estimates for Thanet Extension, using the estimates presented in the ES.

Table 1.10 Red-throated diver cumulative displacement mortality calculated on the basis of the SNCB recommended rates of 100% displacement and 10% mortality and the evidence-based rates of 90% displacement and 1% mortality (all within the 4 km buffer).

Wind farm	Autumn	Midwinter	Spring	Annual
Older projects (see Table 13.65)	N/A	N/A	N/A	6 - 56
Thanet Extension	0	2 - 19	0 - 4	2 - 23
East Anglia ONE	0.4 - 5	1 - 10	1.4 - 15	3 -30
East Anglia THREE	0.4 - 5	0.2 – 2	2 - 20	0.6 - 27
Norfolk Vanguard East	0.4 – 5	0.2 – 3	1 – 12	1.6 - 20
Norfolk Vanguard West	0 – 3	3 - 36	2 - 20	5 - 59
Total (rounded)	1 - 18	6 - 70	6 - 71	18 - 215

49. The estimated cumulative red-throated diver mortality for all wind farms in the south west North Sea BDMPS region is between 18 and 215, on the basis that all individuals within 4 km of the wind farms are displaced. If it is assumed that the abundance at the other wind farms within 2 km is 33% lower than within 4 km (as is the case at Norfolk Vanguard), then application of the evidence-based finding that displacement extends no further than 1.5 km (Annex 1), these totals would decline to between 12 and 142.

50. The largest BDMPS for red-throated diver is 13,277 (Furness 2015). At the average baseline mortality rate for red-throated diver of 0.228 (Vattenfall 2018) the number of individuals expected to die is 3,027 (13,277 x 0.228). The addition of between 18 and 215 to this would increase the mortality rate by 0.6% to 7% (or 0.4% to 4.7% within 2 km).

51. The biogeographic population for red-throated diver is 27,000 (Furness, 2015). At the average baseline mortality rate for red-throated diver of 0.228 the number of individuals expected to die is 6,156 (27,000 x 0.228). The addition of between 18

and 215 would increase the mortality rate by to 0.3% to 3.5% (or 0.2% to 2.3% within 2 km).

52. To summarise, the various assessment approaches generate the following predicted increases in cumulative mortality for the BDMPS population:

- 7% (4 km buffer, 100% displacement, 10% mortality);
- 4.7% (2 km buffer, 100% displacement, 10% mortality);
- 0.6% (4 km buffer, 90% displacement, 1% mortality);
- 0.4% (2 km buffer, 90% displacement, 1% mortality);

53. The following predicted increases in cumulative mortality for the biogeographic population:

- 3.5% (4 km buffer, 100% displacement, 10% mortality);
- 2.3% (2 km buffer, 100% displacement, 10% mortality);
- 0.3% (4 km buffer, 90% displacement, 1% mortality);
- 0.2% (2 km buffer, 90% displacement, 1% mortality).

54. As discussed in preceding sections, the mortality total combines multiple sources of precaution:

- The evidence review found that 90% displacement and 1% mortality are more appropriate than the 100% and 10% recommended by the SNCBs;
- Each wind farm assessment has assumed that all birds within 4 km of the wind farm lease boundary are potentially affected, whereas the evidence suggests displacement declines with distance from wind farm boundaries and in some cases has been reported as zero by 2 km;
- It includes an unknown degree of double counting across seasons since some individuals will be present within more than one season and could also potentially move between these sites;
- The Norfolk Vanguard East 4 km buffer includes part of the East Anglia THREE wind farm and 4 km buffer and vice versa so including the buffer for both sites leads to double counting birds in the overlapping area; and
- One third of the total is predicted to occur during the spring migration period when the potential consequences of displacement are expected to be much lower since most individuals are on migration and passing through at this time.

55. Furthermore, the method used for assessing displacement impacts has no means to explicitly incorporate wind farm design modifications, specifically with respect to turbine spacing. Most wind farms are constructed with fewer, larger diameter turbines than specified in their consents. Due to the need to minimise the turbulence downwind from a turbine, as turbine rotor diameter increases, so the spacing between turbines increases (since the wake effect is a function of rotor

diameter). Since the underlying assumption for displacement from operational wind farms is that birds avoid the turbines themselves, it follows logically that as turbine spacing increases so the stimulus for avoidance behaviour decreases, thereby permitting more individuals to enter a wind farm. This is relevant because the displacement assessments for other wind farms represent the predictions for the consented designs not the final ones utilising fewer larger turbines. Thus, in addition to the sources of precaution listed above, there also needs to be allowance for the reduced displacement from built wind farms compared with the consented versions.

56. For example, East Anglia ONE was originally assessed on the basis of 333 turbines, reduced to 240 for consent with the final design further reduced to 102 turbines with only a small decrease in project area. Thus, the final wind farm will have less than one third the original number of proposed (and assessed) turbines. This will almost certainly reduce the magnitude of displacement. The total also includes an unrealistic worst case scenario for Norfolk Vanguard with complete displacement from both NV East and NV West, calculated as the summed total effect for both sites. This corresponds to two times the actual maximum number of turbines which could be installed across both sites, since the full assessments for NV East and NV West assume all turbines are located in just one of the sites, with no development in the other site. In reality, it is more reasonable to assume that combined displacement would lie between the values obtained for NV East and NV West (e.g. at the SNCB recommended 100% displacement and 10% mortality (within the 4 km buffer), rather than the sum total of 79 (or 6, at 90%-1%), the mortality would be between 20 and 59).

57. To inform the assessment, the potential effect on the background mortality resulting from combinations of displacement and mortality have been calculated for the BDMPS population and for the biogeographic population (Table 1.11 and Table 1.12). The colour shading in these tables indicates the displacement and mortality combinations which result in background mortality increases of less than 1%, between 1% and 2% and between 2% and 3%. For the BDMPs population, at all displacement rates combined with 1% consequent mortality result in background mortality rising by less than 1% (i.e. below the level at which this effect could be detected).

Table 1.11 Red-throated diver cumulative displacement matrix. Levels of mortality which would increase the baseline mortality of the smaller BDMPS population by percentage thresholds indicated by shading: green <1%; orange >1% and <2%; pink >2% and <3%; clear >3%.

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	2	4	6	9	11	13	15	17	19	22
2	4	9	13	17	22	26	30	34	39	43
3	6	13	19	26	32	39	45	52	58	65
4	9	17	26	34	43	52	60	69	77	86
5	11	22	32	43	54	65	75	86	97	108
6	13	26	39	52	65	77	90	103	116	129
7	15	30	45	60	75	90	105	120	135	151
8	17	34	52	69	86	103	120	138	155	172
9	19	39	58	77	97	116	135	155	174	194
10	22	43	65	86	108	129	151	172	194	215

58. Against the larger biogeographic population, 100% displacement would have an undetectable effect (i.e. increase the background rate by less than 1%) when combined with up to 2% mortality, while at displacement rate of 80% and 90% the same applies up to 3% mortality.

Table 1.12 Red-throated diver cumulative displacement matrix. Levels of mortality which would increase the baseline mortality of the biogeographic population by percentage thresholds indicated by shading: green <1%; orange >1% and <2%; pink >2% and <3%; clear >3%.

Mortality (%)	Displacement (%)									
	10	20	30	40	50	60	70	80	90	100
1	2	4	6	9	11	13	15	17	19	22
2	4	9	13	17	22	26	30	34	39	43
3	6	13	19	26	32	39	45	52	58	65
4	9	17	26	34	43	52	60	69	77	86
5	11	22	32	43	54	65	75	86	97	108
6	13	26	39	52	65	77	90	103	116	129
7	15	30	45	60	75	90	105	120	135	151
8	17	34	52	69	86	103	120	138	155	172
9	19	39	58	77	97	116	135	155	174	194
10	22	43	65	86	108	129	151	172	194	215

59. On the basis of the worst case SNCB approach the cumulative red-throated diver operational displacement impact magnitude is assessed as low. Therefore, as the species is of high sensitivity to disturbance, the cumulative impact significance would be **moderate adverse**.

60. On the basis of the evidence review (Annex 1) it is considered that the most realistic (and still precautionary) combination of displacement and consequent mortality rates is 90% and 1%, respectively operating within no more than 2 km of the wind farm boundary. On this basis the cumulative red-throated diver operational displacement impact magnitude is assessed as negligible. Therefore, as the species is of high sensitivity to disturbance, the cumulative impact significance would be **minor adverse**.

1.1.2 Potential impacts during construction

Offshore Export Cable Installation

61. The magnitude of disturbance to red-throated diver from construction vessels has been estimated on a worst case basis. This assumes that there would be 100% displacement of birds within a 2 km buffer surrounding the source, in this case around a maximum of two cable laying vessels. This 100% displacement from vessels is consistent with Garthe and Hüppop (2004) and Schwemmer *et al.,* (2011) since they suggested that all red-throated divers present fly away from approaching vessels at a distance of often more than 1 km.

62. In order to calculate the number of red-throated divers that would potentially be at risk of displacement from the offshore cable corridor during the cable laying process, the density of red-throated divers in the SPA along the section crossed by the offshore cable corridor was estimated. This was derived from a review of the Greater Wash SPA proposal details (Natural England and JNCC, 2016). This indicated that the peak density of birds in the SPA crossed by the cable route was between 1.36 and 3.38 per km^2.

63. The worst case area from which birds could be displaced was defined as a circle with a 2 km radius around each cable laying vessel, which is 25.2 km^2 (2 x 12.6 km^2). If 100% displacement is assumed to occur within this area, then a peak of between 34 and 85 divers could be displaced at any given time. This would lead to a 1 to 1.5% increase in diver density in the remaining areas of the SPA assuming that displaced birds all remain within the SPA. As the vessels move it is assumed that displaced birds return and therefore any individual will be subjected to a brief period of impact. It is considered reasonable to assume that birds will return following passage of the vessel since the cable laying vessels will move at a maximum speed of 400 m per hour if surface laying, 300 m per hour for ploughing and 80 m per hour if trenching (Chapter 5 Project Description). This represents a maximum speed of 7 m per minute. For context, a modest tidal flow rate for the region would be in the region of 1 m per second (60 m per minute). The tide would therefore be flowing about nine times faster than the cable laying vessel. Consequently, for the purposes of this assessment it can be assumed that the estimated number displaced by a cable

laying vessel represents the total number displaced for the duration of cable laying operations. The export cable may be installed in a single phase or two phases, therefore this impact could occur in a maximum of two winters. However, the total duration is the same in each case (single phase of 6 months or two phases of 3 months, separated by up to three years) and therefore the magnitude of impact is the same for both options.

64. In the ES, the potential consequence of displacement from construction vessels was based on a mortality rate of 5% for displaced individuals. NE requested that this should be increased to 10% (although note that the review in Annex 1 does not support such use of such a precautionary rate).

65. At the 5% mortality rate it was predicted that a maximum of between 2 to 4 birds would be expected to die across the equivalent of one entire winter period (September to April, although note this could occur in a single phase of 6 months or two phases of 3 months) as a result of any potential displacement effects from the offshore cable installation activities, which would be restricted to the equivalent of a single season, and only if cable laying takes place during these months. This would be doubled (4 to 8 birds) at the NE recommended rate of 10%. Even when compared to the smaller winter BDMPS for this species (10,177; Furness, 2015) it is clear that this highly precautionary assessment will generate an effect of negligible magnitude.

66. The construction works, specifically offshore cable laying, are temporary and localised in nature and the magnitude of effect has been determined as negligible. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

Norfolk Vanguard East

67. There is potential for disturbance and displacement of red-throated divers due to construction activity and associated vessel traffic. However, construction will not occur across the whole of the proposed wind turbine array area simultaneously or every day but will be phased with a maximum of two foundations expected to be installed simultaneously. Consequently, the effects will occur only in the areas where vessels are operating at any given point and not the entire NV East site.

68. For the ES a precautionary mortality rate of 5% mortality for individuals displaced by construction vessels was used. However, NE requested that a more precautionary morality rate of 10% should be used, and the following provides an update to the original assessment with also this mortality rate used.

69. During **autumn migration**, with a seasonal peak density of 0.09/km^2 and a precautionary 2 km radius of disturbance around each construction vessel, 2

individuals (0.09 x 12.56 x 2) could be at risk of displacement and up to 0.1 at risk of mortality (at 5%) and 0.2 (at 10%) in a maximum of two autumn periods.

70. At the average baseline mortality rate for red-throated diver of 0.228 the number of individuals expected to die in the autumn BDMPS is 3,027 (13,277 x 0.228). The addition of a maximum of 0.2 to this increases the mortality rate by 0.006%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the autumn migration period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

71. During **winter**, with a seasonal peak density of 0.06/km^2 and a precautionary 2 km radius of disturbance around each construction vessel, 2 individuals (0.06 x 12.56 x 2) could be at risk of displacement and up to 0.1 at risk of mortality (at 5%) and 0.2 at 10%) during a maximum of two winter periods.

72. At the average baseline mortality rate for red-throated diver of 0.228 (**Error! Reference source not found.**) the number of individuals expected to die in the winter BDMPS is 2,320 (10,177 x 0.228). The addition of a maximum of 0.2 to this increases the mortality rate by 0.008%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the winter period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

73. During **spring**, with a seasonal peak density of 0.26/km^2 and a precautionary 2km radius of disturbance around each construction vessel, 7 individuals (0.26 x 12.56 x 2) could be at risk of displacement and up to 0.3 at risk of mortality (at 5%) and 0.6 (at 10%) during a maximum of two spring periods.

74. At the average baseline mortality rate for red-throated diver of 0.228 (**Error! Reference source not found.**) the number of individuals expected to die in the spring BDMPS is 3,027 (13,277 x 0.228). The addition of a maximum of 0.6 to this increases the mortality rate by 0.02%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the spring period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

75. The combined nonbreeding impact of construction, with approximately 0.5 individuals (at 5%) or 1 (at 10%) at risk of construction displacement mortality, will

be similarly undetectable against background levels (this would increase the background mortality of the smallest BDMPS population by a maximum of 0.04%). Therefore, during the combined nonbreeding period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

Norfolk Vanguard West

76. There is potential for disturbance and displacement of red-throated divers due to construction activity, including wind turbine construction and associated vessel traffic. However, construction will not occur across the whole of the proposed wind turbine array area simultaneously or every day but will be phased with a maximum of two foundations expected to be installed simultaneously. Consequently, the effects will occur only in the areas where vessels are operating at any given point and not the entire NV West site.

77. For the ES a precautionary mortality rate of 5% mortality for individuals displaced by construction vessels was used. However, NE requested that a more precautionary morality rate of 10% should be used, and the following provides an update to the original assessment with the addition of this mortality rate.

78. During **autumn migration**, with a seasonal peak density of 0.01/km^2 and a precautionary 2 km radius of disturbance around each construction vessel, less than 1 individual (0.01 x 12.56 x 2 = 0.25) could be at risk of displacement and up to 0.01 (at 5%) and 0.02 (at 10%) at risk of mortality during a maximum of two autumn periods.

79. At the average baseline mortality rate for red-throated diver of 0.228 (**Error! Reference source not found.**) the number of individuals expected to die in the autumn BDMPS is 3,027 (13,277 x 0.228). The addition of a maximum of 0.02 to this increases the mortality rate by 0.006%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the autumn migration period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

80. During **winter**, with a seasonal peak density of 0.48/km^2 and a precautionary 2 km radius of disturbance around each construction vessel, 12 individuals (0.48 x 12.56 x 2) could be at risk of displacement and up to 0.6 (at 5%) and 1.2 (at 10%) at risk of mortality during a maximum of two winter periods.

81. At the average baseline mortality rate for red-throated diver of 0.228 (**Error! Reference source not found.**) the number of individuals expected to die in the winter BDMPS is 2,320 (10,177 x 0.228). The addition of a maximum of 1.2 to this increases the mortality rate by 0.06%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the winter period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

82. During spring, with a seasonal peak density of $0.37/km^2$ and a precautionary 2 km radius of disturbance around each construction vessel, 9 individuals (0.37 x 12.56 x 2) could be at risk of displacement and up to 0.5 (at 5%) and 1 (at 10%) at risk of mortality during a maximum of two spring periods.

83. At the average baseline mortality rate for red-throated diver of 0.228 (**Error! Reference source not found.**) the number of individuals expected to die in the spring BDMPS is 3,027 (13,277 x 0.228). The addition of a maximum of 1 to this increases the mortality rate by 0.04%. This magnitude of increase in mortality would not materially alter the background mortality of the population and would be undetectable. Therefore, during the spring period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

84. The combined nonbreeding impact of construction, with approximately 2.2 individuals at risk of construction displacement mortality, will be similarly undetectable against background levels (this would increase the background mortality of the smallest BDMPS population by 0.1%). Therefore, during the combined nonbreeding period, the magnitude of effect is assessed as negligible even on the basis of this highly precautionary approach. As the species is of high sensitivity to disturbance, the impact significance is **minor adverse**.

Norfolk Vanguard East and Norfolk Vanguard West

85. Although construction may occur on both NV East and NV West at the same time, the maximum number of simultaneous piling events would remain two. Therefore, the combined impact across the two sites would not exceed that assessed for the two sites alone and no further assessment is required.

1.1.3 Operational vessel movements

86. Vessel movements during the operation of the wind farm for maintenance activities have the potential to disturb red-throated divers. However, within the confines of

the wind farm sites and the 4 km buffer, the magnitude of displacement due to the wind farm itself (assessed as 90-100%) is such that there would be virtually no additional effect caused by vessel movements (i.e. almost all individuals will already have been displaced). Therefore, no further assessment for operational vessel movements within the wind farm sites (and buffers) is required.

87. The operation and maintenance port has not been confirmed at this stage. However, it is clear from consideration of the existing volume of shipping traffic through the region (Shipping and Navigation assessment, Appendix 15.01 vol 3, figures 15.1 and 15.2), which includes the Greater Wash SPA and Outer Thames Estuary SPA, that the addition of vessels transiting to and from the port and the wind farm (approx. 1.2 vessel movements per day) will have a negligible effect on the levels of shipping disturbance over and above the average of almost 100 vessel movements per day (derived from AIS data, and therefore not including smaller vessels).

2 ANNEX 1. RED-THROATED DIVER DISPLACEMENT AND CONSEQUENT MORTALITY: ASSESSMENT OF EVIDENCE

Are red-throated divers displaced from operational offshore wind farms?

88. Yes. Dierschke et al. (2016) reviewed studies published up to 2016 that compared seabird abundances within offshore wind farms post-construction with baseline data from before construction. Studies at ten different offshore wind farms found ‘strong avoidance’ by red-throated divers or ‘divers’ (a category that was predominantly red-throated divers but included some black-throated divers) at nine of these sites, and ‘weak avoidance’ at one site. These results imply consistent displacement from offshore wind farms.

89. Skov et al. (2018) reported the results of the ORJIP bird avoidance study at Thanet offshore wind farm. They obtained 82 radar tracks and 42 laser rangefinder tracks of red-throated divers, which would appear to provide an adequate sample size to assess macro-avoidance of that wind farm, but they do not report on the avoidance behaviour of that species as it was not one of the key species in that study. Percival and Ford (2018) stated that there was clear evidence that red-throated divers strongly avoid Kentish Flats Extension and a 0-500m zone around that wind farm. Mendel et al. (2019) used aerial survey data in a BACI design for an area of the German Bight that included hot-spots for red-throated divers (including a Special Protection Area designated with nonbreeding red-throated diver as the feature) and four offshore wind farms. Red-throated diver density decreased highly significantly within and around the wind farms post-construction. In contrast however, APEM (2016) found clear evidence of displacement of red-throated divers by London Array offshore wind farm during construction, but no clear evidence of displacement during operation. Similarly, Gill et al. (2018) found statistically significant displacement of red-throated divers from Greater Gabbard offshore wind farm during construction, but no statistically significant difference between pre-construction and post-construction, although the trend was in the direction of a displacement effect similar to that reported in other studies.

How strong is the displacement effect?

90. Petersen et al. (2006) estimated that displacement of red-throated divers occurred up to at least 2 km from the outer turbines at Horns Rev, with a decline of about 90% in diver numbers within the operational wind farm and a similar decline in the 2 km buffer around the outer turbines but no significant change beyond 2 km. Fox and Petersen (2006) provide a methodological framework for assessing the degree of habitat loss caused by displacement, and suggest that divers were significantly affected within 2 km of Horns Rev.

91. Percival (2013) reported an 82% decline in red-throated diver density within Thanet wind farm but no significant effect outside the wind farm. Percival (2014) concluded that displacement of red-throated divers by Kentish Flats offshore wind farm was not evident more than 1 km from the outer turbines.

92. Welcker and Nehls (2016) reported a reduction of 90% in red-throated diver density within Alpha Ventus offshore wind farm during construction and the first three years of operation compared with baseline numbers before construction. Diver abundance reached an asymptote at a distance of about 1.5 km from the outer turbines, suggesting no displacement effect beyond 1.5 km.

93. Peterson et al. (2014) noted red-throated diver densities at Horns Rev II were lower than pre-construction densities up to 13 km from the wind farm, but they considered that the difference was likely to be due to other factors, and suggested that displacement by the wind farm may occur up to about 5-6 km from the outer turbines. Diver density within 4 km of the wind farm was reduced by about 25%.

94. At Kentish Flats extension, the overall reduction in density was 89% within the wind farm and 70% in the 0-500 m buffer. However, there was no statistically significant effect detectable beyond 500 m from the outer wind turbines (Percival and Ford 2018).

95. Mendel et al. (2019) found overall a 94% reduction in red-throated diver abundance within 3 km of operational wind farms in the German Bight compared with the pre-construction baseline. When they separated displacement effects of ship traffic from those of offshore wind farm structures, they found a 71% reduction in red-throated diver densities in the area within 3 km of the operational offshore wind farms attributable to the offshore wind farm structure, with an additional displacement of at least a further 14% attributable to associated ship traffic. Mendel et al. (2019) found that statistically significant avoidance could be detected up to 12 km from the wind farms, although beyond 3 km the displacement was at much lower levels than closer to the wind farm.

96. APEM (2016) found a decrease in diver density up to at least 10 km from the wind farm during construction, whereas during operation the density was similar to that pre-construction and higher than pre-construction at 2 km from the wind farm, suggesting a displacement distance of <1.5 km.

97. Gill et al. (2018) estimated displacement of 75% from Greater Gabbard offshore wind farm during post-construction but the change was not statistically significant due to high variance in the data. However, their distribution maps indicate that there

was little or no displacement of divers from the immediate area surrounding the wind farm.

98. Whereas most studies consistently found a marked decrease in red-throated diver densities within operational wind farms when compared to pre-construction data (Table 2.1), the distance outside the wind farm over which diver densities were reduced varied greatly among sites (Table 2.1). At the extremes, Percival (2013) found no reduction in diver density outside Thanet offshore wind farm even within 500 m of the outer turbines, whereas Mendel et al. (2019) found a statistically detectable reduction in density up to 12 km from the outer turbines. This variation is unexplained. However, it might relate to ecological conditions or to the seascape/landscape of the site. Behaviour may vary seasonally, for example, depending on ecological constraints at different times of year, such as may arise during flight-feather moult when birds may become flightless. Birds might show greater avoidance distances where they are unconstrained. At sites where suitable or optimal habitat is limited, birds might show lower displacement distances because of constraints imposed by habitat availability. Alternatively, divers may show stronger avoidance of visible structures at sea where these are against an ‘empty’ background seascape. Where structures are in front of a cluttered background of coast, perhaps especially a coast with industrial development, the offshore turbines may appear less prominent and/or may be seen by divers as less threatening. The largest distances from offshore wind farms over which diver densities were reduced were in the German Bight, a very large area of open sea far from the coast. The smallest displacement distances from offshore wind farms were at sites close to the UK coast where anthropogenic influences on the coastal scenery are high (Thanet, Kentish Flats).

Table 2.1 Summary of distances from offshore wind farms over which displacement of red-throated divers was assessed to occur, and the percentage reduction in red-throated diver densities within operational wind farms by comparison with baseline pre-construction densities.

Wind farm	Distance from outer turbines over which diver density was significantly reduced (km)	Percentage reduction in diver density within wind farm area	Reference
Thanet	0.0	82	Percival 2013
Kentish Flats Extension	0.5	89	Percival and Ford 2018
Greater Gabbard	<1.0	(75)*	Gill et al. 2018
Kentish Flats	1.0	-	Percival 2014
Gunfleet Sands	1.0	-	Barker 2011
London Array	<1.5	<50	APEM 2016
Alpha Ventus	1.5	90	Welcker & Nehls 2016
Horns Rev 1	2.0	90	Petersen et al. 2006
North Hoyle	2.5	-	May 2008

Wind farm	Distance from outer turbines over which diver density was significantly reduced (km)	Percentage reduction in diver density within wind farm area	Reference
Lincs	2-6	-	Webb et al. 2015
Horns Rev 2	5.5	50	Petersen et al. 2014
Butendiek, Amrumbank, Nordsee Ost, Meerwind Süd/Ost, Dan Tysk	12.0	94	Mendel et al. 2019

*But not statistically significant due to high variance in data so a tentative estimate

99. To conclude this section, it seems reasonable to assume that 90% of red-throated divers will be displaced from offshore wind farm sites. It is less clear how many will be displaced from waters surrounding offshore wind farms, as this seems to be highly variable among sites. For sites in UK waters it seems that an appropriate precautionary estimate would be that 90% of red-throated divers will be displaced from a buffer zone of 1.5 km from an offshore wind farm, although site-specific data from UK suggest that displacement of birds may only extend as far as 500 m beyond the outer turbines at least in some cases (Table 2.1).

Is there evidence for habituation of divers to offshore wind farms?

100. If red-throated divers habituate over time to the presence of offshore wind farms, then habitat loss might be negligible in the long term. Leopold and Verdaat (2018) found some evidence for auks habituating to Luchterduinen offshore wind farm and suggested a methodology to assess the extent of such habituation. However, Petersen and Fox (2007) suggested that there was no evidence of habituation of red-throated divers at Horns Rev, at least in the first few years of operation. Percival (2010) suggested that red-throated divers were starting to habituate to Kentish Flats offshore wind farm, with increasing numbers entering the wind farm in 2009-10. However, subsequent data suggest no habituation at all at this site (Percival 2014). Mendel et al. (2019) found no evidence of any habituation of red-throated divers to offshore wind farms in the German Bight. From this we conclude that currently there does not seem to be any clear evidence of red-throated divers habituating to offshore wind farms.

What are the likely consequences of displacement for individuals?

101. Consequences of displacement of red-throated divers have been summarised by Dierschke et al. (2017). They state that displacement could influence individual divers if offshore wind farm barrier effects or habitat loss result in a change in the bird’s energy budget. Under some circumstances, though not all, displacement could increase energy costs, or could result in decreased energy intake. The former could arise if birds had to fly more to avoid offshore wind farms or to reach more distant

foraging areas. The latter could arise if displacement was to lower quality habitat where food capture rates were lower, or if displacement resulted in an increase in diver density with a consequent increase in intra-specific competition. Alternatively, displacement may have no effect on individuals if birds are displaced into equally good habitat so that their energy budget is unaffected, or if birds could buffer any impact on energy budget by adjusting their time budget (for example by spending a higher proportion of the time foraging rather than resting in order to compensate for an increase in energy budget). According to Dierschke et al. (2017) 'red-throated divers appear capable of utilising a range of marine habitats and prey species. They also tend to occur at relatively low densities and not in large aggregations. Consequently, reduced prey intake caused by increased density-dependent competition or interference would seem unlikely. Red-throated divers are highly mobile in winter which may mean that they are able to find alternative foraging sites following displacement.' Red-throated divers are mostly widely dispersed during the nonbreeding season, with fewer than 4 birds per km^2 (Dierschke et al. 2017), which makes it difficult to imagine that density-dependent competition for food could apply in this species during the nonbreeding season, as birds at such low densities would not be able to deplete their food resource (small fish such as sprats, young herring, sandeels; Guse et al. 2009, Dierschke et al. 2017). Dierschke et al. (2017) also state 'year-round energetic budgets are unknown but this information is key to understanding the possible consequences of displacement… if red-throated divers tend to be in poorer condition in the nonbreeding season when displacement is occurring, displacement could have an impact on survival and productivity. However, if individuals are in relatively good condition during the nonbreeding season and spend only a small proportion of their daily activity budget foraging, they may have the capacity to buffer against any additional energetic expense of displacement and barrier effects'. Dierschke et al. (2017) also noted that red-throated divers use staging areas during migration between breeding and wintering grounds and spend several weeks undergoing a post-breeding moult during which they are flightless. Displacement at moulting grounds or staging areas may have different consequences from displacement in wintering areas.

102. In the context of overwinter survival, it is relevant that in many seabird species, most mortality occurs during winter (e.g. Coulson et al. 1983). This may be caused by a variety of factors, such as winter storms (Anker-Nilssen et al. 2018). However, there is also evidence that seabirds tend to be heavier in winter than during the breeding season (e.g. Coulson et al. 1983). It is inferred from this that most seabirds have relatively little difficulty in finding enough food during the nonbreeding season so can achieve higher body condition that buffers against short periods of adverse weather conditions. For example, puffins are 20-30% heavier in winter than in

summer as a result of storing fat during the nonbreeding season, and the same is true of guillemots (Anker-Nilssen et al. 2018). If the same pattern occurs in red-throated divers, which seems likely, an implication is that their body condition would not be greatly affected by plausible levels of displacement or disturbance, since their time budgets may not be constrained during this period.

103. For red-throated diver, Horswill and Robinson (2015) recommend use of baseline age of first breeding at 3 years old, adult (third year and older) survival 0.84, immature (second year) survival 0.62 and juvenile (first year) survival 0.6. The lower survival rates of juveniles suggests that if there were impacts of displacement from offshore wind farms then those might be most likely to arise among juvenile birds rather than adults.

104. The annual mortality of adult red-throated divers is around 16% per annum and this will include mortality (if any) caused by human disturbance in marine environments that has been occurring in previous decades. The amount of general ship traffic has increased up to the present time, but has been high since the 1950s (IMO, Oskin 2014), while numbers of fishing vessels increased during the early 20th century but have decreased slightly in recent decades (Uberoi 2017). It is known that red-throated divers often tend to fly off when an approaching ship is about 1-2km away (Schwemmer et al. 2011). There is a case to be made that the net energy costs of flying away from approaching ships (and consequent loss of foraging time and opportunity) is likely to be considerably greater than the energy cost of avoiding static structures such as offshore wind turbines. Given that all offshore wind farms in UK North Sea waters combined represent an extremely small fraction of potential foraging habitat of red-throated divers within UK North Sea waters, it would seem appropriate to assess the plausible additional mortality caused by offshore wind farm displacement, barrier effects and associated increases in ship traffic (both during construction and operation) as also being extremely small in relation to the existing total annual mortality (also given that this total annual mortality already includes any impact of existing (baseline) ship disturbance impacts: in 2012 an average of 86 vessel transits were identified by Automated Identification System data per day in the waters off East Anglia[1]; MMO 2014). In that context, to suggest that displacement from an offshore wind farm might add 5% or more to the baseline mortality for all individuals that are displaced seems inconsistent with a total annual mortality of red-throated diver adults of only 16%. Furthermore, this magnitude of effect appears even less likely when it is considered that the baseline mortality already includes the impacts from existing shipping activities, which almost certainly

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[1] Note these data excluded commercial vessels less than 300 tonnes, recreational vessels, fishing vessels and military and government vessels on deployment.

cause disturbance to many red-throated divers many times per nonbreeding season. For example, Jarrett et al. (2018) reported that red-throated divers they watched in Orkney during the nonbreeding period (observations being of 1 to 3 individuals at a time in areas where vessel activity was likely) were subject to disturbance by vessels in 3 out of 30 five-minute observation periods. Out of a total of 7 disturbance incidents, birds flew away from the approaching vessel in 3 cases and swam away or dived in 4 cases. Such observations strongly suggest that disturbance by ships is likely to be of a much greater magnitude than displacement by offshore wind farms, and yet the impact of historical disturbance by ships must already be incorporated in the existing estimate of survival.

105. To set this in context, Goss-Custard et al. (2006a) assessed the impact of human disturbance of overwintering oystercatchers on mudflats and concluded that in winters with good feeding conditions, oystercatchers could be disturbed up to 1.5 times per hour before there was a reduction in their survival, whereas in winters with poor feeding conditions, being disturbed more than 0.5 times per hour resulted in an increased mortality risk. Madsen (1995) showed that pink-footed geese that were subject to high levels of human disturbance causing them to disperse from a spring staging area had lower breeding success than geese that were not subject to disturbance. By contrast, red-throated divers are almost certainly subject to multiple instances of disturbance due to vessel movements each winter and these evidently add up to less than a 16% per annum mortality (and obviously much less since much of the 16% per annum must be due to a wide range of natural mortality factors).

What are the likely consequences of displacement for the population?

106. Sutherland (1996) and Newton (1998) pointed out that for migrant birds, such as red-throated divers, population change following habitat loss in their nonbreeding area would depend on the relative strength of density-dependence in the breeding area and in the wintering area. If the population was regulated by density-dependent competition for breeding resources then habitat loss in the nonbreeding area may be unimportant. Goss-Custard et al. (1997) also pointed out that nonbreeding season habitat loss would only result in a decrease in a waterbird population if the population was subject to density-dependent competition for resources and population size was at carrying capacity of the environment.

107. Evidence strongly indicates that red-throated divers are limited by competition for safe breeding sites within range of foraging waters (Merrie 1978, Nummi et al. 2013, Rizzolo et al. 2014, Dahlen and Eriksson 2016), but they are probably not in competition for resources during the nonbreeding season (Dierschke et al. 2012, 2017). This would suggest that their population size will be limited by breeding habitat suitability and not by wintering habitat (Newton 1998). Loss of wintering

habitat would, therefore, have little or no impact on red-throated diver numbers unless habitat loss was so extensive that nonbreeding season habitat became a limiting factor for the population because their density increased so much that interference competition or prey depletion became a driving factor.

108. Topping and Petersen (2011) used an Individual Based Model to assess cumulative impact of displacement by offshore wind farms for the nonbreeding red-throated diver population migrating to Danish and Baltic waters. Their model made many assumptions and simplifications with very limited supporting evidence, but concluded that any population level cumulative impact of all operational and proposed offshore wind farms in Danish waters was likely to reduce diver numbers in the Baltic flyway by about 0.1%, whereas the cumulative impact of all operational and proposed offshore wind farms in Baltic waters might reduce numbers by 1.7% (Danish Energy Agency 2013).

109. Probably the most likely consequence is that displacement of red-throated divers will have effects which are too small to detect, as they are unlikely to be subject to density-dependent competition for resources during the nonbreeding season (Dierschke et al. 2017). Even though there are now many offshore wind farms in the southern North Sea and in the Baltic, the total area of these represents a very small fraction of the habitat used by nonbreeding red-throated divers throughout the southern North Sea and Baltic, so that cumulative habitat loss for red-throated divers is very small. The increase in density of red-throated divers caused by displacement away from offshore wind farms will therefore be extremely slight at the regional or biogeographic scale. However, the proportion of habitat lost may be much higher over certain small areas. For example, Mendel et al. (2019) estimated that displacement from offshore wind farms in the German Bight results in the effective loss of 8.8% of the Eastern German Bight SPA habitat for these birds. However, it is important to note that while the SPA boundary reflects historical distributions of red-throated divers, it does not necessarily follow that this represents the actual extent of suitable habitat in the area. So, displacement may move a proportion of birds out of the SPA, but this does not necessarily mean they will no longer be able to forage successfully.

110. Displacement from offshore wind farms must also be considered alongside disturbance caused by ships, as red-throated divers tend to fly away from approaching ships and that will increase their energy budget. Disturbance by ships seems likely to have the potential to affect red-throated diver energy budgets more than displacement from offshore wind farms, but these two effects may also interact. For example, birds displaced from offshore wind farms might move into habitat with higher levels of ship traffic.

111. To set the red-throated diver example in context, it is likely that red-throated divers do not experience prey depletion or interference competition while foraging (Dierschke et al. 2017). This means they are unlikely to be subject to density-dependent effects that would increase mortality if habitat loss (displacement) resulted in an increase in density in remaining areas. At the other extreme, it is known that many shorebirds that feed on mudflats are subject to strong interference competition and prey depletion (Goss-Custard et al. 2006b). These effects reflect the much more limited resource availability for waders feeding on inter-tidal areas, both spatially and temporally and the very much higher density of waders.

112. Estuarine habitat loss caused by barrages at Cardiff Bay and Rhymney resulted in an increase in mortality of 3.17% of displaced redshanks, a species known to be subject to strong density-dependent competition for food in winter due to both prey depletion and interference (Goss-Custard et al. 2006b). Oystercatchers are also known to be strongly susceptible to interference competition in winter on tidal mudflats. At Oosterschelde, Netherlands, two-thirds of the tidal mudflat area was destroyed by coastal engineering works (the Delta Works). There was no difference in oystercatcher winter adult survival or in movement rates before and after this habitat loss, although survival was reduced in severely cold winters compared to mild winters (Duriez et al. 2009). A study of the consequences of saltmarsh habitat loss for individually colour marked dark-bellied Brent geese followed the fate of displaced geese for 13 years after loss of saltmarsh habitat (Ganter et al. 1997). Displaced birds moved more often to less preferred sites that were not filled to capacity than did control birds. However, no significant differences in subsequent survival or fecundity of displaced birds could be found compared to control birds, although there may have been a slight but not statistically significant trend towards displaced birds performing less well than controls (Ganter et al. 1997). The researchers concluded that ‘if alternative sites are available there may be no significant fitness consequences to forced dispersal’ (i.e. displacement).

113. Based on our understanding of their winter feeding ecology and susceptibility to density-dependent competition, any effect of displacement of red-throated divers would be expected to be much less than seen in redshanks, and would be unlikely to be greater than seen in oystercatchers or dark-bellied Brent geese.

114. Despite the uncertainty about impacts on nonbreeding red-throated divers, the available evidence suggests that the most likely result is that there will be little or no impact on adult survival, and that any impact would probably be undetectable at the population level. A tracking study is currently underway which should provide the first estimates of the time-activity budgets of nonbreeding red-throated divers, and on their movements within winters (O’Brien et al. 2018). This should enable stronger

conclusions to be reached about the potential consequences of displacement for this species.

115. To conclude this section, we must acknowledge that the impact of displacement of red-throated divers by offshore wind farms is unknown. However, we do know that natural mortality of adult red-throated divers (including impacts of disturbance and displacement by ships) is low (16% per annum), and that disturbance/displacement of red-throated divers by offshore wind farms is likely to be very much less than disturbance/displacement by ships. This suggests that impacts of displacement from offshore wind farms are unlikely to represent levels of mortality anywhere near to the 16% mortality that occurs due to the combination of many natural factors plus disturbance/displacement by ships. In general, seabirds achieve higher body condition during the non-breeding season than they do while breeding, and the ecology of red-throated divers (foraging over large areas in a highly dispersed manner at low densities at sea) strongly suggests that density-dependent mortality is unlikely during the nonbreeding season. On that basis, it is unlikely that displacement by offshore wind farms would result in an additional mortality exceeding 1% of displaced birds, and any impact is more likely to be close to zero. Assuming that 1% of displaced birds die as a consequence of displacement would appear to be highly precautionary. In addition, strong evidence for density-dependent limitation of breeding numbers of red-throated divers suggests that a small increase in winter mortality would have little or no influence on the size of the red-throated diver population because it is likely to be breeding habitat suitability which sets the carrying capacity.

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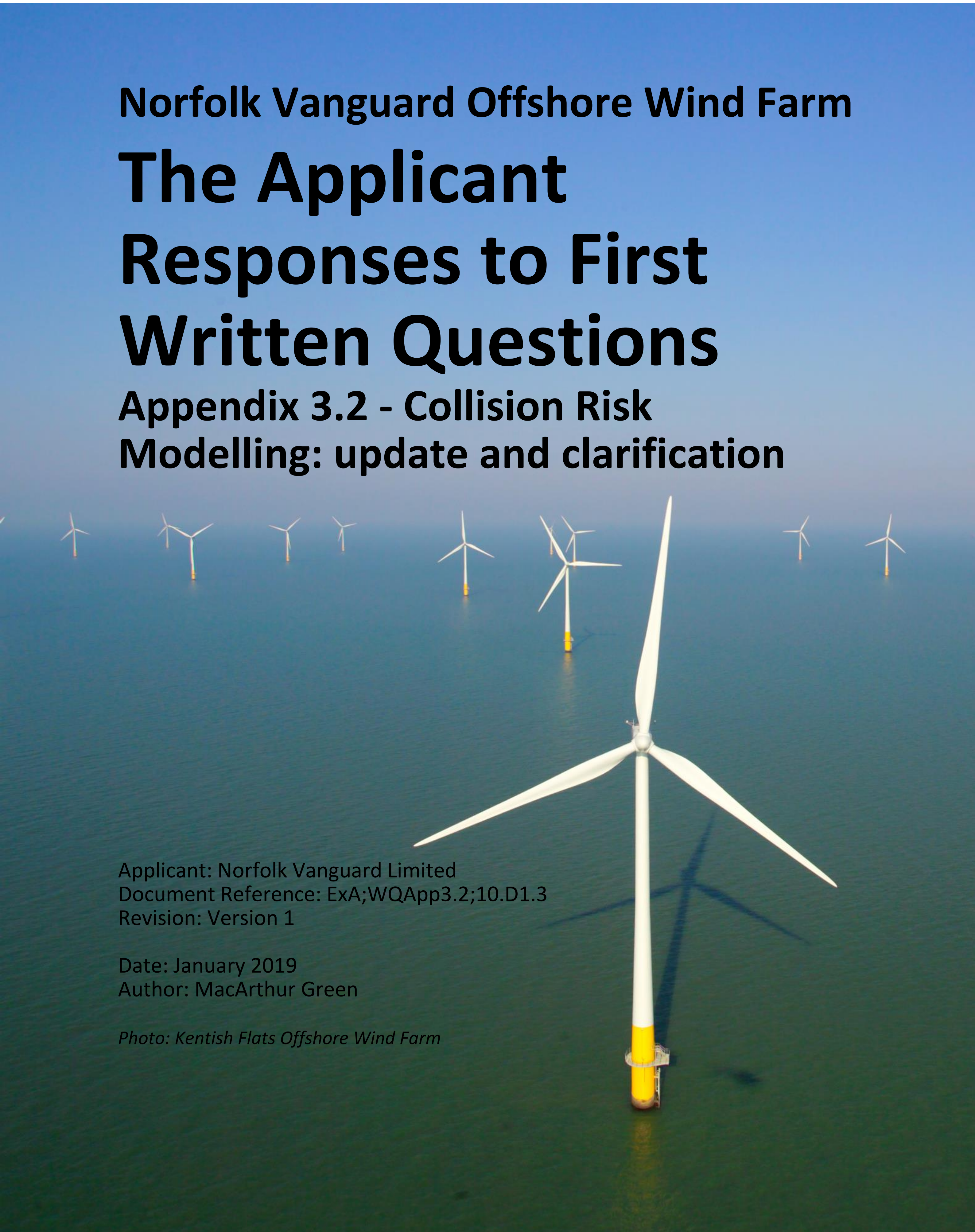
Norfolk Vanguard Offshore Wind Farm

The Applicant Responses to First Written Questions

Appendix 3.2 - Collision Risk Modelling: update and clarification



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Date	Issue No.	Remarks / Reason for Issue	Author	Checked	Approved
12/12/2018	01D	First draft for Norfolk Vanguard Ltd review	MT	JKL / RWF	EV
07/01/2019	02D	Final	MT	RWF	EV

Executive Summary

This note provides an update to the collision risk modelling (CRM) presented in the Norfolk Vanguard ES, and addresses comments received from Natural England (NE) in their relevant representation.

The aspects covered include derivation of seabird densities used an input to the CRM, complete tables of input parameters (to enable NE to check the results obtained), comparison of the CRM estimate for Norfolk Vanguard with those obtained using the Band (2012) spreadsheet and the Marine Scotland Science (MSS) commissioned stochastic version of the Band model, assessment of potential effects of collisions at Norfolk Vanguard on herring gull and presentation of the annual outputs calculated using alternative summary metrics.

The note only provides collision estimates for the Norfolk Vanguard project alone; cumulative and in-combination estimates will be provided in separate notes.

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Tables

Glossary

BDMPS	Biologically Defined Minimum Population Scale
CRM	Collision Risk Model
SPA	Special Protection Area
NE	Natural England
PVA	Population Viability Analysis
MSS	Marine Scotland Science
PCH	Potential Collision Height

1 INTRODUCTION

1. This note provides an update to the Norfolk Vanguard collision risk modelling (CRM) assessment (Vattenfall 2018) which addresses comments from Natural England (NE) in their Relevant Representation for the Norfolk Vanguard application and updates to the project design.

2. The detailed comments provided by NE and where they have been addressed are provided in Table 1.

Table 1 Comments on the collision risk modelling provided by Natural England (2018) in their relevant representation.

Paragraph	Comment	Response and section of this document where more detail provided
4.2.7	Natural England is aware that the non-stochastic CRM has been undertaken using R code for the Band model rather than by using the Band (2012) model spreadsheet. Whilst Annex 3 of Appendix 13.1 contains the majority of the CRM input data, it does not contain all of the required information (e.g. on the wind farm width and latitude used for both Vanguard East and Vanguard West). As a result, we have been unable to run the CRM using the Band (2012) spreadsheet and hence check the CRM results presented by the Applicant for the deterministic model outputs. The R code has not been supplied by the Applicant either. Therefore, in order for Natural England to fully appraise the CRM and hence reach conclusions on the level of impact due to Norfolk Vanguard alone, we again advise the Applicant that the full set of input parameters are required in order to be able to run the Band (2012) spreadsheets are presented.	The full set of parameters is provided in Tables A1.1 to A1.7 in Annex 1. Outputs from the deterministic Band (2012) CRM spreadsheet are included in Annex 2.
4.2.8	Natural England notes that the method that has previously been used in offshore wind farm assessments to estimate design-based bird density from a grid of images has been to calculate mean bird density from the images (i.e. number of birds counted / number of images). However our understanding is that Norfolk Vanguard has taken an alternative approach using median rather than mean densities. Natural England has identified areas of potentially significant concern regarding this approach, and would welcome further clarification from the Applicant regarding the approach taken. Please see our detailed advice in Annex 1.	Further discussion on the determination of appropriate measures of central tendency (i.e. mean or median) has been provided in section 1.1. Deterministic CRM outputs calculated using the mean density have been provided to allow NE to compare the outputs with those obtained using the median in Annex 3.
4.2.9 & 4.2.10	In the gannet and kittiwake CRM, the Applicant has used nocturnal activity rates calculated from recent reviews of evidence from tracking studies undertaken by Furness et al. Natural England has previously provided comments on drafts of these reviews and	The gannet nocturnal review has now been published (Furness et al. 2018), although this

Paragraph	Comment	Response and section of this document where more detail provided
	identified aspects that we did not agree with (particularly with regard to the kittiwake review), but we have not seen updated versions of this work. The work on gannet is referred to as Furness et al. (subm.) and that on kittiwake as Furness et al. (in prep.), which suggests that this work has not yet been accepted and is therefore not published and publically available. We are also uncertain of the journal that the gannet work has been submitted to, as no information is provided by the Applicant. Therefore our position remains as previously outlined to the Applicant: We currently do not have any agreed 'empirically derived' nocturnal activity factors that can be used with the Band model. We recognise from recent evidence presented e.g. by MacArthur Green (2015a) that nocturnal activity levels for some species may be lower than the levels that equate to the nocturnal activity factors currently used in CRM, however we also note that there is uncertainty about the empirical activity levels and uncertainty about how these might translate into nocturnal factors applicable to the Band model. We advise that CRM outputs covering a range of nocturnal activity factors are considered to account for the uncertainty/variability (in the same way as has been recommended for bird densities, avoidance rates and flight heights) and the suggested range of nocturnal flight activities to be considered within the Band model CRM are: 1-2 (equating to 0-25% nocturnal activity) for gannet and 2-3 (equating to 25-50% of nocturnal activity) for kittiwake (and the large gulls, which has been used by the Applicant).	recommended slightly different nocturnal activity rates than those used in the ES. Therefore, the CRM for gannet has been updated in line with the recommendations of Furness et al. (2018; Annex 4). The kittiwake review has not yet been published, however the nocturnal rates are not expected to change from those used in the ES. Nevertheless, kittiwake collisions are provided calculated using NE recommended rates (25% and 50%). Large gull collisions have also been presented using the NE recommended rates (25% and 50%), see Annex 4.
4.2.11	We note that herring gull is not fully assessed for CRM from Vanguard alone as it has been excluded due to the collision predictions currently being predicted to be less than 10 per year. The exclusion of herring gull from full assessment of collision impacts and hence consideration of cumulative impacts under EIA is of particular concern to Natural England. Given our concerns regarding the CRM, there is potential for herring gull collision predictions to increase above 10 collisions per year.	Further assessment for herring gull collision risk has been provided in section 1.6. This includes consideration of the various general points on the CRM outlined above.
Detailed comments – Seabird Collision Risk Modelling (CRM) for EIA and HRA		
4.1.1	**Collision risk models** The Applicant has undertaken the CRM using their own version of a stochastic CRM in order to present the uncertainty in the various CRM parameters (PCH, avoidance rates, densities, nocturnal activity) and also to cover off the development scenarios split across the Vanguard East and West sites. Whilst the Applicant's stochastic CRM simulations may be valid, Natural England notes that the potential use of such simulation was discussed with Natural England as part of the Evidence Plan Process. In response to this suggestion, Natural England advised the Applicant that we did not think that it would be possible at that stage to accept the proposal to use the Applicant's stochastic CRM and that the only way at this stage will be to present multiple tables	The additional deterministic tables have been provided in Annex 4.

Paragraph	Comment	Response and section of this document where more detail provided
	where the Applicant varied each parameter in turn using the Band (2012) model, and not all of them at once. As a result the Applicant agreed that they would provide outputs from their stochastic model along with the output tables requested by Natural England. The issues Natural England raised regarding the use of the Applicant's stochastic CRM still stand with regard to the stochastic simulations the Applicant has used and presented outputs for, namely:	
	1. We are not sure what R code the Applicant has for their stochastic CRM. The Marine Science Scotland (MSS) work to produce an agreed version of a stochastic CRM and shiny app was underway at the time of discussions with the Applicant, and we noted to the Applicant that whilst this would be correcting any bugs in the coding of the existing Masden (2015) model and making other fixes, it would also completely recode the Band (2012) model and the R code would be moved across to GIT which is a version control system for R - so there will be a detailed audit-trail of modifications and other developers will be able take over future development relatively easily. This it will provide a level of scrutiny that we don't have with the R code for the Band model available at the moment. Therefore, anything the Applicant used for Vanguard at this stage might not be the same as is eventually produced from the MSS work. The MSS work has recently completed and is available online. As we remain uncertain of the R code the Applicant has used, we do not know whether this is the same as the MSS model and means that potentially we would not end up with the same set of results from Vanguard as with the MSS work. So we could have another set of interim data. We note that now the MSS stochastic CRM is available there is a six month period of ongoing technical support. The general view of Natural England is that the stochastic CRM can be used for assessments, but that assessments should also provide the outputs from the standard Band model spreadsheets as well.	The MSS funded stochastic version of the Band (2012) model is now available and the results it produces have been compared with those produced using the version coded for Norfolk Vanguard. The two stochastic models and the Band model produce identical results (allowing for rounding variations). However, it is not straightforward to compare stochastic outputs for reasons which are detailed in section 1.5. It is important to note that these all relate to how random numbers are generated or inputted and none are structural. Selected outputs from the MSS model have been provided in section 1.5 to permit comparison with the Norfolk Vanguard and Band versions.
	2. Additionally we were not certain about the sampling distributions the Applicant has used and we note that these are not necessarily what is in the MSS stochastic CRM: - We previously noted to the Applicant that we have never seen the raw boot strapped aerial data – we are only ever presented with the mean monthly estimates along with the upper and lower confidence intervals.	The sampling distributions used in the Masden CRM (on which the MSS model is based) and those used in the Norfolk Vanguard model have been compared and

Paragraph	Comment	Response and section of this document where more detail provided
	- For the PCH – in our response to the Applicant we queried whether this will be drawn from the full set of BTO data or based on the confidence limits. - We noted to the Applicant that at present we have not thought about the distribution of nocturnal flight activities and so what would be suitable to use. - We also noted that we would also need to look in more detail at how all of these things would be integrated at the same time.	the merits of each discussed as relevant in a report commissioned for Natural England (NECR237; Trinder 2017). This report has been appended to this note (Annex 5) and the key points summarised in section 1.5.
	3. The Applicant has not presented the multiple tables of non-stochastic (i.e. Band 2012 model) outputs where each parameter in turn is varied that were requested by Natural England. We therefore advise that tables similar to those produced by Hornsea 2 in Appendix J of their Deadline 1 submission are produced for Norfolk Vanguard: https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010053/EN010053-001016-Appendix%20J_Collision%20Risk%20Modelling;%20Addressing%20Uncertainty%20Clarification%20Note.pdf	These tables have been provided in Annex 4.
	4. .During the Evidence Plan Process, Natural England was made aware that the non-stochastic CRM for Vanguard has been undertaken using R code for the Band model rather than by using the Band (2012) model spreadsheet. We therefore requested that the Applicant provide evidence to clearly demonstrate that the R code that is used is producing the same results as the Band spreadsheet version for all Band model options presented. Therefore, we requested that in the ES submission, the Applicant provides all of the input parameters used in their R model along with the R code in an Appendix, so that the results can then be checked. Whilst we note that Annex 3 of Appendix 13.1 contains the majority of the CRM input data, it does not contain information on the wind farm width and latitude used for both Vanguard East and Vanguard West. Therefore, we have been unable to run the CRM using the Band (2012) spreadsheet and hence check the CRM results presented by the Applicant for the deterministic model outputs. The R code has not been supplied by the Applicant either. Therefore, in order for us to be able to check the CRM and hence reach conclusions on the level of impact due to Vanguard alone, we again request that the full set of input parameters required in order to be able to run the Band (2012) spreadsheets are presented, i.e.: • Density of birds in flight within each of the Vanguard sites (noting comments above regarding use of the median and mean densities); • Proportion of birds at Vanguard rotor heights (using the Johnston et al. 2014a & b generic data given the issues noted by the Applicant with the site-specific data); • Bird parameters for each species (bird length, wing span, flight speed, nocturnal activity factor, flight type (flapping/gliding);	The comparison between the different implementations of the CRM (Band 2012, MSS and Norfolk Vanguard) have been provided in section 1.5, section 1.3 and Annex 2. Tables of input data are provided in Annex 1.

Paragraph	Comment	Response and section of this document where more detail provided
	• Proportion of flights upwind; • Wind farm data (latitude, number of turbines, width of wind farm, tidal offset); • Turbine data (model, number of blades, rotation speed, rotor radius, hub height, monthly proportion of time operational, maximum blade width, pitch).	
	Derivation of bird densities used in the CRM Natural England notes that the method that has previously been used in offshore wind farm assessments to estimate design-based bird density from a grid of images (as have been collected for Vanguard) has been to calculate mean bird density from the images (i.e. number of birds counted / number of images). Bootstrapping has typically then been applied to provide variance estimates and confidence limits (e.g. East Anglia 1). Our understanding of the approach in Section 4.2 of Appendix 13.1 (paragraph 14 – 16) is that the Applicant has: • Calculated monthly estimates in this way and averaged these to feed mean monthly densities into the displacement assessment (which we agree with); • Then also pooled all resampled estimates from data pertaining to any given month; • Used all of these estimates for stochastic CRMs; • Used the median of these estimates for CRMs not incorporating stochasticity. Based on this, Natural England has a number of queries/areas of uncertainty where it would welcome further clarification from the Applicant regarding the approach taken in order to reach conclusions around the applicability of the CRM outputs presented. These are: • We are uncertain as to why in the stochastic CRMs the Applicant has not used the monthly density estimate +/- 95% confidence limits to give a range of predicted collisions and would welcome clarity regarding this. • We consider the use of a bootstrapped median to estimate density in the non-stochastic CRM to be questionable, when a mean density already exists. We note that the point of bootstrapping is to estimate variance – the Applicant claim's that it has to be this way to enable comparison with stochastic CRM outputs, but we aren't looking to compare the two. Additionally, Appendix 13.1 (Offshore Ornithology Technical Appendix) defends this approach by saying that "all collision predictions accurately reflected the observed densities", but Natural England is not certain that this is true. The observed densities are those derived from the images (average of birds per image), whilst the bootstrapped data is a theoretical distribution of densities, from which the median gives an estimate of central tendency – therefore not a probability of being the 'true' density.	Further discussion on the appropriate values to use to describe seabird densities is provided in section 1.1.

Paragraph	Comment	Response and section of this document where more detail provided
	We note that the use of the median values means that lower monthly densities of birds are used and hence the predicted CRM results will be lower than if the mean densities are used.	
4.1.3	**Nocturnal flight activity rates for gannet and kittiwake** For CRM of Vanguard alone, the stochastic CRM assessment and that where just uncertainty in nocturnal activity was included, the Applicant has used nocturnal activity rates of: • 4.3% (S.E. 2.7%) for the breeding season and 2.3% (S.E. 0.4%) for the non-breeding season for gannet; and • 20% (S.E. 5%) for the breeding season and 17% (S.E. 1.5%) for the non-breeding season for kittiwake. The nocturnal activity factor input parameter used in the Band Model calculation of collision risk is a ranking score from 1 to 5, derived from an assessment of nocturnal activity in different species in Garthe & Huppop (2004), and not a 'nocturnal activity rate' per se. The Band model converts these factors to a percentage 0% (factor 1), 25% (2); 50% (3), 75% (4) and 100% (factor 5) that is applied to the densities of birds in flight collected from surveys during daylight hours to correct for a different pattern of flight behaviour (typically reduced) occurring during the night. Under this broad classification Garthe & Huppop (2004) assigned a factor of 2 to gannet, kittiwake a factor of 3 and herring gull and LBBG a factor of 3 (King et al., 2009, adds great black-backed gull as factor 3). The nocturnal activity rate figures used by the Applicant for gannet and kittiwake are based on the findings of recent reviews of evidence from tracking studies that have been undertaken by Furness et al. Natural England has provided comments on drafts of these reviews, where there were aspects that we did not agree with (particularly with regard to the kittiwake review). However, we note that we have not seen updated version of this work. We also note that the Applicant refers to the work on gannet as being Furness et al. (subm.) and the work on kittiwake as Furness et al. (in prep.), which suggests that this work has not yet been accepted and is therefore not published and publically available. We are also uncertain of the journal that the gannet work has been submitted to, as no information is provided by the Applicant. Therefore Natural England's position remains that which we previously outlined to the Applicant: we currently do not have any agreed 'empirically derived' nocturnal activity factors that can be used with the Band model. We recognise from recent evidence presented e.g. by MacArthur Green (2015a) that nocturnal activity levels for some species may be lower than the levels that equate to the nocturnal activity factors currently used in CRM, however we also note that there is uncertainty about the empirical activity levels and uncertainty about how these might translate into nocturnal factors applicable to the Band model.	The gannet review has now been published (Furness et al. 2018). It should be noted that the final publication recommended slightly higher nocturnal rates than used in the ES. Collisions estimated using the final recommended values have therefore been provided in Annex 4. The kittiwake review is not yet completed, thus while the rates used for this species in the ES are considered robust, the NE recommended ones have also been provided in this update (Annex 4).

Paragraph	Comment	Response and section of this document where more detail provided
	Therefore, Natural England advises that collision risk outputs covering a range of nocturnal activity factors are considered to account for the uncertainty/variability (in the same way as has been recommended for bird densities, avoidance rates and flight heights). The suggested range of nocturnal flight activities to be considered within the Band model CRM are: • Gannet: 1-2 (equating to 0-25% nocturnal activity) • Kittiwake: 2-3 (equating to 25-50% nocturnal activity) • Large gulls: 2-3 (equating to 25-50% nocturnal activity) (as has been used by the Applicant in the stochastic CRM and that where uncertainty in nocturnal activity has been considered).	See above.
4.1.4	**Assessment of herring gull CRM alone and cumulatively at EIA** We note that herring gull is not fully assessed for CRM from Vanguard alone as it has been excluded due to the collision predictions currently being predicted to be less than 10 per year. The exclusion of herring gull from full assessment of collision impacts and hence consideration of cumulative impacts under EIA is of particular concern to Natural England. We note the issues raised above regarding the appropriateness of the use of median values of bird density in the CRM and note that if the mean values of bird density are used in the CRM rather than the median values, then herring gull collision predictions may increase above 10 collisions per year.	Further assessment for herring gull is included in section 1.6.

1.1 Estimation of seabird flight densities

3. NE states that they '*consider use of a bootstrapped median to estimate density in the non-stochastic CRM to be questionable when a mean density already exists*' and that '*the Applicant claims that it has to be this way (use of the median rather than the mean) to enable comparison with stochastic CRM outputs, but we aren't looking to compare the two*'. These two aspects are addressed below.

4. The collision mortalities for the project presented in the ES and technical appendices were calculated using a stochastic implementation of the Band (2012) CRM. One of the inputs to the CRM is the density of birds in flight. To obtain measures of uncertainty in the density estimates for use in the stochastic model, the baseline aerial survey data were analysed using a non-parametric bootstrap method (as described in the Offshore Ornithology Technical Appendix). In brief, this method involved random resampling of the data assigned to each image which was collected during each aerial survey (monthly: 24 for NV West, 32 for NV East). This was done for each survey separately, and in such a way that each randomised resample comprised the same number of images as the original survey. In this manner a series

of datasets is generated which are akin to undertaking repeat surveys. This process was repeated 1,000 times for each survey and the range of densities for each species provides a measure of sampling uncertainty (expressed as variance or confidence intervals) as well as central values (i.e. mean and median). Following this, monthly summaries were calculated since the CRM is based around a calculation for each month.

5. To obtain the random density inputs required as inputs to the stochastic CRM two options were considered:

- Use the summary outputs (e.g. calculated mean and standard deviation) to generate simulated density values using an appropriate probability distribution e.g. mean and standard deviation as inputs to a random normal distribution function (or similar function); or
- Make use of the bootstrapped samples already generated (as described above) for the analysis as direct inputs to the CRM.

6. The latter option was used for two reasons: it simplified the analysis (the random samples were already available and there was no need to repeat this) and, more importantly, because these samples were considered more appropriate. This is due to the fact that the bootstrapped samples are drawn directly from the data and not a pre-defined probability distribution, the latter of which may not be a close match to the data. For example, across the two sampling months available (i.e. April in survey year 1 and April in survey year 2, etc.) there were numerous instances when the seabird density in year 1 was very low (or indeed zero) while that in the second year was higher. This is illustrated in Figure 1.

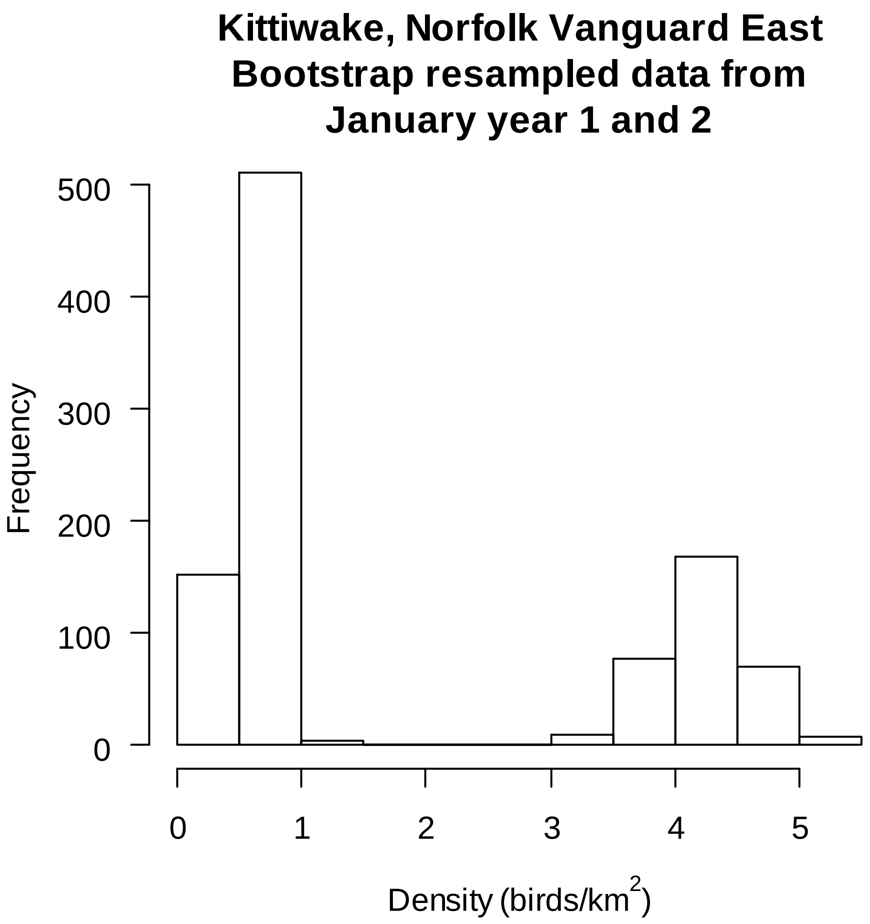

Figure 1. Example histogram of bootstrapped densities of kittiwakes in flight resampled from January survey data collected in 2013, 2014 and 2016.

7. Extracting the mean and standard deviation from the data in Figure 1 for use in a truncated normal distribution (bounded at zero) yields outputs such as those in Figure 2. It is clear that the truncated normal distribution provides a poor representation of the bootstrapped data.

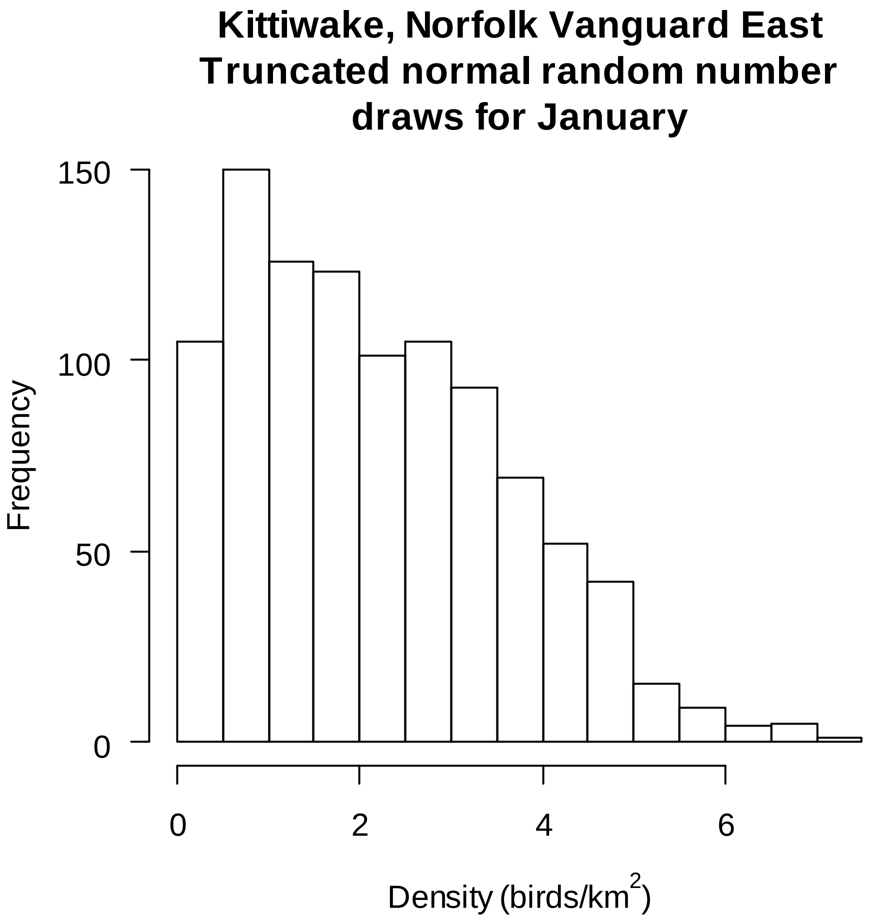

Figure 2. Example histogram of random draws from a truncated normal distribution (bounded at zero) using the mean (1.80) and standard deviation (1.73) calculated from the bootstrapped densities in Figure 1.

8. This was the primary reason for using the bootstrap resampled data directly in the stochastic CRM as it avoided instances such as this with unrepresentative input data.

9. There is a close relationship between the density inputs and the mortality outputs from the CRM, thus the same considerations in terms of how best to present them apply. In the presence of skewed data such as this, it is common practice to use the median as the central value rather than the mean, since the latter is much more heavily influenced by the infrequent but large values (i.e. outliers).

10. On this basis, the median was considered more appropriate for presenting the central mortality from the stochastic model, with the addition of the 95% confidence intervals (and also graphically in box plots, which clearly illustrate the skewed distributions).

11. Since the underlying distributions are present in the data irrespective of whether this is explicitly included in the CRM (i.e. using a stochastic model) or described using a single value for use in a deterministic CRM, it is clear that the median is also more appropriate for reporting the results of the non-stochastic CRM (i.e. deterministic collision predictions).

12. It is also important to note that the difference between the median and mean decreases as the data distribution becomes less skewed, with the two converging for symmetrical data. For seabird densities this convergence of mean and median occurred with more commonly observed species (for which the bootstrap resamples resembled normal distributions). Thus, for more abundant species there is very little difference between the mean and median outputs (and the choice of which to use is unimportant), while for less common species with more skewed distributions the median and mean diverge, with the median more representative (for the reasons outlined above). Thus, using the median ensures that reliable results are obtained irrespective of the underlying data.

13. Thus, the statement from NE that use of the median values means that lower densities are used (than if the mean densities are used) and therefore collision estimates will also be lower is not factually accurate. As explained above, the median is lower (and more representative) than the mean for right-skewed distributions, the same as the mean for symmetrical distributions and higher for left-skewed distributions. For the current data, the median density of gannets in flight on NV East was higher than the mean in September and December, similar in April and August and lower in the remaining months.

14. It seems likely that the reason why the question of whether the mean or median density is more appropriate for offshore wind farm impact assessment has not been considered and discussed in the past is that rather little consideration has been given to the role of uncertainty. The increased emphasis on simulation modelling has highlighted the relevance of the points raised above. It is also pertinent to note that once stochastic CRM has become the standard method used, with results presented as distributions rather than single values, it is likely that this question will become largely irrelevant.

1.2 CRM input parameters

15. The CRM input parameters used in the assessment were provided in annex 3 of the Norfolk Vanguard Offshore Ornithology technical appendix, however as noted by NE in their relevant representation there were three parameters omitted (wind farm latitude, wind farm width and percentage of flights expected to be upwind). These

have now been included in the wind farm details table. The input tables are in Annex 1 of this note.

1.3 Comparison of NV deterministic outputs calculated using the R CRM with those from the Band (2012) Excel

16. The deterministic collision predictions presented for Norfolk Vanguard (Appendix 13.1 Annexes 4 and 5) have been reproduced in Annex 2 along with pdf copies of the Band (2012) Excel input and output sheets. Both sets of outputs were obtained using generic flight height data to obtain the values for the PCH (i.e. option 2). However, to simplify the presentation of the spreadsheet outputs the option 2 PCH (as listed in Table A1.5) was entered on the input sheet as the PCH (i.e. where the option 1 value would typically be entered). This has no effect on the mortality calculated since option 1 and option 2 are identical in structure, differing only in the data source for PCH.

17. The collision predictions obtained using the scripted CRM as presented in the Norfolk Vanguard ES are the same as those obtained using the Band spreadsheet (with minor rounding differences). Thus, the function of the model used to estimate the collision mortalities presented in the ES are robust, subject to agreement over the most appropriate input parameters.

1.4 Deterministic CRM outputs for lower and upper parameter values

18. NE requested tables of deterministic CRM output with alternative parameter values as follows:

- Lower and upper 95% confidence intervals and mean estimates of birds in flight (NB: the ES provided stochastic outputs for the confidence intervals and the median);
- Lower and upper 95% confidence intervals for proportion at potential collision height (PCH; NB the ES provided stochastic outputs for these);
- Lower and upper 95% confidence intervals for the collision avoidance rates (NB the ES provided stochastic outputs for these);
- Nocturnal activity factors reduced from 2 to 1 (gannet) and 3 to 2 (gulls) (NB the ES used evidence based stochastic estimates for gannet and kittiwake).

19. Tables of output for gannet, kittiwake, lesser black-backed gull, herring gull and great black-backed gull are provided in Annex 4.

1.5 Comparison of NV CRM outputs with those obtained using the MSS CRM

20. Marine Scotland Science (MSS) commissioned a project to implement the Band (2012) CRM in a simulation format suitable for producing probabilistic mortality outputs. The stochastic CRM is available on the MSS website to use online and also to download and use in conjunction with the R programming software. The latter has been used here to enable a comparison of the outputs obtained from the MSS model and that coded (also in R) for the Norfolk Vanguard assessment (hereafter the NV model). This was undertaken to provide NE with reassurance that the two models operate in the same manner.

21. There are two aspects to this comparison; the methods used to generate random variables for input to the calculations and the calculations themselves. For the two models to be fully compared it is necessary to consider both aspects.

22. The MSS and NV models do not use the same probability distributions for all the stochastic parameters. Of particular note is the fact that the MSS model (which retains aspects of the Masden (2015) version of the CRM) uses the truncated normal distribution to generate several of the simulated parameter values and this can result in biased outputs for data with an underlying skewed distribution (see Trinder 2017 for further discussion and illustration of this point). For these reasons the NV model uses more reliable distributions for data which have can have central values close to boundaries (e.g. zero for densities and zero or one for proportions such as avoidance rates).

23. In addition, the MSS model does not allow nocturnal flight activity rates to be entered for each month separately, but rather only as a single value applied in all months. In contrast the NV model allows the seasonal variation identified in Furness et al. (2018) to be incorporated.

24. As a consequence, the two models cannot be directly compared in terms of their stochastic outputs, however the structure and collision calculations can be compared for deterministic outputs (i.e. with each input parameter’s standard deviation set to to zero).

25. To simplify this, the MSS model was run as if using option 1, although because the actual PCH values entered were those from Johnston et al. (2014a,b) the outputs are therefore in fact equivalent to the option 2 results reported using the NV model.

26. Thus, the deterministic CRM outputs for the 9 MW turbine presented in the Norfolk Vanguard technical appendix were compared with those obtained using the MSS model with all the standard deviations set to zero.

27. The NV model outputs for kittiwake calculated using option 2 for Norfolk Vanguard East (Table 6 in Annex 4 of Technical Appendix 13.1) are reproduced below alongside those generated by the MSS model (Table 2). Input parameters were those provided in Annex 1.

Table 2. Monthly kittiwake collisions calculated using the NV model and the MSS model (note no randomised parameters).

Model	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
NV	39.78	41.37	19.67	5.79	16.68	3.71	1.78	0	0	1.91	65.1	27.99
MSS	39.81	41.39	19.63	5.79	16.61	3.68	1.81	0	0	1.93	65.17	28.06

28. Although only kittiwake outputs have been presented here, the same agreement of results was obtained for each species. It is clear from the outputs in Table 2 that the two models produce almost identical results, with only minor differences due to rounding.

1.6 Herring gull collision risk assessment

29. Norfolk Vanguard is 92 km from the nearest breeding colony for herring gull. This species has a mean maximum foraging range of 61 km, and a maximum of 92 km, therefore the likelihood that herring gulls breeding at Alde-Ore Estuary SPA would reach the Norfolk Vanguard site is extremely small. Consequently, the breeding season impact on herring gull has been assessed against a reference population estimated using the same approach as that used in the ES for other species for which breeding adults were considered unlikely to be present. This is based on the observation that immature birds tend to remain in wintering areas. Thus, the number of immature birds in the relevant populations during the breeding season may be estimated as the proportion of the relevant biologically defined minimum population scale (BDMPS) season (the one immediately preceding the breeding season) which are sub-adults. Thus, the breeding season reference population can be calculated as 66.4% (the proportion of sub-adults in the population, Table 3) of the nonbreeding BDMPS populations of herring gull. This yields a breeding season population of nonbreeding herring gull of 309,763 (nonbreeding BDMPS for the UK North Sea and Channel, 466,511 x 66.4%). The nonbreeding season reference population was 466,511 (Furness 2015).

30. The impacts of mortality caused by collisions on the populations are assessed in terms of the change in the baseline mortality rate which could result. It has been assumed that all age classes are equally at risk of collisions (i.e. in proportion to their presence in the population), therefore it is necessary to calculate an average baseline mortality rate for all age classes for each species assessed. These were

calculated using the different survival rates for each age class and their relative proportions in the population.

31. The first step is to calculate an average survival rate. The demographic rates for each species were taken from reviews of the relevant literature (e.g. Horswill and Robinson, 2015) and recent examples of population modelling (e.g. EATL, 2016). The rates were entered into a matrix population model to calculate the expected proportions in each age class. For each age class the survival rate was multiplied by its proportion and the total for all ages summed to give the average survival rate for all ages. Taking this value away from 1 gives the average mortality rate. The demographic rates and the age class proportions and average mortality rates calculated from them are presented in Table 3.

Table 3 Average mortality across all age classes. Average mortality calculated using age specific demographic rates and age class proportions.

Species	Parameter	Survival (age class)					Productivity	Average mortality
		0-1	1-2	2-3	3-4	Adult		
Herring gull	Demographic rate	0.798	0.834	0.834	0.834	0.834	0.92	0.174
	Population age ratio	0.235	0.175	0.142	0.110	0.337	-	

32. Table 4 provides the baseline survival rates, the relevant breeding season and nonbreeding season BDMPS and the percentage increase in mortality for each seabird species due to collisions.

Table 4. Percentage increases in the background mortality rate of seasonal and annual populations due to predicted collisions (option 2) calculated with stochasticity in density, avoidance rate, flight height and nocturnal activity for the worst case 9MW turbine and species specific worst case project scenario. Note that the annual mortalities have been assessed against both the biogeographic populations and the largest BDMPS (as advised by Natural England) in order to bracket likely effects.

Species		Herring gull		
		Median	Lower c.i.	Upper c.i
Baseline average mortality		0.174		
Breeding season	Reference population	309,763		
	Seasonal mortality	0	0	0
	Increase in background mortality (%)	**0**	0	0
Wintering	Reference population	466,511		
	Seasonal mortality	5.17	0	172.07
	Increase in background mortality (%)	0.006	0	0.212
Annual – largest BDMPS	Reference population	466,511		
	Seasonal mortality	5.17	0	172.07
	Increase in background mortality (%)	0.006	0	0.212
Annual - biogeographic	Reference population	1,098,000		
	Seasonal mortality	5.17	0	172.07
	Increase in background mortality (%)	**0.003**	0	0.09

33. The median collision prediction for herring gull in all seasons and also summed across the year resulted in increases in background mortality well below 1%. Therefore, the magnitude of effects due to collision mortality for herring gull is considered to be negligible for this low sensitivity species resulting in an impact significance of **negligible adverse**.

1.7 Comparison of annual mortality estimates calculated as the sum of monthly medians, median of months and sum of monthly means

34. In the ES the annual median collision estimate was obtained as the sum of the median value calculated for each individual month. During discussions with the developer of the MSS stochastic CRM it became evident that under certain circumstances, such as if the monthly estimates were heavily skewed, this method would not preserve the complete range of uncertainty associated with each monthly estimate. Consequently, the annual total obtained as the sum of the 12 monthly median estimates will not necessarily equal the median of the annual totals for each model iteration (i.e. the median of the sum of the 12 monthly estimates generated during each iteration of the model).

35. For clarity the two approaches to obtaining the annual median total are:

- Sum of the monthly medians = median estimate for January, plus the median estimate for February, plus the median estimate for March, etc. (i.e. the sum of the median for each month),
- Median of the summed months = median calculated for the annual total for each model iteration (i.e. the sum of January to December for model iteration 1, for model iteration 2, etc.).

36. Note that the monthly estimates are unchanged, but the method to combine these to obtain an annual total is different.

37. The annual totals obtained using the different summation approaches, and the confidence intervals associated with each are provided in Table 5. The confidence intervals for the mean annual estimate have not been included because these are the same as those reported for the medians (in columns 1 and 2 of Table 5), and can also be calculated by either method.

Table 5. Comparison of alternative methods for calculating annual collisions for stochastic collision predictions for Norfolk Vanguard East and West.

Species	Site	Median of summed months for each model iteration (lwr-upr 95% c.i.)	Summed monthly medians (lwr-upr 95% c.i.)	Mean
Gannet	NV East	110.63 (14.79 - 524.03)	142.52 (75.26 - 327.3)	159.34
	NV West	44.73 (7.71 - 205.36)	62.78 (35.56 - 105.94)	65.04
Kittiwake	NV East	158.44 (22.43 - 859.65)	315.92 (90.12 - 458.3)	276.55
	NV West	58.54 (6.01 - 225.92)	81.45 (56.53 - 113.21)	82.3
Lesser black-backed gull	NV East	9.1 (0 - 99.49)	20.27 (3.31 - 49.61)	21.7
	NV West	27.35 (0 - 150.09)	40.03 (16.02 - 81.33)	42.34
Herring gull	NV East	5.17 (0 - 172.06)	17.06 (3.11 - 131.35)	37.1
	NV West	1.42 (0 - 11.84)	2.02 (0 - 8.64)	2.59
Great black-backed gull	NV East	19.97 (1.43 - 451.72)	65.12 (7.05 - 346.17)	107.14
	NV West	22.15 (0 - 138.68)	37.2 (15.57 - 69.8)	38.68

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Annex 1. CRM input tables

Table A1.1. Norfolk Vanguard East monthly median densities (and 95% confidence intervals) of birds in flight used in the collision risk modelling.

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Red-throated Diver	0 (0-0)	0 (0-0)	0 (0-0.119)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0.216)	0 (0-0)
Fulmar	0.078 (0-0.412)	0.039 (0-0.251)	0.079 (0-0.349)	0.033 (0-0.373)	0.283 (0-0.866)	0 (0-0.522)	0 (0-0.098)	0.228 (0.046-0.539)	0.105 (0-0.434)	0.031 (0-0.184)	0.163 (0-0.401)	0.123 (0-0.448)
Gannet	0 (0-0.137)	0.031 (0-0.188)	0 (0-0.094)	0.031 (0-0.124)	0 (0-0.2)	0.18 (0-0.678)	0 (0-0.074)	0.137 (0-0.317)	0.276 (0-0.552)	0.123 (0.024-0.46)	1.168 (0.678-5.052)	1.142 (0-1.693)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0.095)	0 (0-0.237)	0 (0-0)	0 (0-0)	0 (0-0)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0.093)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.026 (0-0.193)	0 (0-0.092)	0 (0-0)	0 (0-0)
Kittiwake	0.686 (0.343-4.82)	0.753 (0.193-1.647)	0.305 (0-2.833)	0.088 (0-1.49)	0.233 (0.033-0.971)	0.052 (0-0.261)	0.025 (0-0.141)	0 (0-0.137)	0 (0-0.11)	0.031 (0-0.337)	1.141 (0-1.942)	0.491 (0.138-1.614)
Black-headed Gull	0 (0-0.155)	0 (0-0)	0 (0-0.131)	0 (0-0)	0 (0-0.1)	0 (0-0)	0 (0-0.098)	0 (0-0)	0 (0-0)	0 (0-0.092)	0 (0-0.163)	0 (0-0)
Little Gull	0 (0-0)	0 (0-0.069)	0 (0-0)	0 (0-0)	0 (0-0.647)	0 (0-0)	0 (0-0)	0.063 (0-0.603)	0 (0-0.078)	0 (0-0)	0 (0-0.326)	0 (0-0)
Common Gull	0.032 (0-0.206)	0 (0-0)	0 (0-0.187)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0.215)	0 (0-0.092)	0 (0-0.092)
Lesser Black-backed Gull	0.034 (0-0.272)	0 (0-0.125)	0 (0-0.119)	0 (0-0.098)	0 (0-0)	0 (0-0)	0 (0-0.123)	0.127 (0-0.412)	0 (0-0)	0 (0-0.072)	0 (0-0.123)	0 (0-0.157)
Herring Gull	0.103 (0-1.788)	0 (0-0)	0 (0-0.119)	0 (0-0.093)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0.216)	0 (0-0.207)
Great Black-backed Gull	0.158 (0.032-3.382)	0 (0-0.125)	0 (0-0.093)	0 (0-0.131)	0 (0-0)	0 (0-0)	0 (0-0)	0.016 (0-0.507)	0 (0-0)	0 (0-0)	0 (0-0.431)	0.123 (0-0.394)

Table A1.2. Norfolk Vanguard West monthly median densities (and 95% confidence intervals) of birds in flight used in the collision risk modelling.

Species	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Red-throated Diver	0.062 (0-0.185)	0 (0-0.093)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0.093)
Fulmar	0.062 (0-0.248)	0.062 (0-0.185)	0.031 (0-0.155)	0.032 (0-0.154)	0.031 (0-0.093)	0 (0-0.187)	0 (0-0.248)	0.046 (0-0.465)	0.027 (0-0.093)	0.155 (0-0.963)	0.054 (0-0.163)	0 (0-0.093)
Gannet	0 (0-0.093)	0.031 (0-0.093)	0 (0-0.187)	0 (0-0)	0 (0-0.093)	0.031 (0-0.093)	0.062 (0-0.279)	0.08 (0-0.372)	0.053 (0-0.186)	0.28 (0-0.965)	0.649 (0.371-0.951)	0 (0-0)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.027 (0-0.093)	0 (0-0)	0 (0-0)	0 (0-0)
Kittiwake	0.093 (0-0.247)	0.062 (0-0.185)	0.156 (0.031-0.434)	0 (0-0.255)	0.062 (0-0.186)	0.249 (0.062-0.591)	0 (0-0.372)	0.093 (0-0.217)	0 (0-0.24)	0.093 (0-0.311)	0.362 (0.062-0.896)	0 (0-0.093)
Black-headed Gull	0 (0-0)	0 (0-0.185)	0 (0-0.093)	0 (0-0.124)	0 (0-0)	0 (0-0)	0 (0-0.072)	0 (0-0)	0 (0-0)	0.031 (0-0.125)	0 (0-0)	0 (0-0)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.027 (0-0.093)	0 (0-0)	0.054 (0-0.154)	0 (0-0)
Common Gull	0 (0-0)	0 (0-0.154)	0 (0-0.156)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0.08)	0.031 (0-0.093)	0.124 (0-0.402)	0 (0-0)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	0 (0-0.093)	0 (0-0.093)	0 (0-0)	0.093 (0-0.28)	0.109 (0-0.362)	0.186 (0.031-0.453)	0.013 (0-0.342)	0.093 (0-0.311)	0 (0-0)	0 (0-0)
Herring Gull	0 (0-0)	0 (0-0.092)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.027 (0-0.093)	0 (0-0)
Great Black-backed Gull	0.031 (0-0.154)	0.093 (0-0.247)	0 (0-0.093)	0 (0-0)	0 (0-0.093)	0 (0-0)	0 (0-0.124)	0 (0-0.186)	0.134 (0.009-0.321)	0 (0-0.124)	0.054 (0-0.217)	0 (0-0)

Table A1.3. Norfolk Vanguard East. Total number of birds with an estimated flight height, number at collision height (>=22 m) and proportion at collision height. Figures provided for all birds within the 4 km buffer and just those within the wind farm boundary.

	Within 4km buffer			**Within wind farm**		
Species	**No. height estimates**	**No. >=22m**	**Proportion >=22m**	**No. height estimates**	**No. >=22m**	**Proportion >=22m**
Red-throated Diver	6	3	0.500	5	3	0.600
Fulmar	274	4	0.015	113	1	0.009
Gannet	538	77	0.143	263	45	0.171
Arctic Skua	6	2	0.333	4	1	0.250
Great Skua	14	5	0.357	7	4	0.571
Kittiwake	942	219	0.232	437	102	0.233
Black-headed Gull	13	3	0.231	8	2	0.250
Little Gull	33	2	0.061	24	1	0.042
Common Gull	25	7	0.280	11	4	0.364
Lesser Black-backed Gull	90	26	0.289	26	5	0.192
Herring Gull	93	17	0.183	50	12	0.240
Great Black-backed Gull	204	39	0.191	112	9	0.080

Table A1.4. Norfolk Vanguard West. Total number of birds with an estimated flight height, number at collision height (>=22 m) and proportion at collision height. Figures provided for all birds within the 4 km buffer and just those within the wind farm boundary.

	Within 4km buffer			**Within wind farm**		
Species	**No. height estimates**	**No. >=22m**	**Proportion >=22m**	**No. height estimates**	**No. >=22m**	**Proportion >=22m**
Red-throated Diver	8	3	0.375	4	0	0.000
Fulmar	88	13	0.148	37	4	0.108
Gannet	116	17	0.147	51	6	0.118
Arctic Skua	1	0	0.000	0	0	0.000
Great Skua	4	2	0.500	1	0	0.000
Kittiwake	206	74	0.359	75	27	0.360
Black-headed Gull	21	17	0.810	3	2	0.667
Little Gull	7	1	0.143	5	1	0.200
Common Gull	32	11	0.344	15	4	0.267
Lesser Black-backed Gull	44	16	0.364	11	5	0.455
Herring Gull	10	4	0.400	3	2	0.667
Great Black-backed Gull	57	17	0.298	25	8	0.320

Table A1.5. Proportions at collision height (>=22 m) from Johnston et al. (2014).

	Proportion at collision height (>=22m)			
Species	**Maximum likelihood**	**Median**	**Lower confidence interval**	**Upper confidence interval**
Red-throated Diver	0.047	0.046	0.010	0.320
Fulmar	0.006	0.005	0.000	0.073
Gannet	0.102	0.104	0.047	0.173
Arctic Skua	0.018	0.019	0.010	0.086
Great Skua	0.044	0.047	0.025	0.151
Kittiwake	0.124	0.124	0.093	0.147
Black-headed Gull	0.114	0.108	0.042	0.232
Little Gull	0.125	0.114	0.041	0.245
Common Gull	0.188	0.202	0.159	0.276
Lesser Black-backed Gull	0.249	0.249	0.171	0.408
Herring Gull	0.285	0.287	0.216	0.400
Great Black-backed Gull	0.291	0.310	0.247	0.420

Table A1.6. Species biometrics used in the collision risk modelling. Note that nocturnal activity factors are the generic ones derived from Garthe and Hüppop (2004). In collision modelling simulations which included uncertainty in nocturnal activity the values used for gannet, kittiwake, lesser black-backed gull, herring gull and great black-backed gull were as detailed in Technical Appendix 13.1.

Species	**Body length (m)**	**Wingspan (m)**	**Flight speed (m/s)**	**Nocturnal activity factor**	**Flight type (flapping=0, gliding=1)**	**Avoidance rate (%)**	**Flights upwind (%)**
Red-throated Diver	0.73	1.30	17.0	0.50	0	98	50
Fulmar	0.48	1.07	13.0	0.75	0	98	50
Gannet	0.94	1.72	14.9	0.25	0	98.9	50
Arctic Skua	0.44	1.18	13.3	0.00	0	98	50
Great Skua	0.56	1.36	14.9	0.00	0	98	50
Kittiwake	0.39	1.08	13.1	0.50	0	98.9	50
Black-headed Gull	0.37	1.10	11.9	0.50	0	99.2	50
Little Gull	0.26	0.78	12.2	0.25	0	99.2	50
Common Gull	0.42	1.30	13.4	0.50	0	99.2	50
Lesser Black-backed Gull	0.58	1.42	13.1	0.50	0	99.5	50
Herring Gull	0.60	1.44	12.8	0.50	0	99.5	50
Great Black-backed Gull	0.71	1.58	13.7	0.50	0	99.5	50

Table A1.7. Wind farm and turbine specifications used in the collision risk modelling.

Turbine output (MW)	No. of rotor blades	RPM	Rotor radius (m)	Hub height above HAT (m)	Predicted operation time (%)	Max. blade width (m)	Mean blade pitch (deg.)	No. of turbines	Latitude		Wind farm width(km)	
									NV East	NV West	NV East	NV West
9	3	11.26	85.0	107.0	90	7.0	15	200	52.2	52.9	22.3	17.7
20		5.05	151.5	173.5		10.0		90				

Annex 2. Comparison of Norfolk Vanguard deterministic CRM outputs and Band (2012) spreadsheet outputs

Table A2.1. Norfolk Vanguard East (1800MW). Deterministic collision mortality for the 9 MW turbine calculated using Band CRM Option 2.

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	0	0	0	0	0	0	0	0	0	0	0	0	0
Fulmar	0.47	0.21	0.5	0.2	1.87	0	0	1.48	0.64	0.19	0.96	0.74	7.26
Gannet	0	1.55	0	2.02	0	13.24	0	9.52	16.98	7.01	58.14	54.96	163.42
Arctic Skua	0	0	0	0	0	0	0	0	0	0	0	0	0
Great Skua	0	0	0	0	0	0	0	0	0.92	0	0	0	0.92
Kittiwake	39.78	41.37	19.67	5.79	16.68	3.71	1.78	0	0	1.91	65.1	27.99	223.78
Black-headed Gull	0	0	0	0	0	0	0	0	0	0	0	0	0
Little Gull	0	0	0	0	0	0	0	2.28	0	0	0	0	2.28
Common Gull	2.22	0	0	0	0	0	0	0	0	0	0	0	2.22
Lesser Black-backed Gull	2	0	0	0	0	0	0	8.9	0	0	0	0	10.9
Herring Gull	6.92	0	0	0	0	0	0	0	0	0	0	0	6.92
Great Black-backed Gull	12.4	0	0	0	0	0	0	1.5	0	0	0	9.5	23.4

Table A2.2. Norfolk Vanguard West (1800 MW). Deterministic collision mortality for the 9 MW turbine calculated using Band CRM Option 2.

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	3.23	0	0	0	0	0	0	0	0	0	0	0	3.23
Fulmar	0.37	0.34	0.2	0.2	0.2	0	0	0.3	0.16	0.97	0.32	0	3.06
Gannet	0	1.52	0	0	0	2.29	4.62	5.57	3.29	16.01	32.28	0	65.58
Great Skua	0	0	0	0	0	0	0	0	0.93	0	0	0	0.93
Kittiwake	5.38	3.4	10.03	0	4.44	17.69	0	6.48	0	5.82	20.65	0	73.89
Black-headed Gull	0	0	0	0	0	0	0	0	0	1.17	0	0	1.17
Little Gull	0	0	0	0	0	0	0	0	0.85	0	1.4	0	2.25
Common Gull	0	0	0	0	0	0	0	0	0	2.36	8.58	0	10.94
Lesser Black-backed Gull	0	0	0	0	0	6.68	7.95	13.04	0.86	5.85	0	0	34.38
Herring Gull	0	0	0	0	0	0	0	0	0	0	1.8	0	1.8
Great Black-backed Gull	2.44	6.92	0	0	0	0	0	0	11.6	0	4.21	0	25.17

COLLISION RISK ASSESSMENT
Sheet 1 - Input data

🟩 used in overall collision risk sheet
🟩 used in migrant collision risk sheet
🟧 used in single transit collision risk sheet or extended model
🟪 used in available hours sheet
🟨 used in large array correction sheet
🟥 not used in calculation but stated for reference

Bird data	**Units**	**Value**	**Data sources**
Species name		**Gannet**	
Bird length	m	0.94	
Wingspan	m	1.72	
Flight speed	m/sec	14.9	
Nocturnal activity factor (1-5)		2	
Flight type, flapping or gliding		flapping	

Data sources

Bird survey data			Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Daytime bird density	birds/sq km		0	0.031	0	0.031	0	0.18	0	0.137	0.276	0.123	1.168	1.142
Proportion at rotor height	%	10.2%												
Proportion of flights upwind	%	50.0%												

Data sources

Birds on migration data			
Migration passages	birds		
Width of migration corridor	km		
Proportion at rotor height	%		
Proportion of flights upwind	%		

Windfarm data		**Units**	**Value**	**Data sources**
Name of windfarm site			**NV East**	
	53	degrees	52.20	
Number of turbines			200	
Width of windfarm		km	22.3	
Tidal offset		m	0.8	

Turbine data	**Units**	**Value**	**Data sources**											
Turbine model		**9MW turbine**												
No of blades		3												
Rotation speed	rpm	11.26												
Rotor radius	m	85												
Hub height	m	107	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Monthly proportion of time operational	%		90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%
Max blade width	m	7.0												
Pitch	degrees	15												

Avoidance rates used in presenting results		**Data sources (if applicable)**
	90.00%	
	95.00%	
	98.00%	
	98.90%	

COLLISION RISK ASSESSMENT
Sheet 2 - Overall collision risk

All data input on Sheet 1:
no data entry needed on this sheet!

🟩	from Sheet 1 - input data
🟪	from Sheet 6 - available hours
🟧	from Sheet 3 - single transit collision risk
🟨	from survey data
🟦	calculated field

			Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	year average
Bird details:															
Species		Gannet													
Flight speed	m/sec	14.9													
Nocturnal activity factor (1-5)		2													
Nocturnal activity (% of daytime)		25%													
Windfarm data:															
Latitude	degrees	52.2													
Number of turbines		200													
Rotor radius	m	85													
Minimum height of rotor	m	107													
Total rotor frontal area	sq m	4539601													
Proportion of time operational	%		90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90.0%
Stage A - flight activity															
Daytime areal bird density	birds/sq km		0	0.031	0	0.031	0	0.18	0	0.137	0.276	0.123	1.168	1.142	
Proportion at rotor height	%	10.2%													
Total daylight hours per month	hrs		257	276	367	416	486	500	504	455	382	332	266	242	
Total night hours per month	hrs		487	396	377	304	258	220	240	289	338	412	454	502	
Flux factor			0	16667	0	21851	0	143140	0	103498	184344	76587	634401	601419	
Option 1 -Basic model - Stages B, C and D															**per annum**
Potential bird transits through rotors			0	1700	0	2229	0	14600	0	10557	18803	7812	64709	61345	**181754**
Collision risk for single rotor transit	(from sheet 3)	9.1%													
Collisions for entire windfarm, allowing for non-op time, assuming no avoidance	birds per month or year		**0**	**139**	**0**	**183**	**0**	**1196**	**0**	**864**	**1540**	**640**	**5299**	**5023**	**14883**
Option 2-Basic model using proportion from flight distribution			**0**	**121**	**0**	**159**	**0**	**1041**	**0**	**753**	**1340**	**557**	**4613**	**4373**	**12957**
Option 3-Extended model using flight height distribution															
Proportion at rotor height	(from sheet 4)	8.9%													
Potential bird transits through rotors	Flux integral	0.0487	0	812	0	1065	0	6975	0	5043	8983	3732	30683	8249	**51066**
Collisions assuming no avoidance	Collision integral	0.00241	0	36	0	**47**	0	**310**	0	**224**	**400**	**166**	**1447**	**1304**	3936
Average collision risk for single rotor transit		4.9%													
Stage E - applying avoidance rates															
Using which of above options?	**Option 1**	0.00%	0	139	0	183	0	1196	0	864	1540	640	5299	5023	**14883**
Collisions assuming avoidance rate	birds per month or year	90.00%	0	14	0	18	0	120	0	86	154	64	530	502	**1488**
		95.00%	0	7	0	9	0	60	0	43	77	32	265	251	**744**
		98.00%	0	3	0	4	0	24	0	17	31	13	106	100	**298**
		98.90%	0	2	0	2	0	13	0	10	17	7	58	55	**163.71**
Collisions after applying large array correction		90.00%	0	14	0	18	0	119	0	86	153	64	527	500	**1482**
		95.00%	0	7	0	9	0	60	0	43	77	32	264	251	**742**
		98.00%	0	3	0	4	0	24	0	17	31	13	106	100	**297**
		98.90%	0	2	0	2	0	13	0	10	17	7	58	55	**164**

COLLISION RISK ASSESSMENT
Sheet 1 - Input data

🟩 used in overall collision risk sheet
🟨 used in migrant collision risk sheet
🟧 used in single transit collision risk sheet or extended model
🟪 used in available hours sheet
⬜ used in large array correction sheet
🟥 not used in calculation but stated for reference

	Units	Value	Data sources
Bird data			
Species name		Gannet	
Bird length	m	0.94	
Wingspan	m	1.72	
Flight speed	m/sec	14.9	
Nocturnal activity factor (1-5)		2	
Flight type, flapping or gliding		flapping	

	Units	Value	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Bird survey data														
Daytime bird density	birds/sq km		0	0.031	0	0	0	0.031	0.062	0.08	0.053	0.28	0.649	0
Proportion at rotor height	%	10.2%												
Proportion of flights upwind	%	50.0%												

	Units	Value	Data sources
Birds on migration data			
Migration passages	birds		
Width of migration corridor	km		
Proportion at rotor height	%		
Proportion of flights upwind	%		

		Units	Value	Data sources
Windfarm data				
Name of windfarm site			NV West	
	53	degrees	52.90	
Number of turbines			200	
Width of windfarm		km	17.7	
Tidal offset		m	0.8	

	Units	Value	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Turbine data														
Turbine model		9MW turbine												
No of blades		3												
Rotation speed	rpm	11.26												
Rotor radius	m	85												
Hub height	m	107												
Monthly proportion of time operational	%		90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%
Max blade width	m	7.0												
Pitch	degrees	15												

		Data sources (if applicable)
Avoidance rates used in presenting results	90.00%	
	95.00%	
	98.00%	
	98.90%	

COLLISION RISK ASSESSMENT

Sheet 2 - Overall collision risk

All data input on Sheet 1:
no data entry needed on this sheet!

🟩	from Sheet 1 - input data
🟪	from Sheet 6 - available hours
🟧	from Sheet 3 - single transit collision risk
🟨	from survey data
🟦	calculated field

			Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	year average
Bird details:															
Species		Gannet													
Flight speed	m/sec	14.9													
Nocturnal activity factor (1-5)		2													
Nocturnal activity (% of daytime)		25%													
Windfarm data:															
Latitude	degrees	52.9													
Number of turbines		200													
Rotor radius	m	85													
Minimum height of rotor	m	107													
Total rotor frontal area	sq m	4539601													
Proportion of time operational	%		90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90.0%
Stage A - flight activity															
Daytime areal bird density	birds/sq km		0	0.031	0	0	0	0.031	0.062	0.08	0.053	0.28	0.649	0	
Proportion at rotor height	%	10.2%													
Total daylight hours per month	hrs		254	275	366	418	489	504	507	457	382	330	263	238	
Total night hours per month	hrs		490	397	378	302	255	216	237	287	338	414	457	506	
Flux factor			0	16613	0	0	0	24788	50319	60624	35428	174011	350661	0	
Option 1 -Basic model - Stages B, C and D															**per annum**
Potential bird transits through rotors			0	1695	0	0	0	2528	5133	6184	3614	17749	35767	0	**72668**
Collision risk for single rotor transit	(from sheet 3)	9.1%													
Collisions for entire windfarm, allowing for non-op time, assuming no avoidance	birds per month or year		0	**139**	0	0	0	**207**	**420**	**506**	**296**	**1453**	**2929**	0	**5950**
Option 2-Basic model using proportion from flight distribution			**0**	**121**	**0**	**0**	**0**	**180**	**366**	**441**	**258**	**1265**	**2550**	**0**	**5181**
Option 3-Extended model using flight height distribution															
Proportion at rotor height	(from sheet 4)	8.9%													
Potential bird transits through rotors	Flux integral	0.0487	0	810	0	0	0	1208	2452	2954	1726	8480	30683	8249	**51066**
Collisions assuming no avoidance	Collision integral	0.00241	0	36	0	0	0	**54**	**109**	**131**	**77**	**377**	**1447**	0	**2232**
Average collision risk for single rotor transit		4.9%													
Stage E - applying avoidance rates															
Using which of above options?	**Option 1**	0.00%	0	139	0	0	0	207	420	506	296	1453	2929	0	**5950**
Collisions assuming avoidance rate	birds per month or year	90.00%	0	14	0	0	0	21	42	51	30	145	293	0	**595**
		95.00%	0	7	0	0	0	10	21	25	15	73	146	0	**298**
		98.00%	0	3	0	0	0	4	8	10	6	29	59	0	**119**
		98.90%	0	2	0	0	0	2	5	6	3	16	32	0	**65.46**
Collisions after applying large array correction		90.00%	0	14	0	0	0	21	42	50	29	145	291	0	**592**
		95.00%	0	7	0	0	0	10	21	25	15	72	146	0	**297**
		98.00%	0	3	0	0	0	4	8	10	6	29	59	0	**119**
		98.90%	0	2	0	0	0	2	5	6	3	16	32	0	**65**

COLLISION RISK ASSESSMENT
Sheet 1 - Input data

🟩	used in overall collision risk sheet
🟨	used in migrant collision risk sheet
🟧	used in single transit collision risk sheet or extended model
🟪	used in available hours sheet
⬜	used in large array correction sheet
🟥	not used in calculation but stated for reference

Bird data	**Units**	**Value**	**Data sources**
Species name		**Kittiwake**	
Bird length	m	0.39	
Wingspan	m	1.08	
Flight speed	m/sec	13.1	
Nocturnal activity factor (1-5)		3	
Flight type, flapping or gliding		flapping	

Data sources

Bird survey data			Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Daytime bird density	birds/sq km		0.686	0.753	0.305	0.088	0.233	0.052	0.025	0	0	0.031	1.141	0.491
Proportion at rotor height	%	12.4%												
Proportion of flights upwind	%	50.0%												

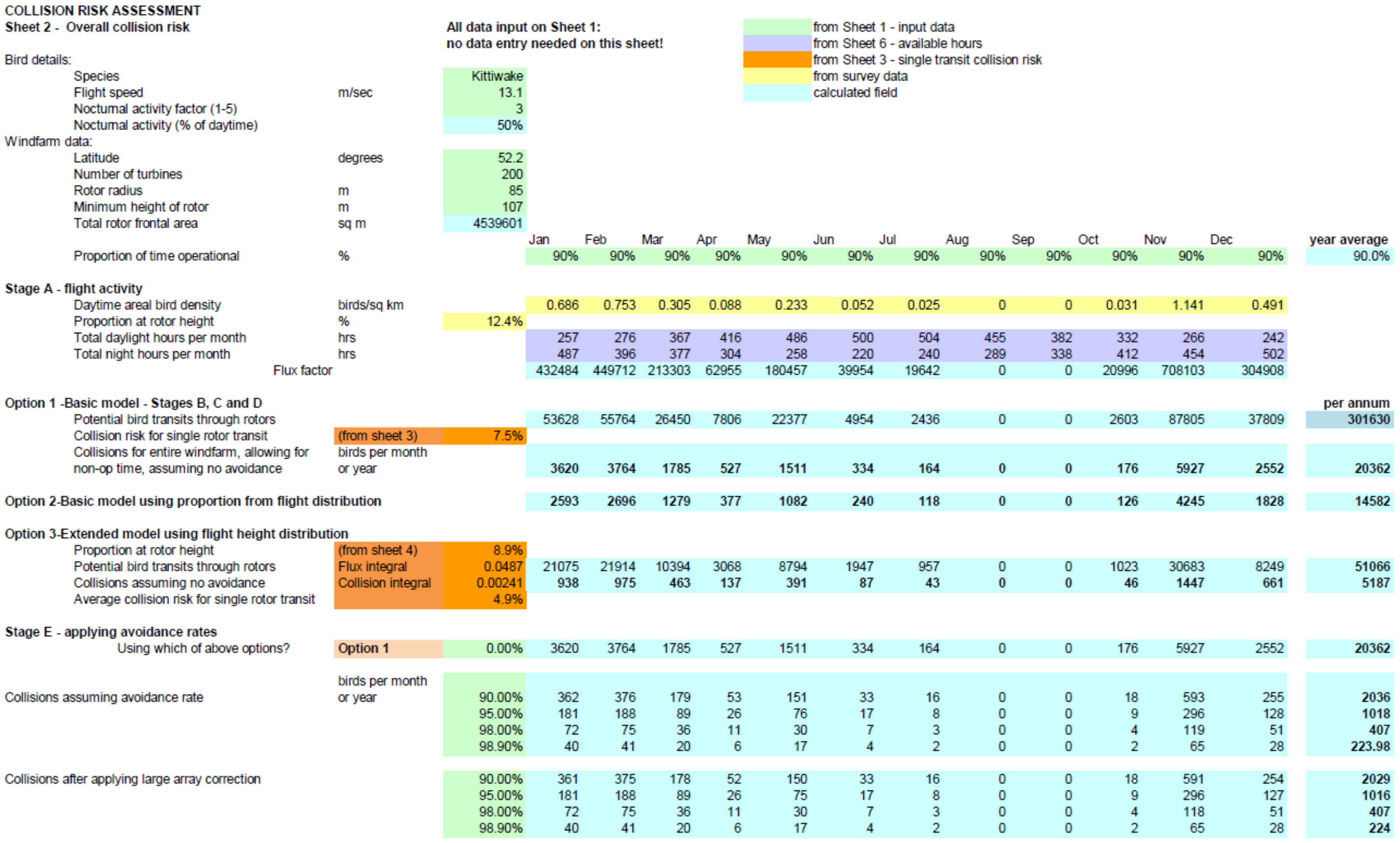
Data sources

Birds on migration data			
Migration passages	birds		
Width of migration corridor	km		
Proportion at rotor height	%		
Proportion of flights upwind	%		

Windfarm data	**Units**	**Value**	**Data sources**
Name of windfarm site		**NV East**	
53	degrees	52.20	
Number of turbines		200	
Width of windfarm	km	22.3	
Tidal offset	m	0.8	

Turbine data	**Units**	**Value**	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Turbine model		**9MW turbine**												
No of blades		3												
Rotation speed	rpm	11.26												
Rotor radius	m	85												
Hub height	m	107												
Monthly proportion of time operational	%		90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%
Max blade width	m	7.0												
Pitch	degrees	15												

Avoidance rates used in presenting results		**Data sources (if applicable)**
	90.00%	
	95.00%	
	98.00%	
	98.90%	



COLLISION RISK ASSESSMENT

Sheet 2 - Overall collision risk

All data input on Sheet 1:
no data entry needed on this sheet!

🟩	from Sheet 1 - input data
🟪	from Sheet 6 - available hours
🟧	from Sheet 3 - single transit collision risk
🟨	from survey data
🟦	calculated field

			Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	year average
Bird details:															
Species		Kittiwake													
Flight speed	m/sec	13.1													
Nocturnal activity factor (1-5)		3													
Nocturnal activity (% of daytime)		50%													
Windfarm data:															
Latitude	degrees	52.2													
Number of turbines		200													
Rotor radius	m	85													
Minimum height of rotor	m	107													
Total rotor frontal area	sq m	4539601													
Proportion of time operational	%		90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90.0%
Stage A - flight activity															
Daytime areal bird density	birds/sq km		0.686	0.753	0.305	0.088	0.233	0.052	0.025	0	0	0.031	1.141	0.491	
Proportion at rotor height	%	12.4%													
Total daylight hours per month	hrs		257	276	367	416	486	500	504	455	382	332	266	242	
Total night hours per month	hrs		487	396	377	304	258	220	240	289	338	412	454	502	
Flux factor			432484	449712	213303	62955	180457	39954	19642	0	0	20996	708103	304908	
Option 1 -Basic model - Stages B, C and D															**per annum**
Potential bird transits through rotors			53628	55764	26450	7806	22377	4954	2436	0	0	2603	87805	37809	**301630**
Collision risk for single rotor transit	(from sheet 3)	7.5%													
Collisions for entire windfarm, allowing for non-op time, assuming no avoidance	birds per month or year		**3620**	**3764**	**1785**	**527**	**1511**	**334**	**164**	**0**	**0**	**176**	**5927**	**2552**	**20362**
Option 2-Basic model using proportion from flight distribution			**2593**	**2696**	**1279**	**377**	**1082**	**240**	**118**	**0**	**0**	**126**	**4245**	**1828**	**14582**
Option 3-Extended model using flight height distribution															
Proportion at rotor height	(from sheet 4)	8.9%													
Potential bird transits through rotors	Flux integral	0.0487	21075	21914	10394	3068	8794	1947	957	0	0	1023	30683	8249	**51066**
Collisions assuming no avoidance	Collision integral	0.00241	**938**	**975**	**463**	**137**	**391**	**87**	**43**	**0**	**0**	**46**	**1447**	**661**	**5187**
Average collision risk for single rotor transit		4.9%													
Stage E - applying avoidance rates															
Using which of above options?	**Option 1**	0.00%	3620	3764	1785	527	1511	334	164	0	0	176	5927	2552	**20362**
Collisions assuming avoidance rate	birds per month or year	90.00%	362	376	179	53	151	33	16	0	0	18	593	255	**2036**
		95.00%	181	188	89	26	76	17	8	0	0	9	296	128	**1018**
		98.00%	72	75	36	11	30	7	3	0	0	4	119	51	**407**
		98.90%	40	41	20	6	17	4	2	0	0	2	65	28	**223.98**
Collisions after applying large array correction		90.00%	361	375	178	52	150	33	16	0	0	18	591	254	**2029**
		95.00%	181	188	89	26	75	17	8	0	0	9	296	127	**1016**
		98.00%	72	75	36	11	30	7	3	0	0	4	118	51	**407**
		98.90%	40	41	20	6	17	4	2	0	0	2	65	28	**224**

COLLISION RISK ASSESSMENT
Sheet 1 - Input data

🟩	used in overall collision risk sheet
🟨	used in migrant collision risk sheet
🟧	used in single transit collision risk sheet or extended model
🟪	used in available hours sheet
⬜	used in large array correction sheet
🟥	not used in calculation but stated for reference

	Units	Value	Data sources
Bird data			
Species name		**Kittiwake**	
Bird length	m	0.39	
Wingspan	m	1.08	
Flight speed	m/sec	13.1	
Nocturnal activity factor (1-5)		3	
Flight type, flapping or gliding		flapping	

Data sources

Bird survey data			Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Daytime bird density	birds/sq km		0.093	0.062	0.156	0	0.062	0.249	0	0.093	0	0.093	0.362	0
Proportion at rotor height	%	12.4%												
Proportion of flights upwind	%	50.0%												

Data sources

Birds on migration data			
Migration passages	birds		
Width of migration corridor	km		
Proportion at rotor height	%		
Proportion of flights upwind	%		

		Units	Value	Data sources
Windfarm data				
Name of windfarm site			**NV West**	
	53	degrees	52.90	
Number of turbines			200	
Width of windfarm		km	17.7	
Tidal offset		m	0.8	

	Units	Value	Data sources											
Turbine data														
Turbine model		**9MW turbine**												
No of blades		3												
Rotation speed	rpm	11.26												
Rotor radius	m	85												
Hub height	m	107	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Monthly proportion of time operational	%		90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%	90.00%
Max blade width	m	7.0												
Pitch	degrees	15												

Avoidance rates used in presenting results		**Data sources (if applicable)**
	90.00%	
	95.00%	
	98.00%	
	98.90%	

COLLISION RISK ASSESSMENT
Sheet 2 - Overall collision risk

All data input on Sheet 1:
no data entry needed on this sheet!

🟩 from Sheet 1 - input data
🟪 from Sheet 6 - available hours
🟧 from Sheet 3 - single transit collision risk
🟨 from survey data
🟦 calculated field

			Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	year average
Bird details:															
Species		Kittiwake													
Flight speed	m/sec	13.1													
Nocturnal activity factor (1-5)		3													
Nocturnal activity (% of daytime)		50%													
Windfarm data:															
Latitude	degrees	52.9													
Number of turbines		200													
Rotor radius	m	85													
Minimum height of rotor	m	107													
Total rotor frontal area	sq m	4539601													
Proportion of time operational	%		90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90%	90.0%
Stage A - flight activity															
Daytime areal bird density	birds/sq km		0.093	0.062	0.156	0	0.062	0.249	0	0.093	0	0.093	0.362	0	
Proportion at rotor height	%	12.4%													
Total daylight hours per month	hrs		254	275	366	418	489	504	507	457	382	330	263	238	
Total night hours per month	hrs		490	397	378	302	255	216	237	287	338	414	457	506	
Flux factor			58439	36965	109080	0	48141	191958	0	70353	0	62922	224054	0	
Option 1 -Basic model - Stages B, C and D															**per annum**
Potential bird transits through rotors			7246	4584	13526	0	5969	23803	0	8724	0	7802	27783	0	99436
Collision risk for single rotor transit	(from sheet 3)	7.5%													
Collisions for entire windfarm, allowing for non-op time, assuming no avoidance	birds per month or year		**489**	**309**	**913**	**0**	**403**	**1607**	**0**	**589**	**0**	**527**	**1875**	**0**	**6713**
Option 2-Basic model using proportion from flight distribution			**350**	**222**	**654**	**0**	**289**	**1151**	**0**	**422**	**0**	**377**	**1343**	**0**	**4807**
Option 3-Extended model using flight height distribution															
Proportion at rotor height	(from sheet 4)	8.9%													
Potential bird transits through rotors	Flux integral	0.0487	2848	1801	5315	0	2346	9354	0	3428	0	3066	30683	8249	51066
Collisions assuming no avoidance	Collision integral	0.00241	**127**	**80**	**237**	**0**	**104**	**416**	**0**	**153**	**0**	**136**	**1447**	**0**	**2700**
Average collision risk for single rotor transit		4.9%													
Stage E - applying avoidance rates															
Using which of above options?	**Option 1**	0.00%	489	309	913	0	403	1607	0	589	0	527	1875	0	**6713**
Collisions assuming avoidance rate	birds per month or year	90.00%	49	31	91	0	40	161	0	59	0	53	188	0	**671**
		95.00%	24	15	46	0	20	80	0	29	0	26	94	0	**336**
		98.00%	10	6	18	0	8	32	0	12	0	11	38	0	**134**
		98.90%	5	3	10	0	4	18	0	6	0	6	21	0	**73.84**
Collisions after applying large array correction		90.00%	49	31	91	0	40	160	0	59	0	52	187	0	**668**
		95.00%	24	15	46	0	20	80	0	29	0	26	94	0	**335**
		98.00%	10	6	18	0	8	32	0	12	0	11	37	0	**134**
		98.90%	5	3	10	0	4	18	0	6	0	6	21	0	**74**

Annex 3. Norfolk Vanguard collision mortality – monthly mean collision estimates

Table A3.1. Norfolk Vanguard East (1800MW). Collision mortality for the 9MW turbine calculated using Band CRM Option 2. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in seabird density, proportions at collision height, avoidance rate and nocturnal activity.

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	0 (0-0)	0 (0-0)	0.98 (0-12.11)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.95 (0-22.74)	0 (0-0)	2.93 (0-34.85)
Fulmar	0.75 (0-5.12)	0.39 (0-1.96)	0.72 (0-4.34)	0.49 (0-1.93)	2.01 (0-14.54)	1.08 (0-5.38)	0.15 (0-0.38)	1.4 (0-12)	0.8 (0-7.18)	0.31 (0-1.67)	0.79 (0-8.4)	0.92 (0-6.67)	9.81 (0-69.57)
Gannet	1.08 (0-5.07)	1.77 (0-7.53)	1.13 (0-6.18)	1.77 (0-8.45)	3.34 (0-15.67)	16.81 (0-56.97)	0.85 (0-5.42)	8.48 (0-25.08)	11.76 (0-35.81)	7.28 (0.62-24.32)	76.51 (14.17-252.69)	28.63 (0-80.84)	159.41 (14.79-524.03)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.24 (0-2.51)	0.51 (0-5.59)	0 (0-0)	0 (0-0)	0 (0-0)	0.75 (0-8.1)
Great Skua	0 (0-0)	0 (0-0)	0.35 (0-4)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.54 (0-11.41)	0.61 (0-4.66)	0 (0-0)	0 (0-0)	2.5 (0-20.07)
Kittiwake	71.19 (11.4-228.5)	32.44 (6.21-75.76)	47.32 (0-172.03)	25.43 (0-97.24)	23.58 (0-69.81)	4.64 (0-16.68)	2.34 (0-8.77)	1.35 (0-8.34)	1.37 (0-6.3)	3.76 (0-16.69)	37.36 (0-90.67)	25.9 (4.82-68.86)	276.68 (22.43-859.65)
Black-headed Gull	0.95 (0-7.68)	0 (0-0)	1.08 (0-6.46)	0 (0-0)	0.72 (0-5.29)	0 (0-0)	1 (0-6.12)	0 (0-0)	0 (0-0)	0.4 (0-3.55)	0.68 (0-6.33)	0 (0-0)	4.83 (0-35.43)
Little Gull	0 (0-0)	0.28 (0-2.6)	0 (0-0)	0 (0-0)	7.29 (0-37.25)	0 (0-0)	0 (0-0)	7.49 (0-37.4)	0.29 (0-2.69)	0 (0-0)	1.39 (0-10.11)	0 (0-0)	16.74 (0-90.05)
Common Gull	3.03 (0-13.75)	0 (0-0)	2.48 (0-15.24)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	3.13 (0-18.9)	1.38 (0-7.22)	0.73 (0-5.68)	10.75 (0-60.79)
Lesser Black-backed Gull	3.85 (0-16.19)	1.03 (0-7.29)	0.77 (0-6.63)	1.35 (0-6.93)	0 (0-0)	0 (0-0)	1.69 (0-8.91)	9.82 (0-33.69)	0 (0-0)	0.45 (0-3.73)	1.46 (0-7.27)	1.31 (0-8.86)	21.73 (0-99.5)
Herring Gull	30.42 (0-131.35)	0 (0-0)	0.91 (0-7.21)	1.44 (0-7.46)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.43 (0-13.93)	1.93 (0-12.12)	37.13 (0-172.07)
Great Black-backed Gull	72.88 (1.43-307.69)	1.31 (0-8.93)	0.83 (0-7.06)	1.82 (0-12.8)	0 (0-0)	0 (0-0)	0 (0-0)	14.15 (0-51.5)	0 (0-0)	0 (0-0)	6.45 (0-33.73)	9.77 (0-30.02)	107.21 (1.43-451.73)

Table A3.2. Norfolk Vanguard East (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 2. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in seabird density only (mean values for proportions at collision height, avoidance rate and nocturnal activity).

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	0 (0-0)	0 (0-0)	0.79 (0-6.9)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.17 (0-11.09)	0 (0-0)	2.96 (0-17.99)
Fulmar	0.75 (0-2.48)	0.36 (0-1.39)	0.63 (0-2.2)	0.58 (0-2.33)	2.3 (0-5.71)	1.05 (0-3.71)	0.16 (0-0.65)	1.6 (0.3-3.52)	0.9 (0-2.68)	0.29 (0-1.14)	0.98 (0-2.36)	0.94 (0-2.69)	10.54 (0.3-30.86)
Gannet	1.58 (0-6.82)	2.28 (0-7.74)	1.38 (0-5.67)	1.98 (0-8.07)	3.65 (0-14.55)	18.18 (0-49.9)	0.9 (0-5.49)	9.54 (0-22.11)	14.09 (0-33.97)	9.33 (1.37-26.32)	106.63 (33.74-251.37)	42.17 (0-81.49)	211.71 (35.11-513.5)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.24 (0-1.44)	0.5 (0-2.99)	0 (0-0)	0 (0-0)	0 (0-0)	0.74 (0-4.43)
Great Skua	0 (0-0)	0 (0-0)	0.35 (0-3.13)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.58 (0-6.76)	0.56 (0-2.78)	0 (0-0)	0 (0-0)	2.49 (0-12.67)
Kittiwake	105.12 (19.89-279.25)	44.52 (10.58-90.51)	60.02 (0-184.57)	29.91 (0-98.38)	26.55 (2.38-69.48)	5.05 (0-18.53)	2.62 (0-10.25)	1.5 (0-9.52)	1.71 (0-7.06)	5.12 (0-21.04)	54.19 (0-110.8)	38.78 (7.87-92.07)	375.09 (40.72-991.46)
Black-headed Gull	0.92 (0-5.43)	0 (0-0)	1.07 (0-5.08)	0 (0-0)	0.7 (0-4.31)	0 (0-0)	1.04 (0-4.29)	0 (0-0)	0 (0-0)	0.39 (0-2.3)	0.66 (0-5.6)	0 (0-0)	4.78 (0-27.01)
Little Gull	0 (0-0)	0.29 (0-1.77)	0 (0-0)	0 (0-0)	7.26 (0-24.34)	0 (0-0)	0 (0-0)	7.43 (0-22.82)	0.27 (0-1.66)	0 (0-0)	1.41 (0-8.37)	0 (0-0)	16.66 (0-58.96)
Common Gull	3.19 (0-14.52)	0 (0-0)	2.61 (0-14.65)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	3.12 (0-16.29)	1.38 (0-6.42)	0.72 (0-6.39)	11.02 (0-58.27)
Lesser Black-backed Gull	4.35 (0-15.87)	1.15 (0-6.94)	0.88 (0-7.71)	1.48 (0-6.53)	0 (0-0)	0 (0-0)	1.82 (0-8.96)	10.52 (0-28.94)	0 (0-0)	0.52 (0-4.51)	1.69 (0-7.08)	1.49 (0-9.04)	23.9 (0-95.58)
Herring Gull	34.88 (0-120.09)	0 (0-0)	1.06 (0-8.88)	1.47 (0-7.13)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.81 (0-14.27)	2.26 (0-13.69)	42.48 (0-164.06)

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Great Black-backed Gull	83.23 (2.48-269.07)	1.49 (0-9.36)	0.89 (0-8.17)	1.9 (0-11.73)	0 (0-0)	0 (0-0)	0 (0-0)	14.89 (0-48.03)	0 (0-0)	0 (0-0)	7.27 (0-33.43)	11.27 (0-30.49)	120.94 (2.48-410.28)

Table A3.3. Norfolk Vanguard East (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 2. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in avoidance rate only (mean values for seabird density and mean values for proportions at collision height, avoidance rate and nocturnal activity).

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Fulmar	0.47 (0.38-0.56)	0.21 (0.17-0.26)	0.5 (0.41-0.6)	0.2 (0.17-0.25)	1.87 (1.51-2.23)	0 (0-0)	0 (0-0)	1.49 (1.21-1.8)	0.64 (0.52-0.78)	0.19 (0.15-0.23)	0.96 (0.78-1.15)	0.74 (0.6-0.89)	7.27 (5.9-8.75)
Gannet	0 (0-0)	1.55 (1.04-2.15)	0 (0-0)	2.01 (1.35-2.81)	0 (0-0)	13.24 (8.86-18.33)	0 (0-0)	9.46 (6.38-13.25)	16.96 (11.5-23.45)	7.05 (4.7-9.78)	58.33 (39.69-80.6)	54.89 (37.29-75.84)	163.49 (110.81-226.21)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.91 (0.75-1.11)	0 (0-0)	0 (0-0)	0 (0-0)	0.91 (0.75-1.11)
Kittiwake	39.77 (26.66-55.31)	41.27 (28.01-57.14)	19.68 (13.19-27.47)	5.81 (3.92-8.08)	16.71 (11.25-23.19)	3.72 (2.54-5.13)	1.77 (1.21-2.46)	0 (0-0)	0 (0-0)	1.91 (1.29-2.65)	64.98 (44.16-89.77)	28.07 (19.13-38.54)	223.69 (151.36-309.74)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.28 (1.31-3.5)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.28 (1.31-3.5)
Common Gull	2.23 (1.29-3.45)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.23 (1.29-3.45)
Lesser Black-backed Gull	2.01 (1.31-2.88)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	8.92 (5.78-12.73)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	10.93 (7.09-15.61)
Herring Gull	6.9 (4.55-9.82)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	6.9 (4.55-9.82)
Great Black-backed Gull	12.41 (8.07-17.7)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.5 (0.98-2.17)	0 (0-0)	0 (0-0)	0 (0-0)	9.47 (6.13-13.52)	23.38 (15.18-33.39)

Table A3.4. Norfolk Vanguard East (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 2. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in proportions at collision height (Option 2) only (mean values for seabird density and mean values for proportions at collision height, avoidance rate and nocturnal activity).

Species	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Red-throated Diver	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Fulmar	0.53 (0-6.61)	0.24 (0-2.67)	0.47 (0-5.37)	0.21 (0-2.48)	2.17 (0-26.45)	0 (0-0)	0 (0-0)	1.34 (0-12.95)	0.56 (0-4.97)	0.19 (0-1.88)	1.17 (0-14.21)	0.69 (0-7.31)	7.57 (0-84.9)
Gannet	0 (0-0)	1.55 (0.67-2.76)	0 (0-0)	2.05 (0.9-3.56)	0 (0-0)	13.33 (5.93-23.54)	0 (0-0)	9.53 (4.13-16.81)	16.84 (7.63-29.28)	6.99 (3.05-12.21)	58.41 (25.37-103.7)	55.09 (24.25-97.78)	163.79 (71.93-289.64)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.01-3.84)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.01-3.84)
Kittiwake	39.82 (30.49-50.07)	41.36 (31.53-52.19)	19.6 (15.09-24.83)	5.79 (4.41-7.32)	16.71 (12.86-21.04)	3.7 (2.81-4.65)	1.78 (1.36-2.25)	0 (0-0)	0 (0-0)	1.91 (1.47-2.41)	65.13 (49.45-82.13)	28.03 (21.45-35.4)	223.83 (170.92-282.29)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.3 (0.42-5.44)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.3 (0.42-5.44)
Common Gull	2.2 (1.47-3.07)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.2 (1.47-3.07)
Lesser Black-backed Gull	1.99 (0.84-3.4)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	9.02 (3.91-15.38)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	11.01 (4.75-18.78)
Herring Gull	6.93 (4.3-9.77)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	6.93 (4.3-9.77)
Great Black-backed Gull	12.35 (8.1-16.96)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.5 (0.99-2.05)	0 (0-0)	0 (0-0)	0 (0-0)	9.52 (6.41-13.02)	23.37 (15.5-32.03)

Table A3.5. Norfolk Vanguard East (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 2. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in nocturnal activity only (mean values for seabird density and mean values for proportions at collision height and avoidance rate).

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Fulmar	0.47 (0.47-0.47)	0.21 (0.21-0.21)	0.5 (0.5-0.5)	0.2 (0.2-0.2)	1.87 (1.87-1.87)	0 (0-0)	0 (0-0)	1.48 (1.48-1.48)	0.64 (0.64-0.64)	0.19 (0.19-0.19)	0.96 (0.96-0.96)	0.74 (0.74-0.74)	7.26 (7.26-7.26)
Gannet	0 (0-0)	1.17 (1.16-1.19)	0 (0-0)	1.76 (1.72-1.85)	0 (0-0)	12.18 (11.99-12.53)	0 (0-0)	8.45 (8.26-8.78)	14.44 (14-15.27)	5.49 (5.45-5.55)	42.16 (41.67-42.77)	37.71 (37.16-38.38)	123.36 (121.41-126.32)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.92 (0.92-0.92)	0 (0-0)	0 (0-0)	0 (0-0)	0.92 (0.92-0.92)
Kittiwake	26.89 (25.79-28.09)	29.93 (28.95-31)	15.69 (14.49-17.12)	4.87 (4.59-5.19)	14.61 (14-15.35)	3.31 (3.2-3.45)	1.58 (1.52-1.65)	0 (0-0)	0 (0-0)	1.43 (1.38-1.47)	45.13 (43.43-46.97)	18.49 (17.7-19.37)	161.93 (155.05-169.66)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.28 (2.28-2.28)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.28 (2.28-2.28)
Common Gull	2.22 (2.22-2.22)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.22 (2.22-2.22)
Lesser Black-backed Gull	1.76 (1.51-2)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	8.37 (7.84-8.9)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	10.13 (9.35-10.9)
Herring Gull	6.04 (5.22-6.92)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	6.04 (5.22-6.92)
Great Black-backed Gull	10.86 (9.36-12.4)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.41 (1.32-1.5)	0 (0-0)	0 (0-0)	0 (0-0)	8.27 (7.06-9.5)	20.54 (17.74-23.4)

Table A3.6. Norfolk Vanguard East (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 2. Values are the mean and 95% confidence intervals calculated across 5,000 simulations with no uncertainty in any parameters.

Species	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Red-throated Diver	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Fulmar	0.47 (0.47-0.47)	0.21 (0.21-0.21)	0.5 (0.5-0.5)	0.2 (0.2-0.2)	1.87 (1.87-1.87)	0 (0-0)	0 (0-0)	1.48 (1.48-1.48)	0.64 (0.64-0.64)	0.19 (0.19-0.19)	0.96 (0.96-0.96)	0.74 (0.74-0.74)	7.26 (7.26-7.26)
Gannet	0 (0-0)	1.55 (1.55-1.55)	0 (0-0)	2.02 (2.02-2.02)	0 (0-0)	13.24 (13.24-13.24)	0 (0-0)	9.52 (9.52-9.52)	16.98 (16.98-16.98)	7.01 (7.01-7.01)	58.14 (58.14-58.14)	54.96 (54.96-54.96)	163.42 (163.42-163.42)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.92 (0.92-0.92)	0 (0-0)	0 (0-0)	0 (0-0)	0.92 (0.92-0.92)
Kittiwake	39.78 (39.78-39.78)	41.37 (41.37-41.37)	19.67 (19.67-19.67)	5.79 (5.79-5.79)	16.68 (16.68-16.68)	3.71 (3.71-3.71)	1.78 (1.78-1.78)	0 (0-0)	0 (0-0)	1.91 (1.91-1.91)	65.1 (65.1-65.1)	27.99 (27.99-27.99)	223.78 (223.78-223.78)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.28 (2.28-2.28)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.28 (2.28-2.28)
Common Gull	2.22 (2.22-2.22)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.22 (2.22-2.22)
Lesser Black-backed Gull	2 (2-2)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	8.9 (8.9-8.9)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	10.9 (10.9-10.9)
Herring Gull	6.92 (6.92-6.92)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	6.92 (6.92-6.92)
Great Black-backed Gull	12.4 (12.4-12.4)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.5 (1.5-1.5)	0 (0-0)	0 (0-0)	0 (0-0)	9.5 (9.5-9.5)	23.4 (23.4-23.4)

Table A3.7. Norfolk Vanguard East (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 1. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in seabird density, proportions at collision height, avoidance rate and nocturnal activity.

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	0 (0-0)	0 (0-0)	8.83 (0-68.03)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	23.27 (0-124.97)	0 (0-0)	32.1 (0-193)
Fulmar	2.03 (0-6.79)	0.94 (0-3.75)	1.69 (0-6.16)	1.53 (0-6.43)	6.13 (0-15.88)	2.78 (0-9.55)	0.44 (0-2.07)	4.26 (0.67-9.52)	2.37 (0-7.11)	0.78 (0-2.87)	2.57 (0-6.43)	2.5 (0-7.49)	28.02 (0.67-84.05)
Gannet	1.51 (0-6.53)	2.44 (0-9.33)	1.57 (0-7.82)	2.45 (0-10.99)	4.61 (0-19.42)	23.06 (0-65.83)	1.17 (0-6.68)	11.82 (0-29.55)	16.36 (0-41.91)	10.13 (1.22-29.23)	106.76 (29.89-284.43)	39.92 (0-87.65)	221.8 (31.11-599.37)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	4.17 (0-25.43)	8.95 (0-50.27)	0 (0-0)	0 (0-0)	0 (0-0)	13.12 (0-75.7)
Great Skua	0 (0-0)	0 (0-0)	2.75 (0-21.97)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	12.11 (0-51.19)	4.49 (0-21.8)	0 (0-0)	0 (0-0)	19.35 (0-94.96)
Kittiwake	133.06 (21.86-406.9)	60.52 (12.4-135.38)	88.48 (0-307.56)	47.25 (0-177)	44.22 (0-126.72)	8.72 (0-31.18)	4.38 (0-15.89)	2.53 (0-15.19)	2.57 (0-11.73)	7.02 (0-30.22)	70.09 (0-162.27)	48.48 (9.4-125.22)	517.32 (43.66-1545.26)
Black-headed Gull	1.95 (0-12.45)	0 (0-0)	2.28 (0-10.94)	0 (0-0)	1.54 (0-9.75)	0 (0-0)	2.2 (0-10.71)	0 (0-0)	0 (0-0)	0.84 (0-6.62)	1.42 (0-11.13)	0 (0-0)	10.23 (0-61.6)
Little Gull	0 (0-0)	0.15 (0-1.17)	0 (0-0)	0 (0-0)	3.84 (0-14.28)	0 (0-0)	0 (0-0)	3.94 (0-13.72)	0.16 (0-1.27)	0 (0-0)	0.74 (0-4.31)	0 (0-0)	8.83 (0-34.75)
Common Gull	4.23 (0-19.12)	0 (0-0)	3.43 (0-20.61)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	4.36 (0-24.45)	1.91 (0-9.64)	1 (0-7.64)	14.93 (0-81.46)
Lesser Black-backed Gull	4.44 (0-16.84)	1.2 (0-7.97)	0.89 (0-7.19)	1.55 (0-7.59)	0 (0-0)	0 (0-0)	1.98 (0-9.55)	11.42 (0-34.4)	0 (0-0)	0.52 (0-4.22)	1.71 (0-7.81)	1.51 (0-9.54)	25.22 (0-105.11)
Herring Gull	19.38 (0-77.65)	0 (0-0)	0.58 (0-4.44)	0.92 (0-4.57)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.56 (0-8.74)	1.23 (0-7.43)	23.67 (0-102.83)

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Great Black-backed Gull	44.89 (0.98-177.57)	0.8 (0-5.26)	0.52 (0-4.2)	1.13 (0-7.75)	0 (0-0)	0 (0-0)	0 (0-0)	8.7 (0-30.11)	0 (0-0)	0 (0-0)	3.98 (0-19.93)	6.03 (0-17.85)	66.05 (0.98-262.67)

Table A3.8. Norfolk Vanguard East (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 1. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in seabird density only (mean values for proportions at collision height, avoidance rate and nocturnal activity).

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	0 (0-0)	0 (0-0)	8.52 (0-74.74)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	23.54 (0-120.16)	0 (0-0)	32.06 (0-194.9)
Fulmar	2 (0-6.62)	0.96 (0-3.72)	1.68 (0-5.86)	1.55 (0-6.21)	6.13 (0-15.24)	2.8 (0-9.89)	0.43 (0-1.74)	4.26 (0.79-9.38)	2.39 (0-7.13)	0.78 (0-3.05)	2.61 (0-6.28)	2.51 (0-7.17)	28.1 (0.79-82.29)
Gannet	2.19 (0-9.42)	3.16 (0-10.69)	1.91 (0-7.84)	2.74 (0-11.16)	5.05 (0-20.12)	25.12 (0-68.96)	1.24 (0-7.59)	13.18 (0-30.56)	19.48 (0-46.95)	12.89 (1.89-36.37)	147.37 (46.63-347.43)	58.29 (0-112.63)	292.62 (48.52-709.72)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	4.2 (0-25.52)	8.87 (0-53.08)	0 (0-0)	0 (0-0)	0 (0-0)	13.07 (0-78.6)
Great Skua	0 (0-0)	0 (0-0)	2.69 (0-24.04)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	12.11 (0-51.88)	4.31 (0-21.35)	0 (0-0)	0 (0-0)	19.11 (0-97.27)
Kittiwake	196.84 (37.24-522.88)	83.35 (19.81-169.48)	112.38 (0-345.59)	56 (0-184.21)	49.7 (4.46-130.1)	9.45 (0-34.7)	4.91 (0-19.2)	2.8 (0-17.82)	3.2 (0-13.23)	9.58 (0-39.39)	101.46 (0-207.46)	72.6 (14.74-172.39)	702.27 (76.25-1856.45)
Black-headed Gull	1.95 (0-11.55)	0 (0-0)	2.27 (0-10.81)	0 (0-0)	1.48 (0-9.17)	0 (0-0)	2.2 (0-9.13)	0 (0-0)	0 (0-0)	0.82 (0-4.91)	1.41 (0-11.93)	0 (0-0)	10.13 (0-57.5)
Little Gull	0 (0-0)	0.15 (0-0.94)	0 (0-0)	0 (0-0)	3.87 (0-12.97)	0 (0-0)	0 (0-0)	3.96 (0-12.17)	0.15 (0-0.88)	0 (0-0)	0.75 (0-4.46)	0 (0-0)	8.88 (0-31.42)
Common Gull	4.42 (0-20.14)	0 (0-0)	3.62 (0-20.32)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	4.32 (0-22.6)	1.91 (0-8.91)	1 (0-8.86)	15.27 (0-80.83)
Lesser Black-backed Gull	5.05 (0-18.44)	1.34 (0-8.06)	1.02 (0-8.96)	1.71 (0-7.58)	0 (0-0)	0 (0-0)	2.12 (0-10.41)	12.21 (0-33.61)	0 (0-0)	0.61 (0-5.24)	1.96 (0-8.23)	1.73 (0-10.5)	27.75 (0-111.03)
Herring Gull	22.19 (0-76.4)	0 (0-0)	0.67 (0-5.65)	0.93 (0-4.53)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.79 (0-9.08)	1.44 (0-8.71)	27.02 (0-104.37)

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Great Black-backed Gull	51.32 (1.53-165.9)	0.92 (0-5.77)	0.55 (0-5.04)	1.17 (0-7.24)	0 (0-0)	0 (0-0)	0 (0-0)	9.18 (0-29.61)	0 (0-0)	0 (0-0)	4.48 (0-20.61)	6.95 (0-18.8)	74.57 (1.53-252.97)

Table A3.9. Norfolk Vanguard East (1800 MW). Collision mortality for the 9MW turbine calculated using Band CRM Option 1. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in avoidance rate only (mean values for seabird density and mean values for proportions at collision height, avoidance rate and nocturnal activity).

Species	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Red-throated Diver	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Fulmar	1.25 (1.01-1.5)	0.57 (0.46-0.69)	1.33 (1.08-1.6)	0.55 (0.44-0.66)	4.97 (4.03-5.95)	0 (0-0)	0 (0-0)	3.96 (3.23-4.8)	1.72 (1.4-2.08)	0.51 (0.41-0.62)	2.55 (2.08-3.07)	1.96 (1.59-2.37)	19.37 (15.73-23.34)
Gannet	0 (0-0)	2.14 (1.43-2.97)	0 (0-0)	2.78 (1.87-3.89)	0 (0-0)	18.31 (12.24-25.33)	0 (0-0)	13.08 (8.82-18.31)	23.44 (15.89-32.41)	9.74 (6.49-13.51)	80.62 (54.86-111.4)	75.87 (51.54-104.83)	225.98 (153.14-312.65)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	7.01 (5.72-8.5)	0 (0-0)	0 (0-0)	0 (0-0)	7.01 (5.72-8.5)
Kittiwake	74.46 (49.92-103.57)	77.28 (52.44-106.99)	36.84 (24.69-51.44)	10.87 (7.34-15.13)	31.29 (21.07-43.43)	6.97 (4.76-9.6)	3.32 (2.27-4.6)	0 (0-0)	0 (0-0)	3.57 (2.42-4.97)	121.66 (82.69-168.09)	52.56 (35.81-72.17)	418.82 (283.41-579.99)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.21 (0.7-1.87)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.21 (0.7-1.87)
Common Gull	3.09 (1.8-4.79)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	3.09 (1.8-4.79)
Lesser Black-backed Gull	2.34 (1.52-3.35)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	10.37 (6.72-14.79)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	12.71 (8.24-18.14)
Herring Gull	4.39 (2.89-6.25)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	4.39 (2.89-6.25)

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Great Black-backed Gull	7.65 (4.97-10.92)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.6-1.34)	0 (0-0)	0 (0-0)	0 (0-0)	5.84 (3.78-8.34)	14.42 (9.35-20.6)

Table A3.10. Norfolk Vanguard East (1800 MW). Collision mortality for the 9MW turbine calculated using Band CRM Option 1. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in proportions at collision height (Option 2) only (mean values for seabird density and mean values for proportions at collision height, avoidance rate and nocturnal activity).

Species	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Red-throated Diver	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Fulmar	1.25 (1.25-1.25)	0.57 (0.57-0.57)	1.33 (1.33-1.33)	0.55 (0.55-0.55)	4.98 (4.98-4.98)	0 (0-0)	0 (0-0)	3.96 (3.96-3.96)	1.72 (1.72-1.72)	0.51 (0.51-0.51)	2.55 (2.55-2.55)	1.96 (1.96-1.96)	19.38 (19.38-19.38)
Gannet	0 (0-0)	2.14 (2.14-2.14)	0 (0-0)	2.79 (2.79-2.79)	0 (0-0)	18.3 (18.3-18.3)	0 (0-0)	13.15 (13.15-13.15)	23.47 (23.47-23.47)	9.69 (9.69-9.69)	80.36 (80.36-80.36)	75.96 (75.96-75.96)	225.86 (225.86-225.86)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	7.02 (7.02-7.02)	0 (0-0)	0 (0-0)	0 (0-0)	7.02 (7.02-7.02)
Kittiwake	74.48 (74.48-74.48)	77.47 (77.47-77.47)	36.84 (36.84-36.84)	10.84 (10.84-10.84)	31.23 (31.23-31.23)	6.94 (6.94-6.94)	3.33 (3.33-3.33)	0 (0-0)	0 (0-0)	3.58 (3.58-3.58)	121.89 (121.89-121.89)	52.41 (52.41-52.41)	419.01 (419.01-419.01)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.22 (1.22-1.22)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.22 (1.22-1.22)
Common Gull	3.08 (3.08-3.08)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	3.08 (3.08-3.08)
Lesser Black-backed Gull	2.33 (2.33-2.33)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	10.34 (10.34-10.34)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	12.67 (12.67-12.67)
Herring Gull	4.4 (4.4-4.4)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	4.4 (4.4-4.4)

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Great Black-backed Gull	7.64 (7.64-7.64)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.93-0.93)	0 (0-0)	0 (0-0)	0 (0-0)	5.86 (5.86-5.86)	14.43 (14.43-14.43)

Table A3.11. Norfolk Vanguard East (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 1. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in nocturnal activity only (mean values for seabird density and mean values for proportions at collision height and avoidance rate).

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Fulmar	1.25 (1.25-1.25)	0.57 (0.57-0.57)	1.33 (1.33-1.33)	0.55 (0.55-0.55)	4.98 (4.98-4.98)	0 (0-0)	0 (0-0)	3.96 (3.96-3.96)	1.72 (1.72-1.72)	0.51 (0.51-0.51)	2.55 (2.55-2.55)	1.96 (1.96-1.96)	19.38 (19.38-19.38)
Gannet	0 (0-0)	1.62 (1.61-1.64)	0 (0-0)	2.44 (2.37-2.55)	0 (0-0)	16.83 (16.58-17.32)	0 (0-0)	11.67 (11.42-12.14)	19.96 (19.35-21.1)	7.59 (7.53-7.67)	58.27 (57.59-59.11)	52.12 (51.35-53.05)	170.5 (167.8-174.58)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	7.02 (7.02-7.02)	0 (0-0)	0 (0-0)	0 (0-0)	7.02 (7.02-7.02)
Kittiwake	50.35 (48.29-52.59)	56.04 (54.2-58.05)	29.38 (27.14-32.05)	9.11 (8.6-9.72)	27.35 (26.2-28.75)	6.2 (5.99-6.46)	2.95 (2.84-3.08)	0 (0-0)	0 (0-0)	2.67 (2.59-2.75)	84.51 (81.31-87.94)	34.62 (33.13-36.27)	303.18 (290.29-317.66)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.22 (1.22-1.22)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.22 (1.22-1.22)
Common Gull	3.08 (3.08-3.08)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	3.08 (3.08-3.08)
Lesser Black-backed Gull	2.05 (1.76-2.33)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	9.72 (9.11-10.34)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	11.77 (10.87-12.67)
Herring Gull	3.84 (3.32-4.4)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	3.84 (3.32-4.4)
Great Black-backed	6.7 (5.77-7.64)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.87 (0.81-0.93)	0 (0-0)	0 (0-0)	0 (0-0)	5.1 (4.35-5.86)	12.67 (10.93-14.43)

Species	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Gull													

Table A3.12. Norfolk Vanguard East (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 1. Values are the mean and 95% confidence intervals calculated across 5,000 simulations with no uncertainty in any parameters.

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Fulmar	1.25 (1.25-1.25)	0.57 (0.57-0.57)	1.33 (1.33-1.33)	0.55 (0.55-0.55)	4.98 (4.98-4.98)	0 (0-0)	0 (0-0)	3.96 (3.96-3.96)	1.72 (1.72-1.72)	0.51 (0.51-0.51)	2.55 (2.55-2.55)	1.96 (1.96-1.96)	19.38 (19.38-19.38)
Gannet	0 (0-0)	2.14 (2.14-2.14)	0 (0-0)	2.79 (2.79-2.79)	0 (0-0)	18.3 (18.3-18.3)	0 (0-0)	13.15 (13.15-13.15)	23.47 (23.47-23.47)	9.69 (9.69-9.69)	80.36 (80.36-80.36)	75.96 (75.96-75.96)	225.86 (225.86-225.86)
Arctic Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	7.02 (7.02-7.02)	0 (0-0)	0 (0-0)	0 (0-0)	7.02 (7.02-7.02)
Kittiwake	74.48 (74.48-74.48)	77.47 (77.47-77.47)	36.84 (36.84-36.84)	10.84 (10.84-10.84)	31.23 (31.23-31.23)	6.94 (6.94-6.94)	3.33 (3.33-3.33)	0 (0-0)	0 (0-0)	3.58 (3.58-3.58)	121.89 (121.89-121.89)	52.41 (52.41-52.41)	419.01 (419.01-419.01)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.22 (1.22-1.22)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.22 (1.22-1.22)
Common Gull	3.08 (3.08-3.08)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	3.08 (3.08-3.08)
Lesser Black-backed Gull	2.33 (2.33-2.33)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	10.34 (10.34-10.34)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	12.67 (12.67-12.67)
Herring Gull	4.4 (4.4-4.4)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	4.4 (4.4-4.4)
Great Black-backed Gull	7.64 (7.64-7.64)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.93-0.93)	0 (0-0)	0 (0-0)	0 (0-0)	5.86 (5.86-5.86)	14.43 (14.43-14.43)

Table A3.13. Norfolk Vanguard West (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 2. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in seabird density, proportions at collision height, avoidance rate and nocturnal activity.

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	3.67 (0-40.69)	0.82 (0-10.17)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.83 (0-10.07)	5.32 (0-60.93)
Fulmar	0.48 (0-3.79)	0.34 (0-2.42)	0.29 (0-2.33)	0.25 (0-2.82)	0.18 (0-1.19)	0.32 (0-1.47)	0.58 (0-2.24)	1.02 (0-5.61)	0.22 (0-1.49)	3.02 (0-12.29)	0.3 (0-2.31)	0.1 (0-0.07)	7.1 (0-38.03)
Gannet	0.53 (0-3.46)	1.2 (0-4.77)	2.46 (0-11.26)	0 (0-0)	1.07 (0-6.61)	2.12 (0-8.47)	6.24 (0-23.17)	7.52 (0-27.6)	3.09 (0-11.28)	17.66 (0-59.45)	23.18 (7.71-49.29)	0 (0-0)	65.07 (7.71-205.36)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.05 (0-6.71)	0 (0-0)	0 (0-0)	0 (0-0)	1.05 (0-6.71)
Kittiwake	4.3 (0-10.18)	3.11 (0-8.35)	9.53 (0.86-25.36)	3.63 (0-14.81)	4.89 (0-13.16)	17.63 (3.19-40.48)	6.97 (0-25.28)	5.92 (0-14.12)	3.37 (0-12.81)	5.71 (0-15.67)	16.65 (1.97-42.05)	0.62 (0-3.65)	82.33 (6.02-225.92)
Black-headed Gull	0 (0-0)	1.53 (0-8.82)	0.58 (0-4.08)	1.17 (0-7.19)	0 (0-0)	0 (0-0)	0.51 (0-3.68)	0 (0-0)	0 (0-0)	1.75 (0-7.37)	0 (0-0)	0 (0-0)	5.54 (0-31.14)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.89 (0-4.15)	0 (0-0)	1.49 (0-5.74)	0 (0-0)	2.38 (0-9.89)
Common Gull	0 (0-0)	2 (0-10.24)	2.48 (0-12.94)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.01 (0-6.25)	2.38 (0-9.01)	10.39 (0-33.72)	0 (0-0)	18.26 (0-72.16)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	0.9 (0-6.15)	0.91 (0-5.91)	0 (0-0)	7.76 (0-24.78)	9.36 (0-32.24)	13.44 (0-40.5)	3.71 (0-16.9)	6.27 (0-23.6)	0 (0-0)	0 (0-0)	42.35 (0-150.08)
Herring Gull	0 (0-0)	0.92 (0-5.59)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.67 (0-6.25)	0 (0-0)	2.59 (0-11.84)
Great Black-backed Gull	3.35 (0-11.25)	7.23 (0-19.51)	1.26 (0-7.9)	0 (0-0)	1.38 (0-8.17)	0 (0-0)	2.91 (0-13.93)	4.13 (0-17.99)	11.01 (0-31.47)	2.27 (0-11.13)	5.16 (0-17.33)	0 (0-0)	38.7 (0-138.68)

Table A3.14. Norfolk Vanguard West (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 2. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in seabird density only (mean values for proportions at collision height, avoidance rate and nocturnal activity).

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	3.98 (0-9.65)	0.79 (0-4.59)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.82 (0-4.79)	5.59 (0-19.03)
Fulmar	0.48 (0-1.49)	0.34 (0-1.03)	0.29 (0-0.78)	0.29 (0-0.97)	0.21 (0-0.61)	0.31 (0-1.21)	0.42 (0-1.65)	0.91 (0-2.83)	0.18 (0-0.57)	2.23 (0-5.98)	0.33 (0-0.96)	0.09 (0-0.56)	6.08 (0-18.64)
Gannet	0.74 (0-4.61)	1.53 (0-4.57)	2.95 (0-11.35)	0 (0-0)	1.18 (0-6.79)	2.26 (0-6.87)	6.79 (0-20.81)	8.5 (0-25.91)	3.68 (0-11.43)	22.37 (0-55.21)	31.9 (18.45-47.31)	0 (0-0)	81.9 (18.45-194.86)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.05 (0-3.74)	0 (0-0)	0 (0-0)	0 (0-0)	1.05 (0-3.74)
Kittiwake	6.35 (0-14.29)	4.35 (0-10.19)	12.04 (2.01-27.95)	4.27 (0-16.82)	5.65 (0-13.33)	19.81 (4.42-42.01)	7.81 (0-26.99)	6.8 (0-15.11)	4.11 (0-15.38)	7.69 (0-19.41)	23.66 (3.53-49.61)	0.89 (0-5.31)	103.43 (9.96-256.4)
Black-headed Gull	0 (0-0)	1.49 (0-6.14)	0.61 (0-3.61)	1.19 (0-4.92)	0 (0-0)	0 (0-0)	0.51 (0-3.17)	0 (0-0)	0 (0-0)	1.77 (0-4.68)	0 (0-0)	0 (0-0)	5.57 (0-22.52)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.89 (0-2.95)	0 (0-0)	1.51 (0-3.97)	0 (0-0)	2.4 (0-6.92)
Common Gull	0 (0-0)	2.06 (0-10.33)	2.48 (0-12.21)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.04 (0-6.24)	2.37 (0-7.09)	10.5 (0-27.9)	0 (0-0)	18.45 (0-63.77)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	0.97 (0-6.06)	0.92 (0-6.16)	0 (0-0)	8.02 (0-20.03)	9.99 (0-26.54)	14.47 (2.17-31.8)	3.99 (0-15.96)	6.91 (0-19.5)	0 (0-0)	0 (0-0)	45.27 (2.17-126.05)
Herring Gull	0 (0-0)	0.99 (0-5.89)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.93 (0-6.13)	0 (0-0)	2.92 (0-12.02)
Great Black-backed Gull	3.74 (0-12.12)	8.14 (0-18.44)	1.32 (0-8.13)	0 (0-0)	1.46 (0-9.05)	0 (0-0)	3.04 (0-12.21)	4.38 (0-17.59)	12.11 (0-27.84)	2.53 (0-10.52)	5.86 (0-19.15)	0 (0-0)	42.58 (0-135.05)

Table A3.15. Norfolk Vanguard West (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 2. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in avoidance rate only (mean values for seabird density and mean values for proportions at collision height, avoidance rate and nocturnal activity).

Species	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Red-throated Diver	3.23 (2.63-3.88)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	3.23 (2.63-3.88)
Fulmar	0.37 (0.3-0.45)	0.34 (0.28-0.41)	0.2 (0.16-0.24)	0.2 (0.16-0.24)	0.2 (0.17-0.25)	0 (0-0)	0 (0-0)	0.3 (0.25-0.37)	0.17 (0.14-0.2)	0.96 (0.79-1.16)	0.32 (0.26-0.38)	0 (0-0)	3.06 (2.51-3.7)
Gannet	0 (0-0)	1.52 (1.03-2.12)	0 (0-0)	0 (0-0)	0 (0-0)	2.29 (1.57-3.16)	4.62 (3.14-6.42)	5.55 (3.76-7.68)	3.28 (2.19-4.52)	15.99 (10.82-22.08)	32.28 (21.67-45.01)	0 (0-0)	65.53 (44.18-90.99)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.76-1.12)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.76-1.12)
Kittiwake	5.4 (3.66-7.49)	3.39 (2.31-4.72)	10.05 (6.83-13.88)	0 (0-0)	4.44 (3.02-6.16)	17.64 (11.97-24.51)	0 (0-0)	6.49 (4.38-9.03)	0 (0-0)	5.8 (3.92-8.03)	20.6 (13.96-28.76)	0 (0-0)	73.81 (50.05-102.58)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.17 (0.66-1.8)	0 (0-0)	0 (0-0)	1.17 (0.66-1.8)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.85 (0.49-1.31)	0 (0-0)	1.39 (0.81-2.15)	0 (0-0)	2.24 (1.3-3.46)
Common Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.36 (1.35-3.63)	8.58 (4.9-13.41)	0 (0-0)	10.94 (6.25-17.04)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	6.7 (4.32-9.55)	7.97 (5.16-11.39)	13.04 (8.41-18.66)	0.86 (0.55-1.22)	5.86 (3.78-8.4)	0 (0-0)	0 (0-0)	34.43 (22.22-49.22)

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Herring Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.8 (1.16-2.58)	0 (0-0)	1.8 (1.16-2.58)
Great Black-backed Gull	2.43 (1.58-3.46)	6.95 (4.45-9.86)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	11.62 (7.56-16.55)	0 (0-0)	4.2 (2.74-6.03)	0 (0-0)	25.2 (16.33-35.9)

Table A3.16. Norfolk Vanguard West (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 2. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in proportions at collision height (Option 2) only (mean values for seabird density and mean values for proportions at collision height, avoidance rate and nocturnal activity).

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	3.55 (0-41.35)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	3.55 (0-41.35)
Fulmar	0.34 (0-3.84)	0.34 (0-3.86)	0.17 (0-1.62)	0.22 (0-2.62)	0.2 (0-2.41)	0 (0-0)	0 (0-0)	0.33 (0-3.77)	0.18 (0-1.91)	1.09 (0-11.19)	0.34 (0-3.37)	0 (0-0)	3.21 (0-34.59)
Gannet	0 (0-0)	1.52 (0.68-2.69)	0 (0-0)	0 (0-0)	0 (0-0)	2.29 (1.04-4.08)	4.59 (1.94-8.12)	5.57 (2.47-9.79)	3.3 (1.49-5.7)	15.99 (7.04-28.36)	32.29 (13.89-56.97)	0 (0-0)	65.55 (28.55-115.71)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.01-3.99)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.01-3.99)
Kittiwake	5.38 (4.11-6.78)	3.4 (2.58-4.29)	10.05 (7.73-12.68)	0 (0-0)	4.45 (3.36-5.62)	17.71 (13.64-22.47)	0 (0-0)	6.49 (4.96-8.17)	0 (0-0)	5.84 (4.42-7.39)	20.67 (15.82-26.11)	0 (0-0)	73.99 (56.62-93.51)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.16 (0.19-2.8)	0 (0-0)	0 (0-0)	1.16 (0.19-2.8)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.85 (0.15-2.06)	0 (0-0)	1.41 (0.26-3.36)	0 (0-0)	2.26 (0.41-5.42)
Common Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.35 (1.53-3.23)	8.56 (5.64-11.97)	0 (0-0)	10.91 (7.17-15.2)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	6.65 (2.81-11.34)	7.98 (3.49-13.58)	12.87 (5.7-21.92)	0.86 (0.36-1.46)	5.8 (2.51-10.05)	0 (0-0)	0 (0-0)	34.16 (14.87-58.35)
Herring Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.8 (1.14-2.55)	0 (0-0)	1.8 (1.14-2.55)
Great Black-backed Gull	2.43 (1.62-3.36)	6.91 (4.55-9.48)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	11.61 (7.68-15.9)	0 (0-0)	4.23 (2.78-5.78)	0 (0-0)	25.18 (16.63-34.52)

Table A3.17. Norfolk Vanguard West (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 2. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in nocturnal activity only (mean values for seabird density and mean values for proportions at collision height and avoidance rate).

Species	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Red-throated Diver	3.23 (3.23-3.23)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	3.23 (3.23-3.23)
Fulmar	0.37 (0.37-0.37)	0.34 (0.34-0.34)	0.2 (0.2-0.2)	0.2 (0.2-0.2)	0.2 (0.2-0.2)	0 (0-0)	0 (0-0)	0.3 (0.3-0.3)	0.16 (0.16-0.16)	0.97 (0.97-0.97)	0.32 (0.32-0.32)	0 (0-0)	3.06 (3.06-3.06)
Gannet	0 (0-0)	1.15 (1.14-1.17)	0 (0-0)	0 (0-0)	0 (0-0)	2.11 (2.07-2.16)	4.22 (4.15-4.35)	4.95 (4.84-5.15)	2.79 (2.71-2.96)	12.55 (12.44-12.68)	23.42 (23.14-23.75)	0 (0-0)	51.19 (50.49-52.22)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.93-0.93)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.93-0.93)
Kittiwake	3.64 (3.49-3.8)	2.46 (2.38-2.54)	7.99 (7.38-8.71)	0 (0-0)	3.89 (3.73-4.08)	15.81 (15.27-16.49)	0 (0-0)	5.55 (5.27-5.9)	0 (0-0)	4.35 (4.22-4.48)	14.32 (13.78-14.92)	0 (0-0)	58.01 (55.52-60.92)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.17 (1.17-1.17)	0 (0-0)	0 (0-0)	1.17 (1.17-1.17)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.85 (0.85-0.85)	0 (0-0)	1.4 (1.4-1.4)	0 (0-0)	2.25 (2.25-2.25)
Common Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.36 (2.36-2.36)	8.58 (8.58-8.58)	0 (0-0)	10.94 (10.94-10.94)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	6.39 (6.09-6.68)	7.57 (7.2-7.95)	12.24 (11.49-13.04)	0.79 (0.73-0.86)	5.29 (4.72-5.85)	0 (0-0)	0 (0-0)	32.28 (30.23-34.38)
Herring Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.59 (1.38-1.8)	0 (0-0)	1.59 (1.38-1.8)
Great Black-backed Gull	2.13 (1.84-2.44)	6.19 (5.47-6.92)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	10.72 (9.82-11.6)	0 (0-0)	3.71 (3.23-4.21)	0 (0-0)	22.75 (20.36-25.17)

Table A3.18. Norfolk Vanguard West (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 2. Values are the mean and 95% confidence intervals calculated across 5,000 simulations with no uncertainty in any parameters.

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	3.23 (3.23-3.23)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	3.23 (3.23-3.23)
Fulmar	0.37 (0.37-0.37)	0.34 (0.34-0.34)	0.2 (0.2-0.2)	0.2 (0.2-0.2)	0.2 (0.2-0.2)	0 (0-0)	0 (0-0)	0.3 (0.3-0.3)	0.16 (0.16-0.16)	0.97 (0.97-0.97)	0.32 (0.32-0.32)	0 (0-0)	3.06 (3.06-3.06)
Gannet	0 (0-0)	1.52 (1.52-1.52)	0 (0-0)	0 (0-0)	0 (0-0)	2.29 (2.29-2.29)	4.62 (4.62-4.62)	5.57 (5.57-5.57)	3.29 (3.29-3.29)	16.01 (16.01-16.01)	32.28 (32.28-32.28)	0 (0-0)	65.58 (65.58-65.58)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.93-0.93)	0 (0-0)	0 (0-0)	0 (0-0)	0.93 (0.93-0.93)
Kittiwake	5.38 (5.38-5.38)	3.4 (3.4-3.4)	10.03 (10.03-10.03)	0 (0-0)	4.44 (4.44-4.44)	17.69 (17.69-17.69)	0 (0-0)	6.48 (6.48-6.48)	0 (0-0)	5.82 (5.82-5.82)	20.65 (20.65-20.65)	0 (0-0)	73.89 (73.89-73.89)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.17 (1.17-1.17)	0 (0-0)	0 (0-0)	1.17 (1.17-1.17)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0.85 (0.85-0.85)	0 (0-0)	1.4 (1.4-1.4)	0 (0-0)	2.25 (2.25-2.25)
Common Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.36 (2.36-2.36)	8.58 (8.58-8.58)	0 (0-0)	10.94 (10.94-10.94)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	6.68 (6.68-6.68)	7.95 (7.95-7.95)	13.04 (13.04-13.04)	0.86 (0.86-0.86)	5.85 (5.85-5.85)	0 (0-0)	0 (0-0)	34.38 (34.38-34.38)
Herring Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.8 (1.8-1.8)	0 (0-0)	1.8 (1.8-1.8)
Great Black-backed Gull	2.44 (2.44-2.44)	6.92 (6.92-6.92)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	11.6 (11.6-11.6)	0 (0-0)	4.21 (4.21-4.21)	0 (0-0)	25.17 (25.17-25.17)

Table A3.19. Norfolk Vanguard West (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 1. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in seabird density, proportions at collision height, avoidance rate and nocturnal activity.

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	31.92 (0-82.02)	6.49 (0-37.18)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	6.6 (0-39.27)	45.01 (0-158.47)
Fulmar	13.04 (0-41.08)	9.24 (0-28.87)	7.71 (0-24.53)	7.92 (0-25.11)	5.38 (0-18.14)	7.97 (0-33.55)	11.02 (0-42.88)	24.43 (0-80.65)	4.77 (0-16.27)	60.3 (0-166.48)	8.8 (0-26.34)	2.45 (0-14.56)	163.03 (0-518.46)
Gannet	0.75 (0-4.66)	1.68 (0-5.8)	3.46 (0-14.5)	0 (0-0)	1.54 (0-8.63)	3.01 (0-10.65)	8.83 (0-29.34)	10.66 (0-33.24)	4.36 (0-14.32)	24.9 (0-67.61)	32.98 (16.97-54.46)	0 (0-0)	92.17 (16.97-243.21)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	11.11 (0-38.67)	0 (0-0)	0 (0-0)	0 (0-0)	11.11 (0-38.67)
Kittiwake	12.47 (0-28.89)	9.03 (0-23.54)	27.51 (2.84-71.15)	10.5 (0-41.54)	14.11 (0-36.84)	51.07 (9.44-114.41)	20.13 (0-70.46)	17.1 (0-39.91)	9.73 (0-35.81)	16.5 (0-44.23)	47.98 (5.98-115.38)	1.78 (0-10.41)	237.91 (18.26-632.57)
Black-headed Gull	0 (0-0)	11.43 (0-51.58)	4.5 (0-27.67)	8.74 (0-42.33)	0 (0-0)	0 (0-0)	3.85 (0-24.04)	0 (0-0)	0 (0-0)	13 (0-43.58)	0 (0-0)	0 (0-0)	41.52 (0-189.2)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.14 (0-4.09)	0 (0-0)	1.9 (0-5.34)	0 (0-0)	3.04 (0-9.43)
Common Gull	0 (0-0)	3.41 (0-16.96)	4.21 (0-21.53)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.72 (0-10.38)	4.04 (0-14.74)	17.64 (0-52.54)	0 (0-0)	31.02 (0-116.15)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	1.32 (0-8.17)	1.32 (0-7.8)	0 (0-0)	11.16 (0-31.37)	13.78 (0-40.8)	19.61 (0-49.29)	5.46 (0-22.75)	9.05 (0-29.57)	0 (0-0)	0 (0-0)	61.7 (0-189.75)
Herring Gull	0 (0-0)	1.27 (0-7.27)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.33 (0-8.6)	0 (0-0)	3.6 (0-15.87)
Great Black-backed Gull	3.22 (0-10.22)	6.93 (0-17.36)	1.23 (0-7.26)	0 (0-0)	1.35 (0-7.92)	0 (0-0)	2.82 (0-13.26)	3.98 (0-16.7)	10.59 (0-28.8)	2.19 (0-10.37)	4.96 (0-16.5)	0 (0-0)	37.27 (0-128.39)

Table A3.20. Norfolk Vanguard West (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 1. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in seabird density only (mean values for proportions at collision height, avoidance rate and nocturnal activity).

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	32.36 (0-78.43)	6.46 (0-37.29)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	6.7 (0-38.9)	45.52 (0-154.62)
Fulmar	13.06 (0-40.32)	9.27 (0-27.73)	7.87 (0-21.18)	7.88 (0-26.06)	5.58 (0-16.59)	8.24 (0-32.56)	11.22 (0-44.42)	24.53 (0-76.32)	4.81 (0-15.46)	60.25 (0-161.44)	8.87 (0-25.85)	2.43 (0-15.06)	164.01 (0-502.99)
Gannet	1.04 (0-6.53)	2.17 (0-6.47)	4.17 (0-16.06)	0 (0-0)	1.66 (0-9.6)	3.2 (0-9.72)	9.61 (0-29.45)	12.03 (0-36.67)	5.2 (0-16.18)	31.65 (0-78.13)	45.15 (26.11-66.95)	0 (0-0)	115.88 (26.11-275.76)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	11.25 (0-40.17)	0 (0-0)	0 (0-0)	0 (0-0)	11.25 (0-40.17)
Kittiwake	18.38 (0-41.33)	12.59 (0-29.48)	34.83 (5.8-80.85)	12.36 (0-48.66)	16.35 (0-38.56)	57.31 (12.79-121.53)	22.6 (0-78.07)	19.68 (0-43.73)	11.9 (0-44.5)	22.24 (0-56.17)	68.46 (10.2-143.52)	2.56 (0-15.37)	299.26 (28.79-741.77)
Black-headed Gull	0 (0-0)	11.16 (0-45.84)	4.56 (0-26.94)	8.89 (0-36.7)	0 (0-0)	0 (0-0)	3.78 (0-23.68)	0 (0-0)	0 (0-0)	13.19 (0-34.94)	0 (0-0)	0 (0-0)	41.58 (0-168.1)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.11 (0-3.71)	0 (0-0)	1.9 (0-4.98)	0 (0-0)	3.01 (0-8.69)
Common Gull	0 (0-0)	3.51 (0-17.59)	4.23 (0-20.79)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.77 (0-10.62)	4.04 (0-12.07)	17.88 (0-47.49)	0 (0-0)	31.43 (0-108.56)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	1.42 (0-8.86)	1.35 (0-9.01)	0 (0-0)	11.72 (0-29.29)	14.61 (0-38.8)	21.15 (3.18-46.49)	5.84 (0-23.33)	10.1 (0-28.51)	0 (0-0)	0 (0-0)	66.19 (3.18-184.29)
Herring Gull	0 (0-0)	1.38 (0-8.2)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.68 (0-8.54)	0 (0-0)	4.06 (0-16.74)
Great Black-backed Gull	3.6 (0-11.66)	7.83 (0-17.74)	1.27 (0-7.82)	0 (0-0)	1.4 (0-8.7)	0 (0-0)	2.92 (0-11.75)	4.21 (0-16.92)	11.65 (0-26.78)	2.43 (0-10.12)	5.64 (0-18.42)	0 (0-0)	40.95 (0-129.91)

Table A3.21. Norfolk Vanguard West (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 1. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in avoidance rate only (mean values for seabird density and mean values for proportions at collision height, avoidance rate and nocturnal activity).

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	26.27 (21.35-31.56)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	26.27 (21.35-31.56)
Fulmar	10.08 (8.2-12.14)	9.24 (7.55-11.16)	5.29 (4.31-6.36)	5.36 (4.36-6.45)	5.48 (4.45-6.62)	0 (0-0)	0 (0-0)	8.2 (6.67-9.9)	4.46 (3.66-5.36)	26.01 (21.19-31.18)	8.63 (7.01-10.33)	0 (0-0)	82.75 (67.4-99.5)
Gannet	0 (0-0)	2.15 (1.46-2.99)	0 (0-0)	0 (0-0)	0 (0-0)	3.24 (2.22-4.47)	6.54 (4.45-9.08)	7.85 (5.32-10.86)	4.64 (3.1-6.4)	22.63 (15.32-31.25)	45.69 (30.67-63.7)	0 (0-0)	92.74 (62.54-128.75)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	10.01 (8.18-12.01)	0 (0-0)	0 (0-0)	0 (0-0)	10.01 (8.18-12.01)
Kittiwake	15.62 (10.58-21.66)	9.8 (6.68-13.65)	29.07 (19.77-40.16)	0 (0-0)	12.86 (8.75-17.82)	51.05 (34.63-70.91)	0 (0-0)	18.79 (12.66-26.11)	0 (0-0)	16.79 (11.34-23.23)	59.59 (40.38-83.2)	0 (0-0)	213.57 (144.79-296.74)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	8.72 (4.94-13.44)	0 (0-0)	0 (0-0)	8.72 (4.94-13.44)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.07 (0.61-1.65)	0 (0-0)	1.75 (1.02-2.7)	0 (0-0)	2.82 (1.63-4.35)
Common Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	4.02 (2.29-6.18)	14.62 (8.34-22.84)	0 (0-0)	18.64 (10.63-29.02)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	9.8 (6.31-13.96)	11.65 (7.55-16.65)	19.06 (12.29-27.29)	1.25 (0.8-1.78)	8.56 (5.52-12.28)	0 (0-0)	0 (0-0)	50.32 (32.47-71.96)
Herring Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.51 (1.62-3.59)	0 (0-0)	2.51 (1.62-3.59)

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Great Black-backed Gull	2.33 (1.52-3.33)	6.68 (4.28-9.48)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	11.17 (7.27-15.92)	0 (0-0)	4.04 (2.64-5.8)	0 (0-0)	24.22 (15.71-34.53)

Table A3.22. Norfolk Vanguard West (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 1. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in proportions at collision height (Option 2) only (mean values for seabird density and mean values for proportions at collision height, avoidance rate and nocturnal activity).

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	26.27 (26.27-26.27)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	26.27 (26.27-26.27)
Fulmar	10.08 (10.08-10.08)	9.24 (9.24-9.24)	5.29 (5.29-5.29)	5.37 (5.37-5.37)	5.49 (5.49-5.49)	0 (0-0)	0 (0-0)	8.18 (8.18-8.18)	4.45 (4.45-4.45)	26.04 (26.04-26.04)	8.62 (8.62-8.62)	0 (0-0)	82.76 (82.76-82.76)
Gannet	0 (0-0)	2.15 (2.15-2.15)	0 (0-0)	0 (0-0)	0 (0-0)	3.24 (3.24-3.24)	6.54 (6.54-6.54)	7.89 (7.89-7.89)	4.65 (4.65-4.65)	22.66 (22.66-22.66)	45.69 (45.69-45.69)	0 (0-0)	92.82 (92.82-92.82)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	10.04 (10.04-10.04)	0 (0-0)	0 (0-0)	0 (0-0)	10.04 (10.04-10.04)
Kittiwake	15.58 (15.58-15.58)	9.83 (9.83-9.83)	29.02 (29.02-29.02)	0 (0-0)	12.85 (12.85-12.85)	51.17 (51.17-51.17)	0 (0-0)	18.74 (18.74-18.74)	0 (0-0)	16.85 (16.85-16.85)	59.76 (59.76-59.76)	0 (0-0)	213.8 (213.8-213.8)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	8.73 (8.73-8.73)	0 (0-0)	0 (0-0)	8.73 (8.73-8.73)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.07 (1.07-1.07)	0 (0-0)	1.75 (1.75-1.75)	0 (0-0)	2.82 (2.82-2.82)
Common Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	4.01 (4.01-4.01)	14.61 (14.61-14.61)	0 (0-0)	18.62 (18.62-18.62)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	9.76 (9.76-9.76)	11.62 (11.62-11.62)	19.07 (19.07-19.07)	1.26 (1.26-1.26)	8.55 (8.55-8.55)	0 (0-0)	0 (0-0)	50.26 (50.26-50.26)
Herring Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.5 (2.5-2.5)	0 (0-0)	2.5 (2.5-2.5)

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Great Black-backed Gull	2.34 (2.34-2.34)	6.65 (6.65-6.65)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	11.16 (11.16-11.16)	0 (0-0)	4.05 (4.05-4.05)	0 (0-0)	24.2 (24.2-24.2)

Table A3.23. Norfolk Vanguard West (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 1. Values are the mean and 95% confidence intervals calculated across 5,000 simulations incorporating uncertainty in nocturnal activity only (mean values for seabird density and mean values for proportions at collision height and avoidance rate).

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Red-throated Diver	26.27 (26.27-26.27)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	26.27 (26.27-26.27)
Fulmar	10.08 (10.08-10.08)	9.24 (9.24-9.24)	5.29 (5.29-5.29)	5.37 (5.37-5.37)	5.49 (5.49-5.49)	0 (0-0)	0 (0-0)	8.18 (8.18-8.18)	4.45 (4.45-4.45)	26.04 (26.04-26.04)	8.62 (8.62-8.62)	0 (0-0)	82.76 (82.76-82.76)
Gannet	0 (0-0)	1.63 (1.62-1.65)	0 (0-0)	0 (0-0)	0 (0-0)	2.98 (2.93-3.06)	5.98 (5.88-6.16)	7 (6.85-7.28)	3.95 (3.83-4.18)	17.76 (17.6-17.94)	33.14 (32.74-33.61)	0 (0-0)	72.44 (71.45-73.88)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	10.04 (10.04-10.04)	0 (0-0)	0 (0-0)	0 (0-0)	10.04 (10.04-10.04)
Kittiwake	10.53 (10.09-10.99)	7.11 (6.88-7.36)	23.11 (21.37-25.21)	0 (0-0)	11.26 (10.78-11.81)	45.74 (44.17-47.7)	0 (0-0)	16.06 (15.26-17.06)	0 (0-0)	12.57 (12.2-12.96)	41.43 (39.86-43.17)	0 (0-0)	167.81 (160.61-176.26)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	8.73 (8.73-8.73)	0 (0-0)	0 (0-0)	8.73 (8.73-8.73)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.07 (1.07-1.07)	0 (0-0)	1.75 (1.75-1.75)	0 (0-0)	2.82 (2.82-2.82)
Common Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	4.01 (4.01-4.01)	14.61 (14.61-14.61)	0 (0-0)	18.62 (18.62-18.62)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	9.34 (8.9-9.76)	11.07 (10.52-11.62)	17.89 (16.79-19.07)	1.16 (1.06-1.26)	7.73 (6.91-8.55)	0 (0-0)	0 (0-0)	47.19 (44.18-50.26)
Herring Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.21 (1.92-2.5)	0 (0-0)	2.21 (1.92-2.5)

Species	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Great Black-backed Gull	2.05 (1.77-2.34)	5.96 (5.26-6.65)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	10.32 (9.45-11.16)	0 (0-0)	3.57 (3.11-4.05)	0 (0-0)	21.9 (19.59-24.2)

Table A3.24. Norfolk Vanguard West (1800 MW). Collision mortality for the 9 MW turbine calculated using Band CRM Option 1. Values are the mean and 95% confidence intervals calculated across 5,000 simulations with no uncertainty in any parameters.

Species	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Red-throated Diver	26.27 (26.27-26.27)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	26.27 (26.27-26.27)
Fulmar	10.08 (10.08-10.08)	9.24 (9.24-9.24)	5.29 (5.29-5.29)	5.37 (5.37-5.37)	5.49 (5.49-5.49)	0 (0-0)	0 (0-0)	8.18 (8.18-8.18)	4.45 (4.45-4.45)	26.04 (26.04-26.04)	8.62 (8.62-8.62)	0 (0-0)	82.76 (82.76-82.76)
Gannet	0 (0-0)	2.15 (2.15-2.15)	0 (0-0)	0 (0-0)	0 (0-0)	3.24 (3.24-3.24)	6.54 (6.54-6.54)	7.89 (7.89-7.89)	4.65 (4.65-4.65)	22.66 (22.66-22.66)	45.69 (45.69-45.69)	0 (0-0)	92.82 (92.82-92.82)
Great Skua	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	10.04 (10.04-10.04)	0 (0-0)	0 (0-0)	0 (0-0)	10.04 (10.04-10.04)
Kittiwake	15.58 (15.58-15.58)	9.83 (9.83-9.83)	29.02 (29.02-29.02)	0 (0-0)	12.85 (12.85-12.85)	51.17 (51.17-51.17)	0 (0-0)	18.74 (18.74-18.74)	0 (0-0)	16.85 (16.85-16.85)	59.76 (59.76-59.76)	0 (0-0)	213.8 (213.8-213.8)
Black-headed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	8.73 (8.73-8.73)	0 (0-0)	0 (0-0)	8.73 (8.73-8.73)
Little Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	1.07 (1.07-1.07)	0 (0-0)	1.75 (1.75-1.75)	0 (0-0)	2.82 (2.82-2.82)
Common Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	4.01 (4.01-4.01)	14.61 (14.61-14.61)	0 (0-0)	18.62 (18.62-18.62)
Lesser Black-backed Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	9.76 (9.76-9.76)	11.62 (11.62-11.62)	19.07 (19.07-19.07)	1.26 (1.26-1.26)	8.55 (8.55-8.55)	0 (0-0)	0 (0-0)	50.26 (50.26-50.26)
Herring Gull	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	2.5 (2.5-2.5)	0 (0-0)	2.5 (2.5-2.5)

Species	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Great Black-backed Gull	2.34 (2.34-2.34)	6.65 (6.65-6.65)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	0 (0-0)	11.16 (11.16-11.16)	0 (0-0)	4.05 (4.05-4.05)	0 (0-0)	24.2 (24.2-24.2)

Annex 4. Deterministic tables of CRM with lower and upper parameter estimates

Table A4.1. Norfolk Vanguard East gannet CRM deterministic outputs calculated using median, mean and 95% confidence intervals of density, mean and 95% confidence intervals for PCH (10.22%, 4.66%, 17.25% respectively), mean and 95% confidence intervals for avoidance rates (98.9% +/-0.2) and nocturnal activity of 25%, 0% (all year round) and 8% (breeding season) and 3% (nonbreeding season) as recommended in Furness et al. (2018).

Density	**PCH (%)**	**AR (%)**	**NAF (%)**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Median	10.22	98.9	25	0	1.55	0	2.02	0	13.24	0	9.52	16.98	7.01	58.14	54.96	163.42
Mean	10.22	98.9	25	1.61	1.92	1.5	2.05	3.64	18	0.92	9.81	14.07	9.43	106.31	42.14	211.40
Lwr95	10.22	98.9	25	0	0	0	0	0	0	0	0	0	1.37	33.74	0	35.11
Upr95	10.22	98.9	25	6.82	7.74	5.67	8.07	14.55	49.9	5.49	22.11	33.97	26.32	251.37	81.49	513.50
Median	4.66	98.9	25	0	0.71	0	0.92	0	6.04	0	4.34	7.75	3.2	26.52	25.07	74.55
Median	17.25	98.9	25	0	2.62	0	3.41	0	22.36	0	16.08	28.67	11.84	98.18	92.81	275.97
Median	10.22	99.1	25	0	1.27	0	1.65	0	10.83	0	7.79	13.89	5.74	47.57	44.97	133.71
Median	10.22	98.7	25	0	1.83	0	2.39	0	15.65	0	11.25	20.07	8.28	68.71	64.95	193.13
Median	10.22	98.9	BS 8% / NBS 3%	0	1.19	0	1.81	0	12.35	0	8.63	14.89	5.55	42.81	38.46	125.69
Median	10.22	98.9	0	0	1.14	0	1.71	0	11.96	0	8.23	13.91	5.34	40.53	35.92	118.74

Table A4.2 Norfolk Vanguard East kittiwake CRM outputs calculated using median, mean and 95% confidence intervals of density, mean and 95% confidence intervals for PCH (12.36%, 9.32%, 14.72% respectively), mean and 95% confidence intervals for avoidance rates (98.9% +/-0.2) and nocturnal activity of 50% and 25% (all year round).

Density	PCH (%)	AR (%)	NAF (%)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Median	12.36	98.9	50	39.78	41.37	19.67	5.79	16.68	3.71	1.78	0	0	1.91	65.1	27.99	223.78
Mean	12.36	98.9	50	105.36	44.65	60.22	29.98	26.73	4.98	2.6	1.59	1.73	5.1	53.88	39.28	376.10
Lwr95	12.36	98.9	50	19.89	10.58	0	0	2.38	0	0	0	0	0	0	7.87	40.72
Upr95	12.36	98.9	50	279.25	90.51	184.57	98.38	69.48	18.53	10.25	9.52	7.06	21.04	110.8	92.07	991.46
Median	9.32	98.9	50	30	31.19	14.83	4.37	12.58	2.8	1.34	0	0	1.44	49.09	21.11	168.75
Median	14.72	98.9	50	47.38	49.27	23.43	6.9	19.86	4.42	2.12	0	0	2.27	77.53	33.33	266.51
Median	12.36	99.1	50	32.55	33.85	16.09	4.74	13.65	3.04	1.46	0	0	1.56	53.26	22.9	183.10
Median	12.36	98.9	50	47.01	48.89	23.25	6.84	19.71	4.38	2.1	0	0	2.26	76.94	33.08	264.46
Median	12.36	98.9	25	30.01	32.69	16.33	5.02	14.96	3.38	1.61	0	0	1.54	49.96	20.79	176.29

Table A4.3 Norfolk Vanguard East lesser black-backed gull CRM outputs calculated using median, mean and 95% confidence intervals of density, mean and 95% confidence intervals for PCH (24.85%, 17.14%, 40.84% respectively), mean and 95% confidence intervals for avoidance rates (99.5% +/-0.1) and nocturnal activity of 50% and 25% (all year round).

Density	**PCH**	**AR**	**NAF**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Median	24.85	99.5	50	2	0	0	0	0	0	0	8.9	0	0	0	0	10.90
Mean	24.85	99.5	50	4.3	1.16	0.86	1.41	0	0	1.79	10.5	0	0.5	1.74	1.51	23.77
Lwr95	24.85	99.5	50	0	0	0	0	0	0	0	0	0	0	0	0	0.00
Upr95	24.85	99.5	50	15.87	6.94	7.71	6.53	0	0	8.96	28.94	0	4.51	7.08	9.04	95.58
Median	17.14	99.5	50	1.38	0	0	0	0	0	0	6.14	0	0	0	0	7.52
Median	40.84	99.5	50	3.29	0	0	0	0	0	0	14.63	0	0	0	0	17.92
Median	24.85	99.6	50	1.6	0	0	0	0	0	0	7.12	0	0	0	0	8.72
Median	24.85	99.4	50	2.4	0	0	0	0	0	0	10.68	0	0	0	0	13.08
Median	24.85	99.5	25	1.51	0	0	0	0	0	0	7.84	0	0	0	0	9.35

Table A4.4 Norfolk Vanguard East herring gull CRM outputs calculated using median, mean and 95% confidence intervals of density, mean and 95% confidence intervals for PCH (28.53%, 21.6%, 40.03% respectively), mean and 95% confidence intervals for avoidance rates (99.5% +/-0.1) and nocturnal activity of 50% and 25% (all year round).

Density	**PCH**	**AR**	**NAF**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Median	28.53	99.5	50	6.92	0	0	0	0	0	0	0	0	0	0	0	6.92
Mean	28.53	99.5	50	34.5	0	0.99	1.54	0	0	0	0	0	0	2.72	2.28	42.03
Lwr95	28.53	99.5	50	0	0	0	0	0	0	0	0	0	0	0	0	0.00
Upr95	28.53	99.5	50	120.09	0	8.88	7.13	0	0	0	0	0	0	14.27	13.69	164.06
Median	21.6	99.5	50	5.24	0	0	0	0	0	0	0	0	0	0	0	5.24
Median	40.03	99.5	50	9.71	0	0	0	0	0	0	0	0	0	0	0	9.71
Median	28.53	99.6	50	5.54	0	0	0	0	0	0	0	0	0	0	0	5.54
Median	28.53	99.4	50	8.3	0	0	0	0	0	0	0	0	0	0	0	8.30
Median	28.53	99.5	25	5.22	0	0	0	0	0	0	0	0	0	0	0	5.22

Table A4.5 Norfolk Vanguard East great black-backed gull CRM outputs calculated using median, mean and 95% confidence intervals of density, mean and 95% confidence intervals for PCH (29.11%, 24.68%, 41.96% respectively), mean and 95% confidence intervals for avoidance rates (99.5% +/-0.1) and nocturnal activity of 50% and 25% (all year round).

Density	**PCH (%)**	**AR (%)**	**NAF (%)**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Median	29.11	99.5	50	12.4	0	0	0	0	0	0	1.5	0	0	0	9.5	23.40
Mean	29.11	99.5	50	83.39	1.56	0.91	1.95	0	0	0	15.01	0	0	7.16	11.24	121.22
Lwr95	29.11	99.5	50	2.52	0	0	0	0	0	0	0	0	0	0	0	2.52
Upr95	29.11	99.5	50	269.07	9.36	8.17	11.73	0	0	0	48.03	0	0	33.43	30.49	410.28
Median	24.68	99.5	50	10.51	0	0	0	0	0	0	1.27	0	0	0	8.05	19.83
Median	41.96	99.5	50	17.87	0	0	0	0	0	0	2.16	0	0	0	13.69	33.72
Median	29.11	99.6	50	9.92	0	0	0	0	0	0	1.2	0	0	0	7.6	18.72
Median	29.11	99.4	50	14.88	0	0	0	0	0	0	1.8	0	0	0	11.4	28.08
Median	29.11	99.5	25	9.36	0	0	0	0	0	0	1.32	0	0	0	7.06	17.74

Table A4.6. Norfolk Vanguard West gannet CRM deterministic outputs calculated using median, mean and 95% confidence intervals of density, mean and 95% confidence intervals for PCH (10.22%, 4.66%, 17.25% respectively), mean and 95% confidence intervals for avoidance rates (98.9% +/-0.2) and nocturnal activity of 25%, 0% (all year round) and 8% (breeding season) and 3% (nonbreeding season) as recommended in Furness et al. (2018).

Density	PCH (%)	AR (%)	NAF (%)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Median	10.22	98.9	25	0	1.52	0	0	0	2.29	4.62	5.57	3.29	16.01	32.28	0	65.58
Mean	10.22	98.9	25	0.74	1.52	2.93	0	1.18	2.29	6.78	8.41	3.66	22.34	31.93	0	81.78
Lwr95	10.22	98.9	25	0	0	0	0	0	0	0	0	0	0	18.45	0	18.45
Upr95	10.22	98.9	25	4.61	4.57	11.35	0	6.79	6.87	20.81	25.91	11.43	55.21	47.31	0	194.86
Median	4.66	98.9	25	0	0.69	0	0	0	1.04	2.11	2.54	1.5	7.3	14.73	0	29.91
Median	17.25	98.9	25	0	2.57	0	0	0	3.87	7.8	9.41	5.56	27.04	54.51	0	110.76
Median	10.22	99.1	25	0	1.24	0	0	0	1.87	3.78	4.56	2.69	13.1	26.41	0	53.65
Median	10.22	98.7	25	0	1.8	0	0	0	2.71	5.46	6.58	3.89	18.92	38.15	0	77.51
Median	10.22	98.9	BS 8% / NBS 3%	0	1.12	0	0	0	2.07	4.14	4.82	2.69	12.2	22.5	0	49.54
Median	10.22	98.9	0	0	1.17	0	0	0	2.14	4.29	5.06	2.89	12.65	23.68	0	51.88

Table A4.7 Norfolk Vanguard West kittiwake CRM outputs calculated using median, mean and 95% confidence intervals of density, mean and 95% confidence intervals for PCH (12.36%, 9.32%, 14.72% respectively), mean and 95% confidence intervals for avoidance rates (98.9% +/-0.2) and nocturnal activity of 50% and 25% (all year round).

Density	**PCH (%)**	**AR (%)**	**NAF (%)**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Median	12.36	98.9	50	5.38	3.4	10.03	0	4.44	17.69	0	6.48	0	5.82	20.65	0	73.89
Mean	12.36	98.9	50	6.35	4.33	11.98	4.27	5.66	19.71	7.87	6.88	4.23	7.7	23.1	0.9	102.98
Lwr95	12.36	98.9	50	0	0	2.01	0	0	4.42	0	0	0	0	3.53	0	9.96
Upr95	12.36	98.9	50	14.29	10.19	27.95	16.82	13.33	42.01	26.99	15.11	15.38	19.41	49.61	5.31	256.40
Median	9.32	98.9	50	4.06	2.56	7.56	0	3.35	13.34	0	4.89	0	4.39	15.57	0	55.72
Median	14.72	98.9	50	6.41	4.05	11.95	0	5.29	21.07	0	7.72	0	6.93	24.59	0	88.01
Median	12.36	99.1	50	4.4	2.78	8.21	0	3.63	14.47	0	5.3	0	4.76	16.9	0	60.45
Median	12.36	98.9	50	6.36	4.02	11.85	0	5.25	20.91	0	7.66	0	6.88	24.4	0	87.33
Median	12.36	98.9	25	4.06	2.69	8.33	0	3.98	16.13	0	5.71	0	4.7	15.85	0	61.45

Table A4.8 Norfolk Vanguard West lesser black-backed gull CRM outputs calculated using median, mean and 95% confidence intervals of density, mean and 95% confidence intervals for PCH (24.85%, 17.14%, 40.84% respectively), mean and 95% confidence intervals for avoidance rates (99.5% +/-0.1) and nocturnal activity of 50% and 25% (all year round).

Density	**PCH**	**AR**	**NAF**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Median	24.85	99.5	50	0	0	0	0	0	6.68	7.95	13.04	0.86	5.85	0	0	34.38
Mean	24.85	99.5	50	0	0	0.99	0.96	0	7.99	10.02	14.35	2.91	6.89	0	0	44.11
Lwr95	24.85	99.5	50	0	0	0	0	0	0	0	2.17	0	0	0	0	2.17
Upr95	24.85	99.5	50	0	0	6.06	6.16	0	20.03	26.54	31.8	15.96	19.5	0	0	126.05
Median	17.14	99.5	50	0	0	0	0	0	4.61	5.48	8.99	0.59	4.03	0	0	23.70
Median	40.84	99.5	50	0	0	0	0	0	10.98	13.07	21.43	1.41	9.61	0	0	56.50
Median	24.85	99.6	50	0	0	0	0	0	5.34	6.36	10.43	0.69	4.68	0	0	27.50
Median	24.85	99.4	50	0	0	0	0	0	8.02	9.54	15.65	1.03	7.02	0	0	41.26
Median	24.85	99.5	25	0	0	0	0	0	6.09	7.2	11.48	0.73	4.72	0	0	30.22

Table A4.9 Norfolk Vanguard West herring gull CRM outputs calculated using median, mean and 95% confidence intervals of density, mean and 95% confidence intervals for PCH (28.53%, 21.6%, 40.03% respectively), mean and 95% confidence intervals for avoidance rates (99.5% +/-0.1) and nocturnal activity of 50% and 25% (all year round).

Density	**PCH**	**AR**	**NAF**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Median	28.53	99.5	50	0	0	0	0	0	0	0	0	0	0	1.8	0	1.80
Mean	28.53	99.5	50	0	0.99	0	0	0	0	0	0	0	0	1.91	0	2.90
Lwr95	28.53	99.5	50	0	0	0	0	0	0	0	0	0	0	0	0	0.00
Upr95	28.53	99.5	50	0	5.89	0	0	0	0	0	0	0	0	6.13	0	12.02
Median	21.6	99.5	50	0	0	0	0	0	0	0	0	0	0	1.36	0	1.36
Median	40.03	99.5	50	0	0	0	0	0	0	0	0	0	0	2.53	0	2.53
Median	28.53	99.6	50	0	0	0	0	0	0	0	0	0	0	1.44	0	1.44
Median	28.53	99.4	50	0	0	0	0	0	0	0	0	0	0	2.16	0	2.16
Median	28.53	99.5	25	0	0	0	0	0	0	0	0	0	0	1.38	0	1.38

Table A4.10 Norfolk Vanguard West great black-backed gull CRM outputs calculated using median, mean and 95% confidence intervals of density, mean and 95% confidence intervals for PCH (29.11%, 24.68%, 41.96% respectively), mean and 95% confidence intervals for avoidance rates (99.5% +/-0.1) and nocturnal activity of 50% and 25% (all year round).

Density	PCH (%)	AR (%)	NAF (%)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Median	29.11	99.5	50	2.44	6.92	0	0	0	0	0	0	11.6	0	4.21	0	25.17
Mean	29.11	99.5	50	3.79	8.04	1.34	0	1.47	0	3.05	4.41	12	2.58	6.6	0	43.28
Lwr95	29.11	99.5	50	0	0	0	0	0	0	0	0	0.74	0	0	0	0.74
Upr95	29.11	99.5	50	12.12	18.44	8.13	0	9.05	0	12.21	17.59	27.84	10.52	19.15	0	135.05
Median	24.68	99.5	50	2.07	5.87	0	0	0	0	0	0	9.83	0	3.57	0	21.34
Median	41.96	99.5	50	3.52	9.97	0	0	0	0	0	0	16.72	0	6.07	0	36.28
Median	29.11	99.6	50	1.95	5.54	0	0	0	0	0	0	9.28	0	3.37	0	20.14
Median	29.11	99.4	50	2.93	8.3	0	0	0	0	0	0	13.92	0	5.05	0	30.20
Median	29.11	99.5	25	1.84	5.47	0	0	0	0	0	0	9.82	0	3.23	0	20.36

Annex 5. Natural England Commissioned Report NECR237.

Natural England Commissioned Report NECR237

Offshore wind farms and birds: incorporating uncertainty in collision risk models: a test of Masden (2015)

First published 28 September 2017

www.gov.uk/natural-england

Foreword

Natural England commission a range of reports from external contractors to provide evidence and advice to assist us in delivering our duties. The views in this report are those of the authors and do not necessarily represent those of Natural England.

Background

Operational offshore wind farms are known to have a number of potential impacts on birds and these include mortality from collision with turbine blades and ancillary structures (moving and stationary). Offshore windfarm developers routinely use collision risk models (CRMs) to assess this potential impact on birds when undertaking environmental impact assessments. In the UK, for offshore windfarms, the most frequently used avian collision risk model is the Band model (Band 2012).

The Band (2012) model requires a number of input parameters, including information on the density of birds in the windfarm area, bird avoidance rates, flight speed, flight height and size information for the bird species involved and various turbine parameters like rotor diameter, pitch and operational time. All of these input parameters have variability and uncertainty associated with them and since the predicted collision risk from the Band model is sensitive to the input parameters, variability in the input parameters can have a significant effect on predicted collision risk.

However, consideration of this variability in the key input parameters is not routinely included when collision risk modelling is undertaken as part of the Environmental Impact Assessment (EIA) process, and uncertainty/variability around the collision predictions is rarely presented in environmental statements from offshore windfarm (OWF) developers.

For these reasons a project was undertaken to develop the Band (2012) model using a simulation approach to incorporate variability and uncertainty in the collision risk modelling process. The output of this project was the development of a stochastic version of the Band (2012) collision risk model (Masden 2015) which allows variability around input parameters to be entered in the model and used to calculate a distribution of collision risk estimates which reflects the variability in the input parameters.

Natural England, as part of its statutory advice responsibilities in relation to Nationally Significant Infrastructure Projects (NSIPs) in the offshore environment, would like developers to take account of variability and uncertainty in their assessment of potential collision impacts, and the stochastic version of the Band model developed by Masden (2015) offers a means of doing that. However, there has been limited testing of the application of this stochastic version of the Band model to datasets typically used by developers for collision risk modelling. Therefore Natural England commissioned this project to review and test the stochastic version of the model to determine the best way to parameterise the model using data available from EIAs, and to compare outputs derived from the stochastic version of the model against those generated by the Band (2012) model.

Natural England will use the results of this project to inform our advice to offshore windfarm developers and the Planning Inspectorate regarding the assessment and significance of potential collision impacts to birds as part of the Environmental Impact Assessment (EIA) and Habitats Regulations Assessment (HRA) processes.

The results of this Natural England project will also be used in a project commissioned by Marine Scotland that is developing an updated version of the stochastic Band model that builds on the work undertaken to date and will address the gaps and issues identified in the current version by industry and statutory agencies..

This report should be cited as: TRINDER, M., (2017*) Offshore wind farms and birds: incorporating uncertainty in collision risk models: a test of Masden (2015)* Natural England Commissioned Reports, Number 237. York.

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Key words – Nationally Significant Infrastructure Project (NSIP), Habitats Regulations, HRA, Environmental Impact Assessment, EIA, marine, seabird, collision risk modelling, CRM, Band model, uncertainty, stochastic modelling, offshore renewables, development planning, sustainable development, marine casework

Further information
This report can be downloaded from the Natural England Access to Evidence Catalogue: **http://publications.naturalengland.org.uk/** . For information on Natural England publications contact the Natural England Enquiry Service on 0300 060 3900 or e-mail **enquiries@naturalengland.org.uk** or **MENE@naturalengland.org.uk**

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Incorporating Uncertainty in Collision Risk Models: a test of Masden (2015)

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CONTENTS

1 INTRODUCTION

Natural England would like offshore wind farm developers to be able to present robust collision mortality estimates for birds which reflect parameter uncertainty.

Offshore windfarm developers routinely use collision risk models (CRMs) to assess the potential impacts of wind turbines on birds when undertaking environmental impact assessments. In the UK, the most frequently used avian collision risk model is the Band model (Band 2000, Band et al. 2007), which was subsequently updated to be applicable to the offshore environment for a Strategic Ornithological Support Services (SOSS) project (Band 2012).

The Band (2012) model requires a number of input parameters, including information on the density of birds in the windfarm area, bird avoidance rates, flight speed, flight height and size information for the bird species involved and various turbine parameters like rotor diameter, pitch and operational time. All of these input parameters have variability and uncertainty associated with them and since the predicted collision risk from the Band model is sensitive to the input parameters, variability in the input parameter can have a significant effect on predicted collision risk.

To address this issue, the Band (2012) update of the model includes guidance about how to express uncertainty around the model input parameters when reporting a predicted collision risk. However, this approach is relatively simplistic and is only statistically valid when the sources of variability are independent of one another (Masden 2015). Furthermore, as the approach to considering uncertainty is not an intrinsic part of the modelling process, it is not routinely followed when collision risk modelling is undertaken as part of the EIA process, and uncertainty/variability around the collision predictions is rarely presented in environmental statements from offshore windfarm (OWF) developers.

For these reasons, a stochastic version of the (deterministic) Band (2012) Collision Risk Model (CRM) for birds was developed by Masden (2015). This simulation based model (hereafter referred to as 'the Masden model') was implemented in the R programming environment and used by Masden (2015) to investigate the magnitude of variation in mortality estimates obtained using realistic levels of parameter variance and to perform a sensitivity analysis.

Natural England is interested to understand how the Masden model operates and if it can be parameterised and run using the format of data typically available in reporting for offshore wind farm assessments. The aim of the current project was to review, test and set out options for incorporating variability and uncertainty in CRM input parameters into the Masden (2015) collision risk model update in a statistically and ecologically appropriate way and to compare outputs from the Masden (2015) model with those derived using the Band (2012) model.

This has included consideration of the way in which parameters are inputted to the Masden model and an investigation of methods for quantifying the variability and/or uncertainty around the input parameters. For the purposes of this review only Band model Option 1 results have been compared.

2 ESTIMATING COLLISION MODEL INPUT PARAMETERS FROM SURVEY DATA

The following section provides an overview of data analysis methods which are appropriate for generating robust input parameters for stochastic collision modelling. The methods proposed are based on an understanding of the type of data most likely to be collected (e.g. repeat samples providing a sequence of counts). Alternative methods may also be suitable, however a key factor of relevance to the current project is that under most circumstances the survey data are very unlikely to be well suited to statistical methods based on the normal distribution.

Density of flying birds

Observations of seabirds in flight at a wind farm site are collected using a form of snapshot sampling (the data are conceptually very similar for either boat or digital aerial survey methods). Count data should be analysed using an appropriate method, such as a Generalized Linear Model (GLM) or a General Additive Model (GAM; if spatial covariates are to be included) with a Poisson error structure (ideally, the method should also allow for over-dispersion, with options to use quasi-Poisson errors). Categorical explanatory variables can be used (e.g. month, year, season or survey ID) to obtain density estimates with an appropriate temporal scale (omitting an intercept term makes the outputs simpler to interpret as a single coefficient is produced for each period specified in the model). If spatial covariates such as distance to coast and sea depth are available a GAM type of model can be used. These may also be structured to account for auto-correlation (using modelling approaches such as MRSea developed by the Centre for Research in Ecological and Environmental Modelling[1]).

Most recent wind farm assessments (e.g. Forewind 2013, SmartWind 2015) have undertaken modelling using methods similar to those described above, although in the past a more basic method of density estimation was often applied, with the total number of individuals observed during a survey divided by the total area of snapshot samples. The advantage of the modelling approach is that the results include measures of parameter uncertainty (e.g. SE and confidence intervals), which are lacking from the simple approach. These are informative in their own right, but also enable subsequent assessment to explicitly consider uncertainty.

Offshore wind farm baseline surveys to inform environmental impact assessments for birds are typically conducted each month for a period of two years. Thus, there will be two density estimates available for calculating collision risk in each month. The common currencies when discussing collisions are the estimate for each month and the annual total predicted collisions. To obtain these, the number of collisions can either be calculated for each monthly survey separately and then averaged by month across the two years, or the average monthly density of birds across the two years can be estimated as a first step from which a single monthly collision is estimated. While averaging means is straightforward, it is less simple to combine estimates which include uncertainty. The simplest solution is to avoid the need to do this by fitting a GLM (or similar) to the counts with month as an explanatory variable, but not year (see Annex 1 for an example GLM summary from analysis of snapshot count data). The resulting monthly estimates will accommodate inter-annual variation (albeit derived across only two years), and measures of variance around the estimates can be calculated. The alternative is to use a method for averaging variables which have been estimated with

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[1] https://creem2.st-andrews.ac.uk/download/mrsea-guidance/

uncertainty in order to obtain a joint mean and joint uncertainty (e.g. the delta method; see Annex 2 for an example for how to calculate the overall variance for two sample variances). Use of the first approach removes the need to consider such options. For completeness in an assessment, stochastic collision estimates could also be presented using the individual monthly density estimates (e.g. 24 values), but with the former monthly averaged values used for the actual impact assessment.

If collision modelling is being conducted deterministically (e.g. using the Band model) then an indication of the range of collision estimates can be obtained by using the upper and lower confidence interval density estimates as well as the mean density. However, this provides no indication of the probability distribution of collisions which can be derived from a stochastic collision model using randomised parameter values.

If deterministic collisions are being calculated then the method used to estimate the mean and SD density has no impact on how collisions are estimated. However, if a stochastic collision model is being used (such as Masden) it is critically important that the method used in the CRM to generate the simulated (random) density estimates shares the same statistical properties as that used to estimate the densities from the survey data. For example, if an over-dispersed Poisson model has been used for data analysis, random number generation should also use this distribution. Although it is possible to back-calculate a standard deviation (SD) from model coefficients derived with a Poisson error structure (e.g. SD = square-root(n) x (upper C.I. – lower C.I.)/3.92) this makes the assumption that the confidence intervals are symmetrical around the mean. This assumption of symmetrical confidence intervals is often violated for Poisson data, particularly at lower values.

Normal or truncated normal random numbers generated from a back-calculated SD of an asymmetrical confidence interval will therefore be biased to the right (i.e. over-estimated). To illustrate this effect, consider an example with 1,000 random numbers generated using a Poisson random number function with lambda=1 (i.e. mean=SD=1), modelled as an intercept only model (i.e. to obtain the mean) using a GLM with Poisson errors, from which a symmetrical SD is back-calculated and used to generate 1,000 truncated normal random numbers:

```
rnd.pois = rpois(n=1000, lambda = 1)
model1 = glm(rnd.pois~1, family=”poisson”)
ci = exp(confint(model1))
sd = sqrt(1000) * (((ci)[2]-(ci)[1])/3.92)
rnd.rt.norm = rtnorm(1000, mean=1, sd=sd, lower=0, upper=Inf)
mean(rnd.rt.norm) = 1.309
mean(rnd.pois) = 0.98
```

The mean of the 1,000 truncated normal random numbers was 1.309, 30% higher than the mean of the original data (1) for the underlying process. This effect will be more pronounced if the underlying distribution is over-dispersed. In this case, using this approach (truncated normal numbers estimated for a Poisson variable) to generate densities for CRM would produce a mean estimate 30% higher than it should be.

This is relevant for the current project because the Masden model generates random density values using a truncated normal distribution which uses a mean and SD (like a normal distribution) but also upper and lower limits (the lower limit in this case set to zero). Although the lower limit prevents 'impossible' (i.e. negative) values, there is still an underlying assumption of symmetry. The consequence is that the 'centre' of the distribution is shifted away from the limits (in this case zero).

Using a different probability distribution for random number generation than that which best fits and is used for data analysis is likely to result in a poor match between the resulting random draws and the original data. Further discussion on this is provided in a later section with respect to parameterising the Masden model.

Proportion at collision height

A similar data analysis approach can be used for calculating the proportion of individuals at collision height (PCH), using a GLM with binomial errors (e.g. a binary response of 'at PCH / not at PCH'). Explanatory variables can include month and year, although the temporal resolution that can be used will depend on the sample sizes available. Thus, if sufficient data on flight heights are available in all months (or surveys) then monthly (or survey) PCH can be estimated, but if sample sizes are small, seasonal or annual estimates may be more appropriate.

As for density estimates, randomised values are most appropriately estimated using the same probability distribution (e.g. binomial) to ensure reasonable correspondence between data and simulations. While the Masden model uses a normal distribution to simulate PCH, the risk of generating skewed values is lower because the mean is typically farther away from the constraints of 0 and 1 which apply to proportional data.

If site specific data are not considered suitable for estimating PCH (e.g. insufficient observations) then an alternative is to use the modelled estimates presented in Johnston et al. (2014). This is incorporated in the Masden model (Option 2) and discussed below.

3 MASDEN MODEL REVIEW

Model structure

To convert the deterministic Band model to a stochastic one it is necessary to run the model multiple times with the input parameters for each run drawn at random from appropriate probability distributions. Each iteration of the model generates a different result and summary outputs can be obtained from the multiple iterations that are run (e.g. the mean and confidence intervals). The Masden model generates stochastic mortality estimates by nesting the calculations within a loop. New random numbers are drawn at the beginning of each run through the loop and the outputs of the model are stored at the end of each iteration. The number of simulations (i.e. runs through the loop) is user defined.

Input parameters

Input parameters (e.g. mean and SD) for the Masden model are entered in pro-forma text files ('.csv'). Table 1 lists the input parameters and the file name where they are entered. The three input files listed in Table 1 (CountData.csv, BirdData.csv, TurbineData.csv) can have multiple rows; CountData and BirdData have a row for each species and TurbineData has a row for each specification of turbine.

As can be seen in Table 1, the proportion at collision height (PCH) is modelled as a single value and multiple values (e.g. for different months) cannot be entered (without modifying the script) into the Masden model.

Table 1. Input parameters required for the Masden model. For most parameters the mean is entered in the cell with parameter name and the SD is identified with a suffix ('SD'). Further details on parameter inputs are provided in Masden (2015). Note some parameters are also entered in the model code (e.g. wind speed).

Filename	Parameter	Value	Note
CountData.csv	Monthly density (labelled as Jan-Dec)	Mean & SD	Density of birds in flight in each month
BirdData.csv	AvoidanceBasic	Mean & SD	Option 1 & 2 avoidance rate
	AvoidanceExtended	Mean & SD	Option 3 & 4 avoidance rate
	Body_Length	Mean & SD	From literature
	Wingspan	Mean & SD	From literature
	Flight_Speed	Mean & SD	From literature
	Nocturnal_Activity	Mean & SD	Value in range 0-1
	Flight	Flapping / Gliding	
	Prop_CRH_Obs	Mean & SD	Single value (i.e. not monthly, etc.)
TurbineData.csv	TurbineModel	Name (e.g. output in MW)	
	Blades	Integer	No. of blades
	RotationSpeed	Mean & SD	RPM
	RotorRadius	Mean & SD	
	HubHeightAdd	Mean & SD	Distance between lower rotor tip and highest astronomical tide (HAT). (NB: added to rotor radius this equals hub height).
	BladeWidth	Mean & SD	Max. width (at c. 25% along length from hub)
	Pitch	Mean & SD	Angle of the blade from plane of rotation, degrees
	(Jan-Dec)Op	Mean	% wind availability in each month
	(Jan-Dec)OpMean	Mean & SD	% maintenance downtime in each month

Probability distributions

The Masden model makes use of two probability distributions to generate the random parameter values for each simulation: the normal distribution and the truncated normal distribution. The truncated normal distribution is used when it is necessary to generate random numbers which are constrained by lower and/or upper limits (e.g. a lower limit of 0 prevents negative values being generated). However, the truncated normal distribution is based on the standard normal distribution and therefore it is not appropriate for parameters in the CRM which are poorly represented by the normal distribution (see previous section on density estimation).

The key aspect is that there is no straightforward method for converting a Poisson distribution to the truncated normal (as required for input to the Masden model). This limits the reliability of the outputs obtained from the Masden model, since biased density estimates will result in biased collision estimates. Further consideration of this aspect is provided in a later section.

In addition to these statistical considerations, there are two instances where the Masden model in its original state (i.e. as downloaded from the Marine Scotland website) has errors in how the random number functions are used. The truncated normal distribution function used to generate seabird densities has an upper limit set at 2 (i.e. seabird densities cannot exceed 2 birds per km^2). While this may not be of concern at some sites, there may be instances when this would cause densities to be under-estimated. The second error is the use of the normal distribution for generating random proportions of birds at collision height, rather than the truncated normal with a lower limit of zero. This error means it is possible to obtain negative values, which will in turn result in negative collision estimates (since collisions are calculated as the product of this and other variables). Guidance on how to correct these errors is provided in a later section.

Turbine parameters

Turbine hub height is modelled as a random addition to the rotor radius, measured from Highest Astronomical Tide (HAT). This is simulated as a normal random number. Surveys are likely to have been conducted over a range of tidal states, so the proportion of birds at collision height would be expected to approximate to Mean Sea Level (MSL; this will depend on the extent to which height observations are pooled, although even across a single survey the span of heights may cover several hours). Thus, to accommodate the difference between HAT and MSL the Masden model includes an offset term in the script (i.e. this is not specified in the input tables but is embedded in the model code) which has a pre-set value of 2.5m. The end-user needs to modify this for their wind farm location.

The rationale for modelling hub height and the other turbine dimensions as random variables is that this captures the uncertainty about turbine model selection which may be present at the assessment stage of wind farm development (note this does not simulate tidal variation as this follows a ‘u’ shaped distribution, not a normal distribution). However, while the final turbine design may not be determined when the collision analysis is undertaken, there will be one or more candidate models. Collision modelling, as with all other aspects of the assessment, proceeds on the basis of the ‘worst case scenario’ for any given feature, following the Rochdale Envelope approach. In the case of collision modelling this requires that each candidate turbine is used in the model in order to establish which produces the highest (and hence most precautionary) collision estimates.

It is therefore unnecessary to model these fixed turbine parameters as random variables since for any given turbine they will be known with certainty (or at least have a fixed range of alternative values). Making these random is also inconsistent with the Rochdale Envelope assessment approach. Adapting the Masden model to 'fix' these parameters to be constant is straightforward, by setting the SDs for rotor radius, hub height and blade width to be zero in the TurbineData.csv file.

However, other turbine parameters in the model (RPM and blade pitch) vary in relation to wind speed and it is therefore appropriate to model these as random variables. In its unmodified form the Masden script derives values for RPM and blade pitch from a table which relates these to wind speed (e.g. '*windpower_6.csv*' and '*windpower_8.csv*' are included with the model code for 6MW and 8MW turbines respectively). This table is automatically read into the R workspace during model execution. Values for wind speed (mean and SD) are entered directly into the model script (i.e. these are not included in the tables of input data), from which normal random variables are generated. During each simulation the value for random wind speed is used to obtain the corresponding RPM and blade pitch for use in that simulation. Note that the wind speed is specified as an annual value, not monthly.

Modelling RPM and blade pitch as related functions of wind speed is a sensible approach. However, the values for this relationship have not been derived from any specific turbine model but are instead generic estimates based on expert opinion (during the current project an approach was made to turbine manufacturers to ask if this relationship could be supplied, but these requests were declined on commercial grounds). Thus, it is impossible to be certain if the tables in Masden are suitable for CRM.

In acknowledgement of this, it is stated in Masden (2015) that if mean and standard deviations for RPM and blade pitch are entered in '*TurbineData.csv*' these will be used instead of the *windpower* relationships. However, review of the model code and testing this aspect found that there is no mechanism to enable this switch, and in fact the model always defaults to use the tabulated relationship in the *windpower_6.csv* and *windpower_8.csv* files??, irrespective of RPM and blade pitch values being entered *in TurbineData.csv*.

Flight height distributions

The Masden model generates outputs using Options 1, 2 and 3 of the Band model. For the current comparisons the focus was on Option 1 (site specific flight heights). For Option 1 the Masden model uses the mean and standard deviation of the proportion of birds at collision height (*Prop_CRH_Obs*) in the *BirdData.csv* file to simulate from a normal distribution, which in most cases will provide a reasonable approximation to the underlying proportion data (although see note above about the potential for negative values). For option 2 the overlap between rotor height and bird height (i.e. PCH) is calculated from a pre-defined sample of bird flight heights using data stored in species-specific files (e.g. *Black_legged_kittiwake_ht.csv*). In Masden (2015) it is stated that these were generated by the BTO from the modelling in Johnston et al. (2014). Each species file contains 200 bootstrap samples (200 columns) of the proportion of birds in 1m height intervals between 0 and 300m (300 rows). During each simulation one column is selected at random from the table and the proportion at collision height calculated as the overlap with rotor heights. This approach is considered robust and appropriate and will not result in the generation of negative PCH values.

4 MODEL COMPARISONS

As noted above, the unedited Masden model always uses the *windpower.csv* relationships (wind speed : RPM & blade pitch) even when these parameters are entered in the *TurbineData.csv* file. For the purposes of comparing the Masden model outputs against the Band model (i.e. to run the Masden model as a deterministic model) it was therefore necessary to provide an alternative *windpower.csv* file. This contained constant RPM and blade pitch values (i.e. these had the same value at all wind speeds) to ensure these parameters could not vary.

A second related modification was required to permit comparison of stochastic outputs from the Masden model with Band model outputs derived from upper and lower parameter values (e.g. as presented in SmartWind 2015). This required editing of one of the model scripts (‘*sampleturbineparams.txt*’), to allow the alternative sampling method to be used (i.e. use of the mean and SD for rotor speed and blade pitch values in the turbine data sheet to generate normal random variables, rather than the relationship in *windpower.csv*). This was necessary to ensure that RPM and blade pitch varied in a predictable manner around their means, rather than the non-linear relationships specified in *windpower.csv*.

It is worth noting that modelling RPM and blade pitch as independent variables in this manner is expected to inflate the variance of collision model outputs because these variables are actually related to one another (as noted by Masden, and hence the tabulated approach). However, in the absence of manufacturer data this covariance cannot be estimated and it is therefore necessary to model these as independent variables. For interest, outputs using the wind speed version are also presented for comparison, using the *windpower_6.csv* provided with the Masden script.

Deterministic comparison - Masden Model outputs compared to Band Model

The generic bird parameters and turbine parameters in Tables 2 and 5 were made up for the purposes of this comparison. The bird densities (Tables 3 and 6) were estimated from a snapshot boat survey dataset, modelled using a GLM with quasi-Poisson errors (see Appendix 1 for model details). The mean densities for use with the Masden model were the monthly coefficients from the model, while the SDs were calculated from the model confidence intervals (using sqrt(n) x (upper c.i. – lower c.i.)/3.92; where n was the number of snapshots). As discussed above, this makes the assumption that the confidence intervals were symmetrical around the mean, which is unlikely to be the case. However, this method was used here to illustrate the potential influence of this assumption on the outputs obtained.

The input parameter values used are provided in tables 2 and 3. The results obtained from each models are provided in Table 4.

Table 2. Generic bird parameters and wind farm parameters used in the Masden and Band models for deterministic comparison.

Category	**Parameter**	**Masden**		**Band**
		Mean	**SD**	
Bird (generic)	Body length	0.39	0	0.39
	Wing span	1.08	0	1.08

Category	Parameter	Masden		Band
		Mean	SD	
	Flight speed	13.1	0	13.1
	Nocturnal activity	50	0	3
	Flight type	Flapping	NA	Flapping
	Avoidance rate	98.9	0	98.9
	PCH	0.20	0	0.20
Wind farm	Latitude	55.80	NA	55.80
	Wind farm capacity	600	NA	NA
	Turbine capacity	6	NA	NA
	No. of turbines	Calculated from previous 2 values	NA	100
	Rotor radius	80.00	0	80.00
	No. of blades	3.00	NA	3.00
	RPM	11.00	0	11.00
	Blade pitch	15.00	0	15.00
	Max. blade width	5.50	0	5.50
	Hub height	NA	NA	106.5
	Hub height addition	26.50	0	NA

Table 3. Monthly bird density and wind farm operational parameters for deterministic comparison. Note that the Operation values for the Band model are Operation minus OperationMean for the Masden model (e.g. for January 96.28 - 6.3 = 89.98)

Model	Parameter	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Masden	Density	0.13	0.31	1.03	0.86	0.77	1.274	0.57	0.11	0.18	0.87	0.48	0.09
	Operation	96.28	96.53	95.83	92.78	90.86	92.22	89.11	89.92	93.71	96.14	97.14	96.41
	OperationMean	6.3	6.3	6.3	6.3	6.3	6.3	6.3	6.3	6.3	6.3	6.3	6.3
	operationSD	0	0	0	0	0	0	0	0	0	0	0	0
Band	Density	0.13	0.31	1.03	0.86	0.77	1.274	0.57	0.11	0.18	0.87	0.48	0.09
	Operation	89.98	90.23	89.53	86.48	84.56	85.92	82.81	83.62	87.41	89.84	90.84	90.11
	OperationMean	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	operationSD	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Table 4. Deterministic collision modelling results obtained from the Masden model (with all variance =0) and Band model.

Model	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Masden	4.9	10.9	42.9	35.5	33.9	56.6	25.1	4.6	7.2	34.9	17.7	3.3	277.5
Band	4.9	10.9	42.9	35.5	33.9	56.6	25.1	4.6	7.2	34.9	17.7	3.3	277.5

With all parameter variances set to zero and RPM and blade pitch fixed (i.e. not taken from the *windpower.csv* table) the Masden model produces identical results to the Band Model. This is the expected result, since the Masden model was derived from the Band model (however as noted above this could only be confirmed following code modifications to allow all parameters to be fixed).

Stochastic comparison - Masden Model outputs compared to Band Model

Masden Model in original format

The following simulations were conducted without making any adjustments to the Masden script. The input parameters used are provided in Tables 5 and 6. Note that rotor RPM and blade pitch used in the Band model were derived from the calculations using wind speed in the Masden model. In order to obtain the same mean values for use in the Band model it was necessary to run the Masden model first and then extract the mean RPM and blade pitch from the outputs.

A mean wind speed of 16ms^{-1} (and SD of 3.2) was entered in the Masden code as this corresponded to a blade angle (in the original *windpower.csv* table) of 15 degrees and an RPM of 10.2, which were considered to be similar to typical values used in collision modelling. Following completion of the Masden simulations the actual mean RPM and mean blade pitch generated during simulations were 9.87 and 13.3 respectively, and these were used in the Band model.

Table 5. Generic bird parameters and wind farm parameters used in the Masden and Band models for stochastic comparison.

Category	Parameter	Masden		Band
		Mean	SD	
Bird (generic)	Body length	0.39	0.005	0.39
	Wing span	1.08	0.04	1.08
	Flight speed	13.1	1.5	13.1
	Nocturnal activity	50	0.0045	3
	Flight type	Flapping	NA	Flapping
	Avoidance rate	98.9	0.001	98.9
	PCH	0.20	0.033	0.20
Wind farm	Wind speed	16	3.2	NA
	Latitude	55.80	NA	55.80
	Wind farm capacity	600	NA	NA
	Turbine capacity	6	NA	NA
	No. of turbines	Calculated from previous 2 values	NA	100
	Rotor radius	80.00	0	80.00
	No. of blades	3.00	NA	3.00
	RPM	NA	NA	9.87
	Blade pitch	NA	NA	13.3
	Max. blade width	5.50	0	5.50
	Hub height	NA	NA	106.5
	Hub height addition	26.50	2	NA

Table 6. Monthly bird density and wind farm operational parameters for stochastic comparison. Note that the Operation values for the Band model are Operation minus OperationMean for the Masden model (e.g. for January 96.28 - 6.3 = 89.98)

Model	Parameter	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Masden	Density (mean)	0.13	0.31	1.03	0.86	0.77	1.274	0.57	0.11	0.18	0.87	0.48	0.09
	Density (SD)	0.10	0.15	0.28	0.25	0.23	0.31	0.21	0.09	0.12	0.24	0.16	0.08
	Operation	96.28	96.53	95.83	92.78	90.86	92.22	89.11	89.92	93.71	96.14	97.14	96.41
	OperationMean	6.3	6.3	6.3	6.3	6.3	6.3	6.3	6.3	6.3	6.3	6.3	6.3
	operationSD	2	2	2	2	2	2	2	2	2	2	2	2
Band	Density	0.13	0.31	1.03	0.86	0.77	1.274	0.57	0.11	0.18	0.87	0.48	0.09
	Operation	89.98	90.23	89.53	86.48	84.56	85.92	82.81	83.62	87.41	89.84	90.84	90.11
	OperationMean	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	operationSD	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

The results obtained from the original Masden model and the Band model are provided in Table 7.

Table 7. Stochastic collision modelling results obtained from the unmodified Masden model (with input variances as defined in tables 4 and 5) and Band model.

Model		Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Masden	Mean	5.6	10.6	40.9	34.0	32.9	54.2	24.5	5.2	7.5	34.0	16.9	3.9	270.2
	SD	3.4	5.2	14.6	13.0	11.9	17.0	10.6	3.2	4.5	11.9	6.8	2.5	
	CV	61.9	48.9	35.6	38.1	36.2	31.3	43.0	62.6	60.5	34.9	40.2	64.5	
	Median	5.0	10.2	39.3	32.7	32.0	53.0	23.6	4.9	7.0	32.9	16.4	3.6	260.6
	IQR	4.8	7.0	18.9	16.6	15.9	23.3	14.6	4.4	6.0	15.8	8.9	3.4	
Band		4.6	10.2	40.1	33.2	31.7	52.9	23.5	4.3	6.8	32.7	16.5	3.1	259.6
Band as percentage of Masden	Mean	82.7	96.0	98.1	97.7	96.3	97.7	95.8	82.1	90.6	96.2	98.0	77.6	96.1
	Median	91.9	100.0	102.1	101.4	99.0	99.8	99.8	87.7	96.6	99.4	100.8	85.7	99.6

Using the parameters detailed in Tables 5 and 6 the unmodified Masden model produced slightly higher mean collision estimates (c. 4% higher), although the median outputs were very similar (<0.5% higher).

Masden Model modified to correct misspecifications

For the following comparison the Masden code was edited to remove the upper limit on bird density and to allow rotor RPM and blade pitch to be entered as independent variables. The input parameters were the same as those used for the unmodified Masden model (Tables 5 and 6).

Table 8. Stochastic collision modelling results obtained from the modified Masden model (with variances as defined in tables 4 and 5) and Band model.

Model		**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Masden	Mean	5.4	10.9	40.2	34.4	32.3	54.8	24.5	5.1	7.6	33.7	17.2	3.9	270.0
	SD	3.3	5.3	14.3	12.9	11.9	18.7	10.6	3.2	4.5	12.1	6.9	2.6	
	CV	61.1	48.5	35.5	37.5	36.8	34.1	43.2	62.5	59.9	35.9	40.1	66.4	
	Median	5.0	10.5	38.2	33.2	31.3	52.4	23.2	4.7	7.0	32.9	16.7	3.5	258.6
	IQR	4.4	7.0	18.5	17.3	16.0	24.6	14.1	4.3	6.2	15.3	9.0	3.6	
Band		4.6	10.2	40.1	33.2	31.7	52.9	23.5	4.3	6.8	32.7	16.5	3.1	259.6
Band as percentage of Masden	Mean	85.2	94.0	99.9	96.6	98.1	96.5	96.0	83.7	89.3	96.9	96.2	77.5	96.1
	Median	93.0	97.4	105.0	99.9	101.2	100.9	101.4	91.1	96.6	99.4	99.0	87.2	100.4

A visual comparison of the results in Table 8 is provided in Figure 1. The Masden model produced mean collision estimates that were consistently higher than the Band model, by up to 23%, although the absolute differences were comparatively small with the annual total only 4% higher. The median estimates were closer to the Band outputs. In both cases the magnitude of difference in each month between Band and Masden is negatively related to the CV of seabird density. Thus, the greater the relative uncertainty on density (i.e. larger CV), the greater the difference between the Masden mean (or median) estimate and the Band output. While greater uncertainty should be reflected in less precise estimates, in this case the difference is one of reduced accuracy (not precision), due to the introduction of positive bias in the resampled densities resulting from use of the truncated normal distribution: the mean of the 1,000 resampled densities for each month were larger than the input means in 10 of the 12 months, by up to 2.3%.

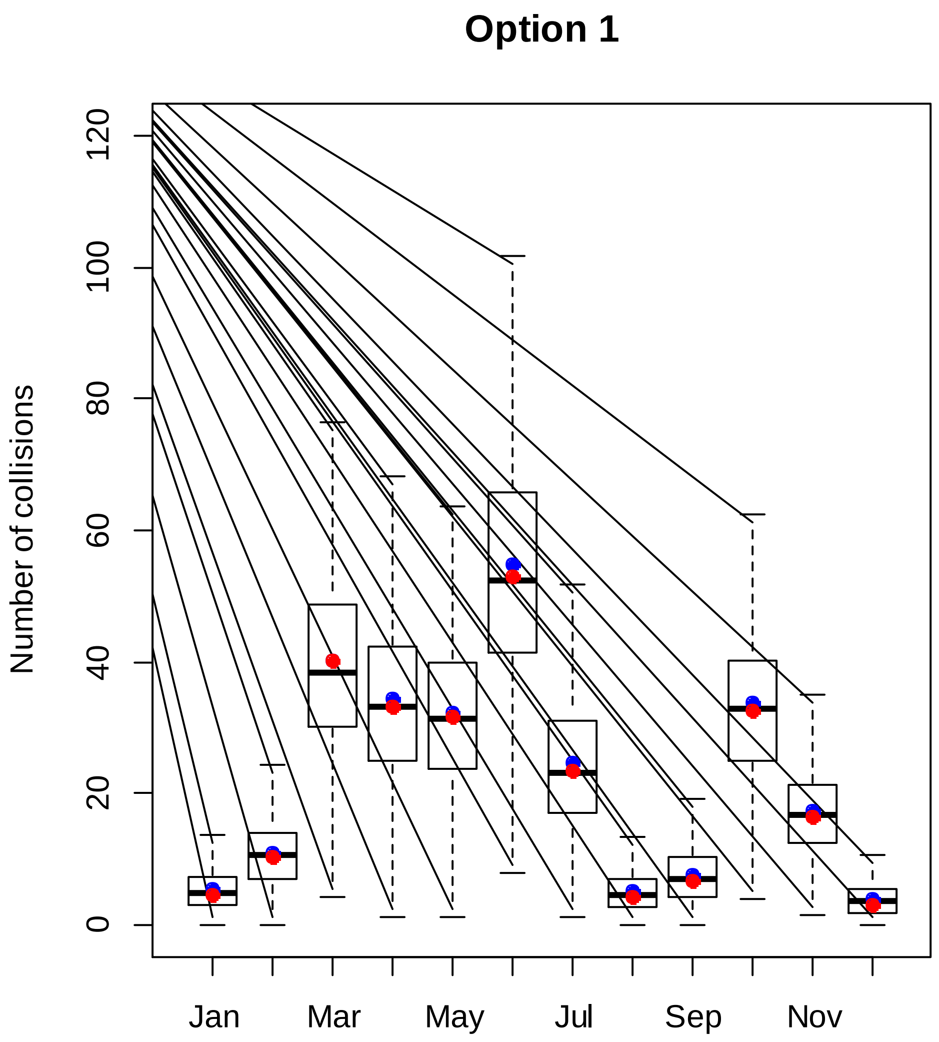

Figure 1. Box and whisker plot of Masden model outputs (in Table 8) using the parameters listed in Tables 5 and 6. The heavy horizontal lines are the medians, the boxes the 25th and 75th percentiles and the whiskers represent the range (the default setting for boxplot as used in the Masden model). The mean Masden values (blue dots) and Band model outputs (red dots) have been overlaid (note the March blue dot is hidden under the red dot).

For the dataset used in this analysis the modified Masden model produced the same results as the unmodified version. However, this would not have been the case if the data contained higher density estimates (i.e. >2/km^2) which would be truncated by the unedited Masden model by the upper limit of 2 defined for that parameter. In addition, the wind speed, RPM and blade pitch values were all standardised across the model runs (to ensure comparisons were based on the same data). However, ensuring the unedited Masden model and the Band model had the same values for RPM and blade pitch can only be achieved through a process of trial and error or by modifying the wind speed table (e.g. setting all RPM and blade pitches to the same value, although this removes the stochastic aspect for these parameters).

An alternative option to present uncertainty in collision predictions without using a stochastic model such as Masden is to calculate Band outputs using the upper and lower values for selected input parameters (e.g. SmartWind 2015). This can't provide a probability distribution of outputs, but does indicate the range over which estimates could lie. The Band model results obtained using upper and lower confidence values for seabird density (i.e. 95% confidence interval values obtained from the GLM of survey data derived using the 'confint' function) on their own and also with the avoidance rate set to upper and lower levels (i.e. +/- 0.002) are provided in Table 9 and Figure 2.

Table 9. Collision modelling results obtained from the modified Masden model (with variances as defined in Tables 5 and 6) and Band model using upper and lower 95% confidence seabird density estimates obtained from a GLM and also with recommended upper and lower avoidance rates (98.7 - 99.1%).

Model		**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Masden	Mean	5.4	10.9	40.2	34.4	32.3	54.8	24.5	5.1	7.6	33.7	17.2	3.9	270.0
	SD	3.3	5.3	14.3	12.9	11.9	18.7	10.6	3.2	4.5	12.1	6.9	2.6	
	CV (%)	61.1	48.5	35.5	37.5	36.8	34.1	43.2	62.5	59.9	35.9	40.1	66.4	
	Median	5.0	10.5	38.2	33.2	31.3	52.4	23.2	4.7	7.0	32.9	16.7	3.5	258.6
	IQR	4.4	7.0	18.5	17.3	16.0	24.6	14.1	4.3	6.2	15.3	9.0	3.6	
Band	Mean	4.6	10.2	40.1	33.2	31.7	52.9	23.5	4.3	6.8	32.7	16.5	3.1	259.6
Density range	Lwr 95%	0.7	3.6	17.8	20.5	16.8	32.9	11.0	0.8	0.9	18.1	8.4	0.3	131.71
	Uppr 95%	15.3	22.4	75.2	50.7	52.9	79.8	43.3	12.3	23.0	54.0	30.7	11.7	471.37
Density range & Avoidance rate range	Lwr 95% & 99.1% AR	0.6	2.9	14.5	16.8	13.7	26.9	9.0	0.7	0.7	14.8	6.9	0.2	107.76
	Uppr 95% & 98.7% AR	18.0	26.5	88.9	60.0	62.5	94.4	51.2	14.6	27.2	63.8	36.3	13.8	557.08

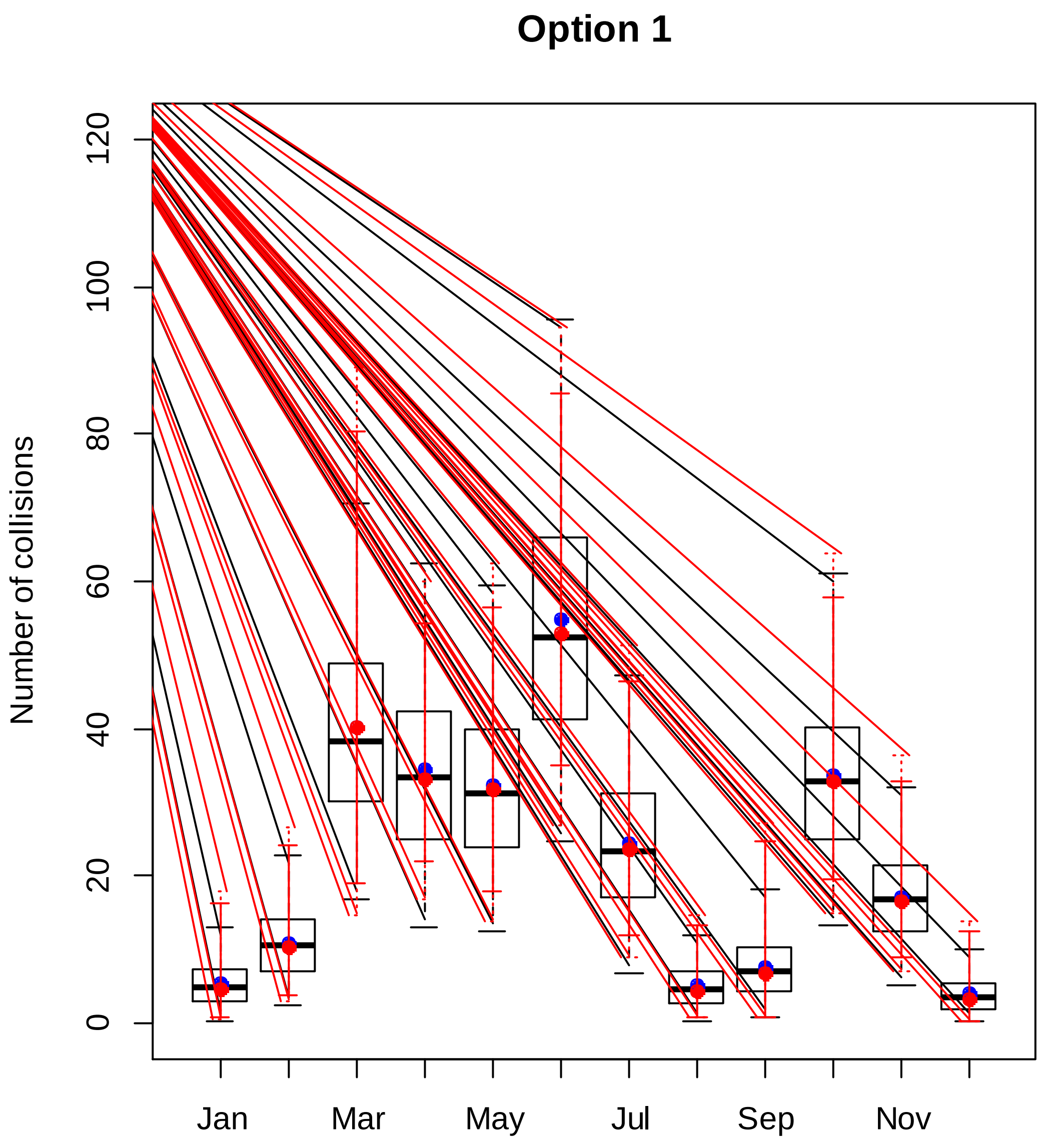

Figure 2. Box and whisker plot of Masden model outputs (in Table 9) using the parameters listed in Tables 5 and 6. The black horizontal lines are the median, the boxes the 25th and 75th percentiles and the black whiskers the 95% confidence interval (i.e. 2.5% - 97.5%). The mean Masden values (blue dots) and Band model outputs (red dots) have been overlaid. The solid red lines indicate the upper and lower Band outputs using 95% confidence intervals (i.e. 2.5% - 97.5%) from the seabird density GLM, the dotted red lines also include +/- 0.002 applied to the avoidance rate (i.e. 98.7 - 99.1%). It should be noted that for this figure the boxplot function has been modified from that defined in the Masden model to generate whiskers (black dotted lines) at the 95% confidence range for comparison with the intervals presented around the Band model outputs (red whiskers).

Comparing the Band model upper and lower estimates with those from the Masden model, it can be seen that the 95% confidence ranges generated by the Masden are generally fairly similar, although there is no consistent pattern (i.e. in some months the Band model outputs are wider and in others the Masden model outputs are wider). It should be noted that for comparison the boxplot function used in the Masden model has been modified slightly for Figure 2 to obtain the equivalent 95% confidence range as that presented for the Band model outputs.

It is interesting to note that the extent of the Band model ranges was more influenced by the uncertainty in the density estimates than the avoidance rate, the latter contributing a maximum of 30% to the range of collision estimates (peaking for the higher absolute collision estimates).

5 RECOMMENDATIONS FOR EDITING THE MASDEN MODEL

The Masden model script without modification (i.e. as downloaded) produces mean collision estimates which may be different (depending on input parameter values) from those obtained by the Band model for the following reasons:

- The simulated proportion at collision height can generate negative values (depending on the mean and SD entered),
- The simulated seabird densities are capped at 2/km^2, and
- Rotor RPM and blade pitch are simulated as a function of a randomly generated wind speed variable, using a tabulated relationship which is not based on actual turbine parameters (it should be noted that the reports which accompany the Masden model state that this relationship is over-ridden if a mean and SD for rotor speed and blade pitch are entered, however this is incorrect as the model code does not include a mechanism to perform this switch).
- The mean of the density values generated from the normal (or truncated normal) distribution may differ from the input mean values, due to inherent differences between the underlying distribution and the normal or truncated normal distributions.

As a consequence, the model should not be used for wind farm assessment without modification. The following steps can be taken to correct the above aspects. These modifications were applied to obtain the outputs in Table 9.

- The R script '*sampleCRH.R*' should be changed from:

```
sampleCRH <- function(meanCRH, sdCRH) {
  rnorm(1, meanCRH, sdCRH)
 }
```

To :

```
sampleCRH <- function(meanCRH, sdCRH) {
  rtnorm(1, meanCRH, sdCRH,lower=0,upper=1)
 }
```

This constrains the resampled collision height estimates to lie between 0 and 1. Note this is only necessary when using site specific flight height data (e.g. Option 1).

- The R script '*samplecount.R*' should be changed from:

```
sampleCount <- function(meancount, sdcount){
   rtnorm(1, meancount, sdcount,0,2)
 }
```

To:

```
sampleCount <- function(meancount, sdcount){
   rtnorm(1, meancount, sdcount,lower=0,upper=Inf)
 }
```

This removes the upper seabird density cap of 2.

- The text file '*sampleturbineparams.txt*' should be modified as follows.

Lines 3 to 10 (inclusive) shown below, should be commented out – add '#' at the beginning of each line. This prevents these lines from being used by R.

```
  ####ROTOR SPEED (related to wind speed)####
source("scripts\\get_rotor_plus_pitch_auto.txt")
randomSample<-sample(length(rotorSpeed),1)
sampledRotorSpeed[i]<-rotorSpeed[randomSample]

  ###PITCH (related to wind speed and linked to above)####
sampledRotorPitch[i]<-rotorPitch[randomSample]
Pitch = sampledRotorPitch[i]*pi / 180 #### Transform Pitch, needed for Collision Risk Sheet
```

Becomes:

```
####ROTOR SPEED (related to wind speed)####
#source("scripts\\get_rotor_plus_pitch_auto.txt")
#randomSample<-sample(length(rotorSpeed),1)
#sampledRotorSpeed[i]<-rotorSpeed[randomSample]

###PITCH (related to wind speed and linked to above)####
#sampledRotorPitch[i]<-rotorPitch[randomSample]
#Pitch = sampledRotorPitch[i]*pi / 180 #### Transform Pitch, needed for Collision Risk Sheet
```

- The following lines should then be pasted in below the commented lines:

```
## Modified script to generate resampled rotor speed and blade pitch from input data in
TurbineData.csv
ifelse(!is.na(TurbineData$RotationSpeedSD[t]), rotorSpeed<-
sampleRotorRadius(TurbineData$RotationSpeed[t], TurbineData$RotationSpeedSD[t]), rotorSpeed<-
TurbineData$RotationSpeed[t])
  sampledRotorSpeed[i]<-rotorSpeed

ifelse(!is.na(TurbineData$PitchSD[t]), rotorPitch<-sampleRotorRadius(TurbineData$Pitch[t],
TurbineData$PitchSD[t]), rotorPitch<-TurbineData$Pitch[t])
  sampledRotorPitch[i]<-rotorPitch
  Pitch=sampledRotorPitch[i]*pi / 180 #### Transform Pitch, needed for Collision Risk Sheet
```

This ensures that the Masden model will sample the RPM and blade pitch from the mean and SD values entered in the *TurbineData.csv* file rather than the windpower.csv file.

There is an option in the Masden script which allows the initial state for the random number generator to be set to a fixed value (this is set to 100 in the code: *'set.seed(100')*). The advantage of this is that results are repeatable (i.e. the same sequence of 'random' numbers is generated on each run of the model). However, failing to switch this off (or alternatively, setting the seed to a new value each time (e.g. using the CPU clock: '*set.seed(as.numeric(Sys.time()))*') will lead to unexpected outputs (e.g. identical results on every simulation).

The above aspects of the code are relatively straightforward to correct through editing of the Masden model code, however this requires an understanding of the R programming language.

More fundamentally, in its current state without modification (i.e. as available on the Marine Scotland Datasets webpage[2]) the Masden model uses inappropriate probability distributions for some parameters. As a consequence, there is a high likelihood that use of the Masden model will result in erroneous collision estimates (i.e. estimates which do not accurately reflect input parameters due to errors in the model code and the way data are simulated).

6 OTHER CONSIDERATIONS FOR STOCHASTIC COLLISION MODELLING

The Masden model in its unedited state samples rotor RPM and blade pitch jointly using wind speed. This approach correctly identifies that these turbine parameters are not independent of one another, but are closely related and jointly dependent on wind speed. However, while this is an appropriate method to model these variables, the relationship between wind speed and the turbine rotor operation has not been made available by the turbine manufacturers, therefore the accuracy of the relationship is unknown. Thus, to permit comparison of outputs with the Band model it was necessary to derive the mean values for RPM and blade pitch from the ones generated by running the Masden model (using the RPM, blade pitch, and windspeed relationships table provided with the Masden model). The alternative is to set the mean and SD using turbine data and modify the code (as described

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[2] http://marinedata.scotland.gov.uk/dataset/developing-avian-collision-risk-model-incorporate-variability-and-uncertainty-r-code

above) to make these variables independent of one another. This allows closer comparability with the Band model, but will inflate the overall variance of the outputs. Furthermore, this highlights the fact that there are several other components of the collision model which are related and which should therefore covary in a stochastic model.

A key example of this is the avoidance rate. Seabird avoidance rates have been estimated from long term datasets (Cook et al. 2014). The estimates are therefore mean values for the study periods used, and equivalent mean parameter estimates should be used for the other model input parameters (e.g. flight speed, proportion at collision height, etc. should be derived over similar time frames). It therefore follows that simulating each parameter around its mean value should ensure that the mean collision estimate obtained will correspond to the individual input parameter means. However, unless the parameters have been combined within each model iteration in such a way as to avoid inappropriate combinations the variance around the mean collision estimate will be inflated. Incorporating covariance in the model is an important consideration for development of a reliable stochastic model.

This is important, since the main objective of a stochastic collision model is to improve understanding of the variance around the mean estimates. As demonstrated above, the Masden model produces mean and median values which are very similar to those from the Band model. But because the parameters are simulated independently the overall ‘parameter space’ generated will be inflated to an unknown extent with a result that the collision estimates will also have wider confidence intervals than if the input parameters were simulated with realistic levels of covariance.

The proportion of birds at collision height can only be entered in the Masden model as a single value (mean and SD) which is applied as an annual average (although the model could be run for a single month or months to apply seasonal variation in this and other parameters). It would be appropriate to model collisions using a monthly value for this parameter if it can be estimated for a given location. This would require considerably more editing of the current scripts and is beyond the scope of the current project.

The simplest robust option for producing randomised density estimates for input to a stochastic collision model is to bootstrap the snapshot counts for a given month. The drawback of this approach is that for low density species there may be a limited number of non-zero counts from which to draw (i.e. there may be a very small range of possible outputs). A more flexible approach is to use a function such as *generateNoise* (MRSea Power) which uses the outputs from a model of the snapshot counts (e.g. GLM or GAM), including any over-dispersion parameter. Unlike bootstrapping, this method is not constrained by the original observations. For example, if the original sample only included snapshot counts of 1, 2 and 5 individuals, the bootstrap resamples will have the same three count sizes. In contrast, resamples obtained using *generateNoise* can take any integer value within the range defined by the model. In both cases the output is a vector of counts the same length as the original number of surveyed samples. The column sum divided by the total area of snapshots is a random density estimate for input to the CRM. Repeating this process generates bird density estimates that can be used to produce collision estimates incorporating uncertainty in species density at the project site in a statistically robust way.

As discussed above, seabird counts used to derive densities are poorly represented by the normal or truncated normal distribution. Thus, a stochastic CRM either needs to permit random number generation using different distributions (e.g. Poisson) or alternative parameter inputs (e.g. external generation of multiple densities using bootstrap methods which can be used in simulations as outlined in the paragraph above).

As described above, one option is to use the results of a Poisson GLM to generate random resamples which correspond to the observed distribution. However, there is no means in the unedited Masden script to specify alternative random number generation or alternative density inputs (the user must supply mean and SD values for use with a truncated normal random number generator).

The best option currently available is to calculate the mean and SD from the resampled GLM data (as above) and use these as Masden model inputs. The drawback to this is that a Poisson (or over-dispersed Poisson) process is likely to be poorly represented by the truncated normal distribution that the Masden model uses to sample densities from. The magnitude of difference between the underlying (over-dispersed Poisson) process and that obtained using the truncated normal as described above, depends on how close the mean density is to zero. At low mean densities (e.g. <0.5 birds / km^2) the truncated normal estimates are biased high (Figure 3), although this bias decreases as the mean increases and is effectively undetectable at higher (e.g. >1 bird / km^2 (Figure 4).

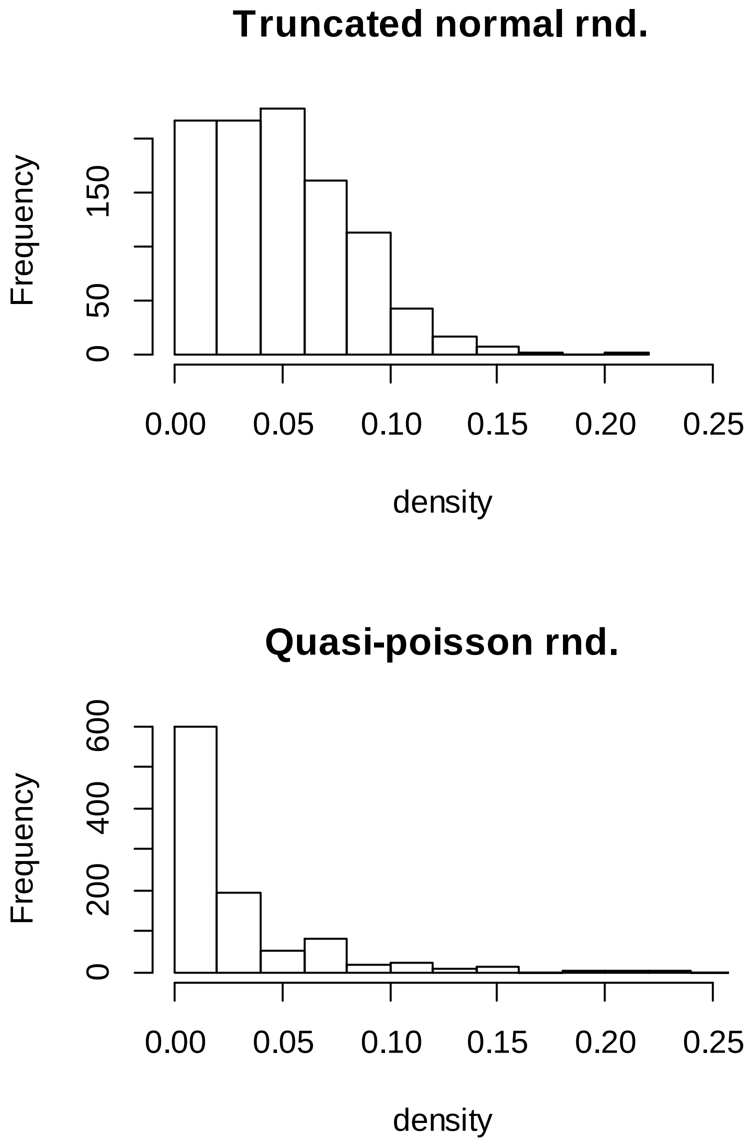

Figure 3. Low density resampled seabird densities, the values in the lower plot have been generated directly from an over-dispersed Poisson GLM using the generateNoise function. The values in the upper plot have been obtained using the mean and standard deviation (of the samples in the lower plot) as inputs to the rtnorm function (truncated normal random numbers). The truncated normal random deviates are shifted to the left compared with the underlying distribution. The original distribution (lower plot) has a mean (sd) of 0.031 (0.044) while the mean (sd) of the truncated normal distribution in the upper plot is 0.049 (0.032).

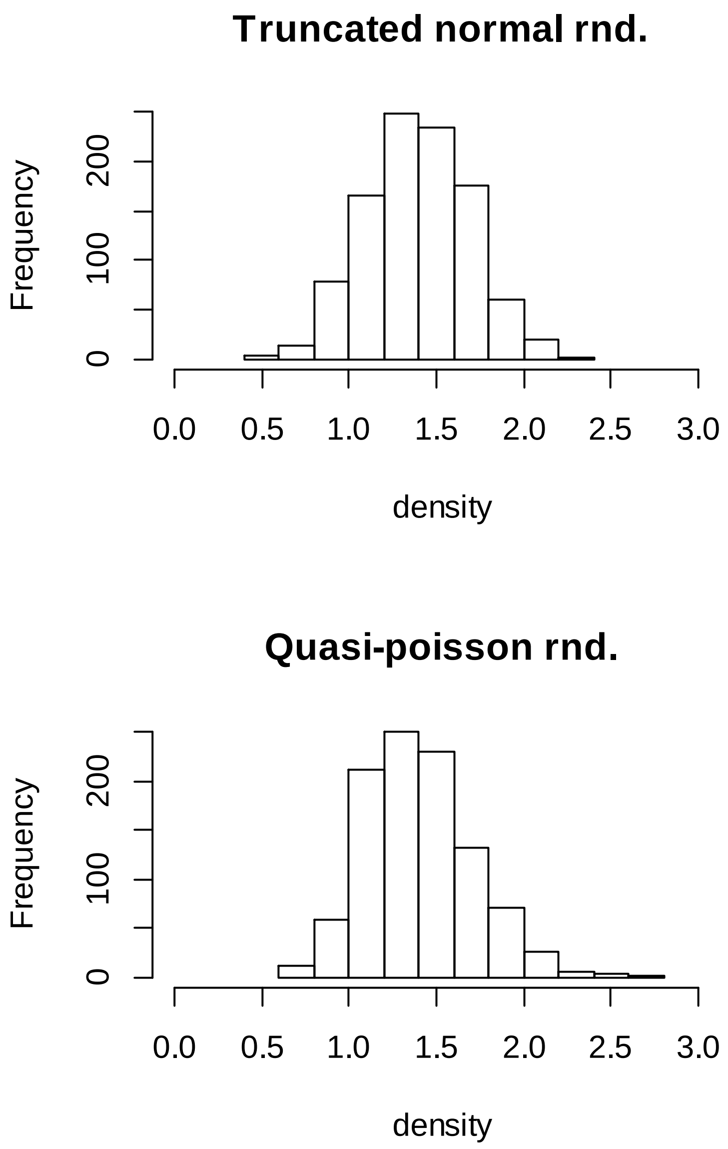

Figure 4. Medium density resampled seabird densities, the values in the lower plot have been generated directly from an over-dispersed Poisson GLM using the generateNoise function. The values in the upper plot have been obtained using the mean and standard deviation (of the samples in the lower plot) as inputs to the tnorm function (truncated normal random numbers). These illustrate that as density increases the bias declines to undetectable levels. The original distribution (lower plot) has a mean (sd) of 1.39 (0.304) while the mean (sd) of the truncated normal distribution in the upper plot is 1.39 (0.0297).

As noted above, the truncated normal distribution is used to obtain normal random numbers which are constrained by lower and/or upper limits. While this can prevent inappropriate values, the results are not necessarily a good match for the underlying process. In addition, the bird densities generated in the unedited Masden model using the truncated normal distribution have an upper limit which has been fixed at 2 birds / km^2. It is assumed that this is an error in the relevant script which if uncorrected risks generating incorrect densities for abundant species.

A GLM approach is also a robust method for estimating the proportion of birds at collision height with variability. The first step is to convert observed flight heights to a binary state (0 = not at PCH, 1 = at PCH). These data can then be modelled using a GLM with binomial errors. As for density estimates, these can be resampled directly to be used as CRM inputs. In order to use PCH data modelled in this manner with the unedited Masden model the mean and SD can be calculated across the resampled values. However, there is a potentially important error in the Masden script when using option 1 and site specific flight height data: the proportion of birds at collision height is simulated using a normal distribution (i.e. these are not truncated at zero) and it is therefore possible to obtain negative values for this parameter if the mean PCH is low, or the SD is large (or both). Using a negative value for PCH will result in a negative collision estimate, and reducing the summary values obtained. Unless there are a lot of negative values (i.e. resulting in a negative lower confidence interval) this is unlikely to be obvious in the summary outputs. This should be corrected (see section 5) prior to use of the Masden model.

On a practical level, the Masden model generates stochastic mortality estimates by nesting the calculations within a loop. New random numbers are drawn at the beginning of each run through the loop and the outputs of the model are stored at the end of each iteration. While this approach is conceptually straightforward, it is inefficient (i.e. the model runs slowly). Simulations can be undertaken much more efficiently through the use of vectorisation. This minimises the use of loops by generating multiple random values for each parameter in a single step and then multiplying these together to obtain tables of outputs which are the same as those obtained at the end of a looped process.

It is important to state that regardless of the method used (looped or vectorised), the results obtained are the same. Therefore, although the Masden code is slow compared with vectorised script, this does not preclude its use (although the time saving may be significant: as an example, to complete 1,000 simulations for a single species the run time for the unedited Masden code was 45 minutes, while a vectorised version achieved the same outputs in less than 4 seconds).

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SmartWind (2015) Collision Risk Modelling; Addressing uncertainty clarification note. Appendix J to the Response submitted for Deadline 1. Application reference: EN010053.

ANNEX 1.

Bird density modelling. Note that no surveys were conducted in November in the example dataset. For the CRM tests density parameters for November (mean, c.i.) were averaged across October and December. The original data comprises 22 surveys across a two year period, with regular snapshot counts (range: 362 – 461) collected by boat survey.

```
> summary(mod1)
Call:
glm(formula = Numbers ~ as.factor(Month) - 1 +  offset(log(area)), family = quasipoisson)

Deviance Residuals:
Min           1Q          Median       3Q             Max
-0.4781       -0.3944     -0.3220      -0.1577        21.1670
```

Coefficient	Estimate*	Std. Error	t value	Pr(>|t|)	Lower c.i. #	Upper c.i. #
Month1	0.1372	0.7360	-2.690	0.00716	0.02088	0.4424
Month2	0.3129	0.4604	-2.538	0.01118	0.10758	0.6814
Month3	1.0340	0.3630	0.075	0.93983	0.45868	1.9414
Month4	0.8727	0.2302	-0.635	0.52571	0.53050	1.3145
Month5	0.7795	0.2899	-0.926	0.35425	0.40813	1.2846
Month6	1.2580	0.2253	1.060	0.28911	0.78830	1.9160
Month7	0.5705	0.3448	-1.600	0.10959	0.26898	1.0572
Month8	0.1071	0.6657	-3.361	0.00078	0.02026	0.3119
Month9	0.1847	0.7806	-2.195	0.02822	0.02342	0.6111
Month10	0.8798	0.2760	-0.476	0.63391	0.4838	1.4401
Month12	0.0851	0.9014	-2.700	0.00695	0.0074	0.3446

* Note these estimates have been converted using ‘exp()’ to obtain values on the response scale.

The confidence intervals were obtained using function ‘confint()’

```
(Dispersion parameter for quasipoisson family taken to be 4.874795)
Null deviance: 3789.4  on 8812  degrees of freedom
Residual deviance: 3360.7  on 8801  degrees of freedom
AIC: NA
Number of Fisher Scoring iterations: 7
```

ANNEX 2.

The following sets out a method for calculating an overall (or average) variance for two variables which have their own mean and variances (i.e. the average variance for two monthly densities which each have their own average and variance).

For a two-sample calculation, the input parameters are:

- n1 and n2 (sample sizes, e.g. n1= 300, n2 = 400)
- x1 and x2 (sample means, e.g. x1= 25, c2 = 15)
- x.bar (mean of x1 and x2, e.g. x.bar = 20)
- v1 and v2 (variance estimates, e.g. v1 = 25, v2 = 9)

Calculate the overall error sum of squares:

ESS.total = (v1 * n1-1) + (v2 * n2-1)

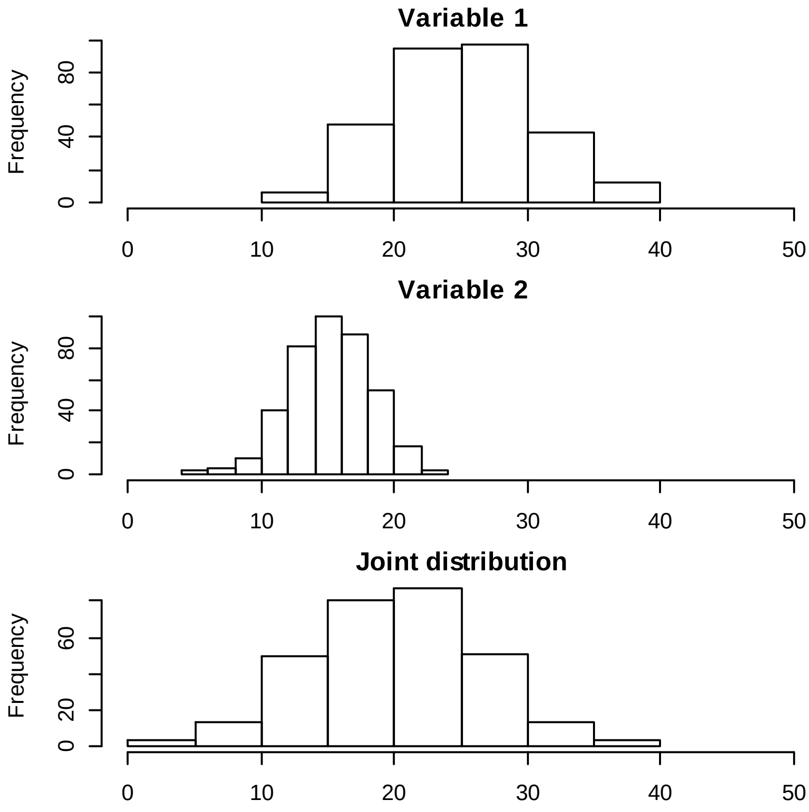
Calculate the overall group sum of squares:

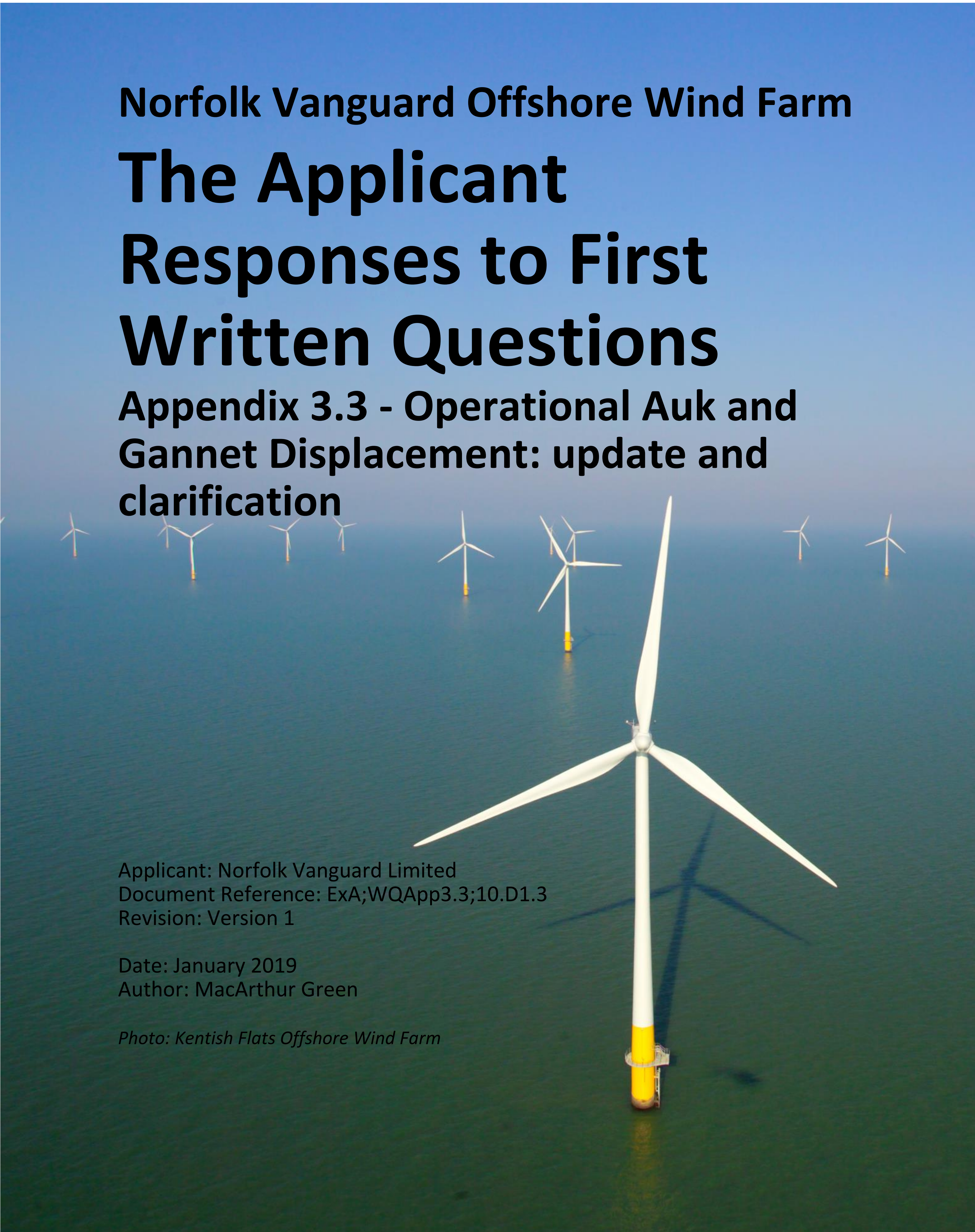
GSS.total = ((x1 – x.bar)^2 *n1-1) + ((x2 – x.bar)^2 *n2-1)

Calculate the overall variance:

Overall variance = (ESS.total + GSS.total) / ((n1+n2)-1)

Using the example values the following distributions are obtained:





Norfolk Vanguard Offshore Wind Farm

The Applicant Responses to First Written Questions

Appendix 3.3 - Operational Auk and Gannet Displacement: update and clarification

Applicant: Norfolk Vanguard Limited
Document Reference: ExA;WQApp3.3;10.D1.3
Revision: Version 1

Date: January 2019
Author: MacArthur Green

Photo: Kentish Flats Offshore Wind Farm

Date	Issue No.	Remarks / Reason for Issue	Author	Checked	Approved
12/12/2018	01D	First draft for Norfolk Vanguard Ltd review	MT	JKL, RWF	EV
07/01/2019	02D	Second draft for Norfolk Vanguard Ltd review	MT	RWF	EV
09/01/2019	03	Final version for submission	MT	RWF	EV

Executive Summary

This note provides an update to the operational displacement assessment for Norfolk Vanguard and addresses comments received from Natural England in their Relevant Representation.

A review of evidence for displacement effects for guillemot and razorbill has been undertaken and is included in this note. This review concludes that appropriate (and still precautionary) rates of displacement from wind farms for these species are 50% from within the wind farm itself and 30% within a 1 km buffer, combined with a maximum consequent mortality for displaced individuals of 1%. These rates, as well as the Natural England recommended rates of 30%-70% displacement and 1%-10% mortality have been applied in this update note.

Assessment is also presented using the upper and lower 95% confidence intervals on population abundance for puffin, razorbill, guillemot and gannet for project alone impacts.

Cumulative assessments for puffin, razorbill and guillemot are provided which include the figures presented in the Environmental Statements (ESs) for Hornsea Project 3 and Thanet Extension, and also include figures for the Hywind and Kincardine projects. Natural England also requested a cumulative displacement assessment for gannet, and this will be provided in a subsequent clarification note.

The conclusions of the updated assessment presented in this note remain the same as those in the original Norfolk Vanguard assessment (as presented in the ES), with no impacts greater than minor adverse for any species, either alone or cumulatively.

Table of Contents

Tables

Glossary

BACI	Before-After-Control-Impact
BDMPS	Biologically Defined Minimum Population Scale
CRM	Collision Risk Modelling
ES	Environmental Statement
FFC	Flamborough and Filey Coast (SPA)
HRA	Habitats Regulations Assessment
LSE	Likely Significant Effect
NE	Natural England
NV	Norfolk Vanguard
OWEZ	Offshore Wind Farm Egmond aan Zee
PEIR	Preliminary Environmental Impact Report
RR	Relevant Representation
SNCB	Statutory Nature Conservation Body
SPA	Special Protection Area
TDR	Time-Depth Recorder

1 INTRODUCTION

1. This note provides an update to the Norfolk Vanguard auk (guillemot, razorbill and puffin) and gannet displacement assessment (Vattenfall 2018) and addresses comments from Natural England (NE) in their Relevant Representation for the Norfolk Vanguard application.

2. The detailed comments provided by NE and where they have been addressed are provided in Table 1. Subsequent sections of this note provide updated displacement assessments for each species in relation to the project alone and cumulatively. A review of the evidence for auk displacement and mortality rates is provided in Annex 1.

Table 1 Comments on the auk displacement assessment provided by Natural England (2018) in their relevant representation.

Paragraph	Comment	Response and section of this document where more detail provided
4.2.5	**Assessment of Displacement Impacts** As advised in our Section 42 (Preliminary Environmental Information Report) response, NE require that the variability (uncertainty) in the underlying population estimates (i.e. through consideration of appropriately calculated upper and lower confidence intervals) is considered in the displacement assessments. Whilst the upper and lower confidence limits around the bird abundance estimates are presented in the tables in Annex 1 of Appendix 13.01, these have not been considered in the impact assessments for construction or operational displacement within the ES, with only the mean peak seasonal abundances considered. This approach needs to be revisited for all relevant species.	Additional displacement estimates have been provided in section 1.1 using the lower and upper 95% confidence interval density estimates. In addition, and in-keeping with the use of densities in the collision risk modelling (CRM) displacement using the median density estimates have also been provided. It should be noted that for the auk species considered in this note only the operational assessment has been updated since the construction assessment was accepted by NE.
4.2.14	**Cumulative and In-combination Assessments** We welcome the attempt by the Applicant to include figures for Hornsea 3 and Thanet Extension projects in the cumulative and in-combination assessments of displacement and collision risk. We assume that the figures presented in the assessments for these two sites have been obtained from the PEIRs for these projects, however it would be useful if this could be confirmed by the Applicant. There are a number of methodological issues and uncertainties identified	The cumulative tables provided in this update note (section 1.2) include the figures for Hornsea 3 and Thanet Extension presented in the ESs for those projects. However, it is

Paragraph	Comment	Response and section of this document where more detail provided
	with the baseline data and assessments completed by Hornsea 3 and some methodological issues identified with the assessments for Thanet Extension. Therefore, at this stage the figures for these projects have not been agreed and therefore this will mean that the cumulative and in-combination assessments will require updating during the process once figures for these projects have been agreed. Whilst we acknowledge that this is beyond the Norfolk Vanguard Applicant's control, this means that in addition to the issues noted above, Natural England are currently unable to reach any conclusions on the scale of impact of any cumulative or in-combination displacement and CRM impacts.	acknowledged that these may not be the final figures for these projects. In addition the cumulative tables include the ES estimates for the Hywind and Kincardine projects.
4.2.17	Natural England also suggests that a similar approach to that undertaken for the auk cumulative displacement assessments is undertaken for gannet. This also applies to the assessment of LSE for in-combination assessment of gannet displacement from the FFC SPA.	Assessment of cumulative gannet displacement will be provided in a separate note. Habitats Regulations Assessment (HRA) in relation to the Flamborough and Filey Coast SPA will be provided in a separate note.
4.2.18	The Applicant has considered that a value of 1% mortality when combined with the 70% displacement rate is appropriate for wintering auks. Whilst Natural England agrees that the mortality for auks is likely to be at the low end of the range, we do not agree that using 1% mortality for the cumulative assessment (with 70% displacement) can be considered the worst case scenario. Therefore, our recommendation is a range of mortality rates of 1-10% and displacement rates of 30-70%, with 70% displacement and 10% mortality as the worst case, which is the same as that used by the Applicant in the assessment of auk displacement impacts from the Vanguard project alone.	A review of the evidence for auk displacement from offshore wind farms has been conducted and is provided in Annex 1. This review has found that, on the basis of evidence from existing wind farms, a precautionary displacement rate from the wind farm itself would be 50%, with 30% displaced from a 1 km buffer. The review also concludes that the consequence, in terms of elevated mortality for displaced individuals, is very unlikely to exceed 1% (and that figure remains highly precautionary). Consideration of the impacts of displacement using the
4.2.19	We note that within the Natural England assessment scenario of 30% displacement and 1% mortality to 70% displacement and 10% mortality, a number of the annual predicted addition auk mortalities equates to greater than 1% of baseline mortality of both the largest BDMSP and the biogeographic populations. This is not insignificant and we advise further consideration be given to this once the figures are agreed. This also applies to the assessment of LSE for in-combination assessment of auk displacement from the FFC SPA.	

Paragraph	Comment	Response and section of this document where more detail provided
		range of rates recommended by NE is provided, along with that derived from the evidence review.
Summary of Natural England's key concerns		
3.1	**Lack of consideration of confidence intervals in bird abundance data for displacement assessments** As advised in our Section 42 (PEIR) response, NE require that the variability (uncertainty) in the underlying population estimates (i.e. through consideration of appropriately calculated upper and lower confidence intervals) is considered in the displacement assessments. Whilst the upper and lower confidence limits around the bird abundance estimates are presented in the tables in Annex 1 of Appendix 13.1, these have not been considered in the impact assessments for construction or operational displacement within the Environmental Statement Chapter, with only the mean peak seasonal abundances considered. This approach needs to be revisited for all relevant species. However, as the confidence limits are presented in the tables in Annex 1 of Appendix 13.1, Natural England has undertaken assessments based on these figures as well. We note that for construction displacement, consideration of the range of impacts predicted by considering the confidence limits does not alter the conclusions made by the Applicant for any species for displacement due to construction. The same is true for assessments of operational displacement (with the exception of red-throated diver, but this is largely due to the errors in the abundance data used within the operational matrices and that Natural England does not agree with the displacement and mortality rates used by the Applicant – see below).	Additional displacement estimates have been provided in section 1.1 using the lower and upper 95% confidence interval density estimates. In addition, and in-keeping with the use of densities in the collision risk modelling (CRM) matrices using the median density estimates have been provided. We note that NE has already concluded that this additional precautionary assessment does not change the conclusions for auks (see separate note for consideration of red-throated diver displacement; *Norfolk Vanguard Offshore Wind Farm The Applicant Responses to First Written Questions Appendix 3.1 - Red-throated diver displacement Doc. Ref. ExA;WQApp3.1;10.D1.3*).
5.4	**Auk (puffin, razorbill and guillemot) cumulative and in-combination operational displacement assessments** In addition to the overarching comment above regarding the issues/uncertainties around the data included for Vanguard alone and for Hornsea 3 and Thanet Extension, the Applicant has considered that a value of 1% mortality when combined with the 70% displacement rate is considered appropriate for wintering auks. Natural England notes that definitive mortality rates associated with displacement for seabirds, including auks are not known and	A review of the evidence for auk displacement from offshore wind farms has been conducted and is provided in Annex 1. This review has found that, on the basis of evidence from

Paragraph	Comment	Response and section of this document where more detail provided
	therefore we advise consideration of a range of mortality rates are used in assessments. Whilst Natural England agrees that the mortality for auks is likely to be at the low end of the range, we do not agree that using 1% mortality for the cumulative assessment (with 70% displacement) can be considered the worst case scenario. Therefore, our recommendation is a range of mortality rates of 1-10% and displacement rates of 30-70%, with 70% displacement and 10% mortality as the worst case, which is the same as that used by the Applicant in the assessment of auk displacement impacts from the Vanguard project alone. We note that within the Natural England assessment scenario of 30% displacement and 1% mortality to 70% displacement and 10% mortality, a number of the annual predicted addition auk mortalities equates to greater than 1% of baseline mortality of both the largest BDSMP and the biogeographic populations. This is not insignificant and we advise further consideration be given to this once the figures are agreed. This also applies to the assessment of LSE for in-combination assessment of auk displacement from the FFC SPA. Therefore, we advise that once the figures are agreed and the summed figures accurately presented that the assessment and conclusion of the LSE screening for auk in-combination displacement from FFC SPA is reviewed by the Applicant. We note that the cumulative displacement tables for all three auk species (guillemot, razorbill and puffin) all list the non-breeding seasons for Seagreen Alpha and Bravo as being N/A. We acknowledge that the Environmental Statement (ES) for these projects does not present displacement figures for the non-breeding seasons. However, graphs of monthly abundances of each auk species at each of the project sites across the two survey years are presented in the ES Chapter (Seagreen Wind Energy 2012). These indicate that both guillemot and razorbill were recorded in in all surveys of both Alpha and Bravo during the study period and puffins were recorded in lower numbers in most months. Therefore, consideration should be given to this in the cumulative assessments.	existing wind farms, a precautionary displacement rate from the wind farm itself would be 50%, with 30% from a 1 km buffer. The review also concludes that the consequence, in terms of elevated mortality for displaced individuals, is very unlikely to exceed 1% (and that figure remains highly precautionary). Consideration of the impacts of displacement using the range of rates recommended by NE is provided, along with that derived from the evidence review. HRA in relation to the Flamborough and Filey Coast SPA is provided in a separate note. Further review of the assessments for the Seagreen projects has been undertaken in order to determine appropriate estimates to be used for these projects in the cumulative assessment (section 1.2).

1.1 Operational displacement including uncertainty in density estimates

3. The displacement assessments presented in the ES (Vattenfall 2018) used the peak mean abundance in each biological season within the wind farm and 2 km buffer for all species considered at risk of displacement effects. NE requested that these assessments should also include consideration of uncertainty in the abundance estimates by providing additional displacement predictions obtained using the lower and upper 95% confidence intervals on the abundance estimates.

4. The rates of displacement and mortality recommended by the Statutory Nature Conservation Bodies (SNCBs) for auks are 30-70% displacement and 1-10% mortality (of displaced individuals) and for gannet 60-80% displacement and 1% mortality (NB: red-throated diver is also considered at risk of displacement effects and has been assessed in a separate note: Norfolk Vanguard Offshore Wind Farm The Applicant Responses to First Written Questions Appendix 3.1 - Red-throated diver displacement. Doc. Ref. ExA;WQApp3.1;10.D1.3). Displacement mortality calculated using the lower and upper bounds defined by these rates (30%-1% and 70%-10%) are provided in Table 2, Table 3 and Table 4, together with the abundance estimates used.
5. In addition, a comprehensive review of evidence relating to auk displacement from offshore wind farms (Annex 1), presents evidence for a displacement rate of 50% for birds within the wind farm and 30% within a 1 km buffer, both combined with a highly precautionary maximum mortality of 1%. As estimates of abundance within a 1 km buffer of the wind farm have not been calculated the following assessment update is based on the original 2 km buffer (i.e. it retains precaution due to both the larger buffer and also the application of a single displacement rate across both the wind farm and buffer). The evidence review also reported studies which suggest the potential for habituation to wind farms. This has not been considered in the assessment, however it indicates yet more precaution in the assessment.
6. It is also noted that, despite the figures in Tables 2 to 5 not having been presented in the ES, Natural England stated in their RR that they had taken the uncertainty into account in the auk displacement assessments and this did not change the conclusions presented in the ES. Thus, the project alone operational displacement impacts remain **negligible to minor adverse**.

Table 2. Puffin abundance estimates and summary displacement impacts.

Species	Site	Season	Abundance metric	Abundance (within wind farm and 2km buffer)	Displacement mortality at:		
					30% - 1%	50%- 1%	70%- 10%
Puffin	East	Breeding	Lwr 95%	0	0	0	0
			Median	40	0.1	0.2	2.8
			Mean	67	0.2	0.3	4.7
			Upr 95%	191	0.6	1	13.4
		Nonbreeding	Lwr 95%	0	0	0	0
			Median	32	0.1	0.2	2.2
			Mean	112	0.3	0.6	7.8
			Upr 95%	417	1.3	2.1	29.2
	West	Breeding	N/A	0	0	0	0
		Nonbreeding	N/A	0	0	0	0
	East & West	Annual	Lwr 95%	0	0	0	0
			Median	72	0.2	0.4	5
			Mean	179	0.5	0.9	12.5
			Upr 95%	608	1.8	3	42.6

Table 3. Razorbill abundance estimates and summary displacement impacts.

Species	Site	Season	Abundance metric	Abundance (within wind farm and 2km buffer)	Displacement mortality at: 30% - 1%	50%- 1%	70%- 10%
Razorbill	East	Breeding	Lwr 95%	156	0.5	0.8	10.9
			Median	526	1.6	2.6	36.8
			Mean	599	1.8	3	41.9
			Upr 95%	1150	3.5	5.8	80.5
		Autumn	Lwr 95%	229	0.7	1.1	16
			Median	520	1.6	2.6	36.4
			Mean	491	1.5	2.5	34.4
			Upr 95%	786	2.4	3.9	55
		Winter	Lwr 95%	74	0.2	0.4	5.2
			Median	291	0.9	1.5	20.4
			Mean	279	0.8	1.4	19.5
			Upr 95%	543	1.6	2.7	38
		Spring	Lwr 95%	212	0.6	1.1	14.8
			Median	762	2.3	3.8	53.3
			Mean	752	2.3	3.8	52.6
			Upr 95%	1302	3.9	6.5	91.1
	West	Breeding	Lwr 95%	96	0.3	0.5	6.7
			Median	259	0.8	1.3	18.1
			Mean	280	0.8	1.4	19.6
			Upr 95%	523	1.6	2.6	36.6
		Autumn	Lwr 95%	89	0.3	0.4	6.2
			Median	345	1	1.7	24.2
			Mean	375	1.1	1.9	26.3
			Upr 95%	729	2.2	3.6	51
		Winter	Lwr 95%	179	0.5	0.9	12.5
			Median	366	1.1	1.8	25.6
			Mean	348	1	1.7	24.4
			Upr 95%	495	1.5	2.5	34.7
		Spring	Lwr 95%	89	0.3	0.4	6.2
			Median	168	0.5	0.8	11.8
			Mean	172	0.5	0.9	12
			Upr 95%	269	0.8	1.3	18.8
	East & West	Annual	Lwr 95%	1124	3.4	5.6	78.7
			Median	3237	9.7	16.2	226.6
			Mean	3296	9.9	16.5	230.7
			Upr 95%	5797	17.4	29	405.8

Table 4. Guillemot abundance estimates and summary displacement impacts.

Species	Site	Season	Abundance metric	Abundance (within wind farm and 2km buffer)	Displacement mortality at:		
					30% - 1%	50%-1%	70%-10%
Guillemot	East	Breeding	Lwr 95%	544	1.6	2.7	38.1
			Median	2853	8.6	14.3	199.7
			Mean	2931	8.8	14.7	205.2
			Upr 95%	5629	16.9	28.1	394
		Nonbreeding	Lwr 95%	1377	4.1	6.9	96.4
			Median	1992	6	10	139.4
			Mean	2197	6.6	11	153.8
			Upr 95%	3441	10.3	17.2	240.9
	West	Breeding	Lwr 95%	439	1.3	2.2	30.7
			Median	1267	3.8	6.3	88.7
			Mean	1389	4.2	6.9	97.2
			Upr 95%	2493	7.5	12.5	174.5
		Nonbreeding	Lwr 95%	1220	3.7	6.1	85.4
			Median	2371	7.1	11.9	166
			Mean	2579	7.7	12.9	180.5
			Upr 95%	4083	12.2	20.4	285.8
	East & West	Annual	Lwr 95%	3580	10.7	17.9	250.6
			Median	8483	25.4	42.4	593.8
			Mean	9096	27.3	45.5	636.7
			Upr 95%	15646	46.9	78.2	1095.2

Table 5. Gannet abundance estimates and summary displacement impacts.

Species	Site	Season	Abundance metric	Abundance (within wind farm and 2km buffer)	Displacement mortality at:	
					60% - 1%	80% - 1%
Gannet	East	Breeding	Lwr 95%	27	0.2	0.2
			Median	149	0.9	1.2
			Mean	162	1	1.3
			Upr 95%	328	2	2.6
		Autumn	Lwr 95%	816	4.9	6.5
			Median	1375	8.3	11
			Mean	1630	9.8	13
			Upr 95%	2854	17.1	22.8
		Spring	Lwr 95%	0	0	0
			Median	557	3.3	4.5
			Mean	419	2.5	3.4
			Upr 95%	773	4.6	6.2
	West	Breeding	Lwr 95%	9	0.1	0.1
			Median	55	0.3	0.4
			Mean	95	0.6	0.8
			Upr 95%	241	1.4	1.9

Species	Site	Season	Abundance metric	Abundance (within wind farm and 2km buffer)	Displacement mortality at:	
					60% - 1%	80% - 1%
		Autumn	Lwr 95%	666	4	5.3
			Median	822	4.9	6.6
			Mean	823	4.9	6.6
			Upr 95%	1013	6.1	8.1
		Spring	Lwr 95%	0	0	0
			Median	9	0.1	0.1
			Mean	18	0.1	0.1
			Upr 95%	65	0.4	0.5
	East & West	Annual	Lwr 95%	1518	9.1	12.1
			Median	2967	17.8	23.7
			Mean	3147	18.9	25.2
			Upr 95%	5274	31.6	42.2

1.2 Cumulative displacement for auks

7. The Norfolk Vanguard cumulative displacement assessment for auks presented effects based on a 70% displacement rate and 1% mortality rate. Natural England requested the assessment should consider more precautionary estimates of displacement and mortality, specifically displacement between 30% and 70% and with mortality between 1% and 10%, affecting all individuals within 2 km of the wind farm boundary.

8. The evidence review provided in Annex 1 found that a precautionary basis for assessment is 50% displacement and a maximum of 1% mortality from within the wind farm and 30% displacement (and a maximum of 1% mortality) within the 1 km buffer of the wind farm. However, the updated displacement matrices provided below present the full ranges of displacement (0-100%) and mortality (0-100%). Thus, displacement impact predictions using both the precautionary rates advised by NE and the evidence-based ones (Annex 1) are presented.

9. Natural England requested updated estimates for several wind farms (Hornsea THREE, Thanet Extension and SeaGreen Alpha and Bravo), although NE also acknowledged that the first two of these were subject to ongoing discussions with the respective developers.

10. The magnitude of additional mortality for each auk species' largest BDMPS which would increase the background mortality by up to 3% is presented in Table 6.

Table 6. Auk populations in UK North Sea waters (see Furness 2015) used in the displacement assessment, the baseline mortality averaged across age classes (Error! Reference source not found.**) and the additional mortality which would increase the baseline rate by 1%, 2% and 3%.**

Species	Largest BDMPS	Average baseline mortality	Magnitude of additional mortality which increases baseline rate by:		
			1%	2%	3%
Guillemot	2,045,078	0.140	2,863	5,726	8,589
Razorbill	591,874	0.174	1,030	2,060	3,090
Puffin	868,689	0.167	1,451	2,901	4,352

1.2.1.1.1 Puffin

11. Norfolk Vanguard East and Norfolk Vanguard West are located beyond the mean maximum foraging range of any puffin breeding colonies. Outside the breeding season, puffins disperse from their breeding sites. Large numbers are found throughout the North Sea in the nonbreeding season (defined as August to February). It was during this period that numbers peaked on the Norfolk Vanguard East site with a mean maximum of 112 individuals (Table 7). The totals at risk on other North Sea wind farms are also presented in Table 7.

Table 7. Cumulative puffin numbers on wind farms in the North Sea.

Project	Breeding season	Non-breeding season
Aberdeen	42.0	81.7
Beatrice	2858.0	2434.8
Blyth Demonstration	235.0	122.8
Dogger Bank Creyke Beck A	37.0	295.2
Dogger Bank Creyke Beck B	102.0	742.9
Dogger Bank Teesside A	34.0	273.0
Dogger Bank Teesside B	35.0	328.7
Dudgeon	1.0	3.2
East Anglia ONE	16.0	32.0
East Anglia THREE	181.0	307.0
Galloper	0.0	0.8
Greater Gabbard	0.0	0.9
Hornsea Project One	1070.0	1257.0
Hornsea Project Two	468.0	2039.0
Hornsea Project Three	253.0	127.0
Humber Gateway	15.0	9.6
Hywind	119.0	85.0
Inch Cape	2956.0	2688.0
Kincardine	19.0	0
Lincs and LID6	3.0	6.0
London Array I & II	0.0	0.6
Moray East	2795.0	656.4

Project	Breeding season	Non-breeding season
Neart na Gaoithe	2562.0	2103.4
Race Bank	1.0	9.6
Seagreen A	2572.0	1526.0
Seagreen B	3582.0	3863.0
Sheringham Shoal	4.0	25.8
Teesside	35.0	18.0
Thanet	0.0	0.1
Thanet Extension	0.0	0.0
Triton Knoll*	23.0	70.7
Westermost Rough	61.0	35.0
Seasonal Total (Ex. NV)	**20079.0**	**19143.2.8**
Annual Total (Ex. NV)		**39222**
Norfolk Vanguard East	0	112
Norfolk Vanguard West	0	0
Seasonal Total (Inc. NV)	**20079**	**19255**
Annual Total (Inc. NV)		**39334**

12. Natural England does not consider a single combination of displacement and mortality in their assessment of impact, instead advising presentation of the ranges from 0 to 100% as provided in this note (e.g. Table 8), with a focus on 30% to 70% displacement and 1%-10% mortality. However, evidence in support of the use of a precautionary displacement rate of 50% within the wind farm, 30% within the 1km buffer and 0% thereafter, combined with a 1% mortality rate for guillemot and razorbill (Annex 1) is also considered appropriate for puffin. For the current assessment presented here application of this level of impact indicates that the baseline mortality rate for the relevant populations (North Sea BDMPS) would increase by less than 1% (Table 8).

13. Consequently, the potential cumulative annual displacement mortality for puffin would not materially alter the background mortality of the population and would be undetectable. Therefore, the magnitude of effect is assessed as negligible. As the species is of low to medium sensitivity to disturbance, the impact significance is **negligible to minor adverse**.

Table 8. Puffin cumulative displacement matrix. Levels of mortality which would increase the baseline mortality by percentage thresholds indicated by shading: green <1%; orange >1% and <2%; pink >2% and <3%; clear >3%:

		Mortality (%)																			
		1	2	3	4	5	6	7	8	9	10	20	25	30	40	50	60	70	80	90	100
Displacement (%)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	2	8	16	24	31	39	47	55	63	71	79	157	197	236	315	393	472	551	629	708	787
	4	16	31	47	63	79	94	110	126	142	157	315	393	472	629	787	944	1101	1259	1416	1573
	6	24	47	71	94	118	142	165	189	212	236	472	590	708	944	1180	1416	1652	1888	2124	2360
	8	31	63	94	126	157	189	220	252	283	315	629	787	944	1259	1573	1888	2203	2517	2832	3147
	10	39	79	118	157	197	236	275	315	354	393	787	983	1180	1573	1967	2360	2753	3147	3540	3933
	12	47	94	142	189	236	283	330	378	425	472	944	1180	1416	1888	2360	2832	3304	3776	4248	4720
	14	55	110	165	220	275	330	385	441	496	551	1101	1377	1652	2203	2753	3304	3855	4405	4956	5507
	16	63	126	189	252	315	378	441	503	566	629	1259	1573	1888	2517	3147	3776	4405	5035	5664	6293
	18	71	142	212	283	354	425	496	566	637	708	1416	1770	2124	2832	3540	4248	4956	5664	6372	7080
	20	79	157	236	315	393	472	551	629	708	787	1573	1967	2360	3147	3933	4720	5507	6293	7080	7867
	25	98	197	295	393	492	590	688	787	885	983	1967	2458	2950	3933	4917	5900	6883	7867	8850	9833
	30	118	236	354	472	590	708	826	944	1062	1180	2360	2950	3540	4720	5900	7080	8260	9440	10620	11800
	40	157	315	472	629	787	944	1101	1259	1416	1573	3147	3933	4720	6293	7867	9440	11013	12587	14160	15734
	50	197	393	590	787	983	1180	1377	1573	1770	1967	3933	4917	5900	7867	9833	11800	13767	15734	17700	19667
	60	236	472	708	944	1180	1416	1652	1888	2124	2360	4720	5900	7080	9440	11800	14160	16520	18880	21240	23600
	70	275	551	826	1101	1377	1652	1927	2203	2478	2753	5507	6883	8260	11013	13767	16520	19274	22027	24780	27534
	80	315	629	944	1259	1573	1888	2203	2517	2832	3147	6293	7867	9440	12587	15734	18880	22027	25174	28320	31467
	90	354	708	1062	1416	1770	2124	2478	2832	3186	3540	7080	8850	10620	14160	17700	21240	24780	28320	31860	35400
	100	393	787	1180	1573	1967	2360	2753	3147	3540	3933	7867	9833	11800	15734	19667	23600	27534	31467	35400	39334

1.2.1.1.2 Razorbill

14. Norfolk Vanguard East and Norfolk Vanguard West are located beyond the mean maximum foraging range of any razorbill breeding colonies. Outside the breeding season, razorbills migrate from their breeding sites. Large numbers are found throughout the North Sea in the nonbreeding seasons (covering the period from August to March). The annual total of razorbills at risk of displacement on the Norfolk Vanguard site (combined across the breeding season and all the nonbreeding seasons) was a mean maximum of 3,296 individuals (Table 9). The totals at risk on other North Sea wind farms are also presented in Table 9.

Table 9. Cumulative razorbill numbers on wind farms in the North Sea.

Project	Breeding season	Post-breeding season	Non-breeding season	Pre-breeding season
Aberdeen	161.0	64.4	7.3	25.7
Beatrice	873.0	833.0	555.3	833.0
Blyth Demonstration	121.0	90.9	60.6	90.9
Dogger Bank Creyke Beck A	1250.0	1576.0	1728.0	4149.0
Dogger Bank Creyke Beck B	1538.0	2097.0	2143.0	5118.7
Dogger Bank Teesside A	834.0	310.3	958.5	1919.0
Dogger Bank Teesside B	1153.0	592.3	1426.0	2953.3
Dudgeon	256.0	346.1	745.4	346.1
East Anglia ONE	16.0	26.0	154.5	336.0
East Anglia THREE	1807.0	1122.0	1499.0	1524.0
Galloper	44.0	43.0	105.5	394.0
Greater Gabbard	0.0	0.0	387.3	83.8
Hornsea Project One	1109.0	4812.3	1517.5	1802.8
Hornsea Project Two	2511.0	4220.5	719.5	1668.0
Hornsea Project Three	630.0	2020.0	3649.0	1236.0
Humber Gateway	27.0	20.0	13.4	20.0
Hywind	30	719.0	10	0
Inch Cape	1436.0	2870.0	651.0	N/A
Kincardine	2.0	0	0	0
Lincs and LID6	45.0	33.5	22.3	33.5
London Array I & II	14.0	20.4	13.6	20.4
Moray	2423.0	1102.6	30.2	168.3
Neart na Gaoithe	331.0	5492.4	507.8	
Race Bank	28.0	42.0	28.0	42.0
Seagreen A	5876		1003	
Seagreen B	3698		1272	
Sheringham Shoal	106.0	1343.0	211.3	30.2
Teesside	16.0	61.5	1.9	20.0
Thanet	3.0	0.0	13.6	20.9
Thanet Extension	N/A	N/A	34.0	50.0
Triton Knoll*	40.0	253.7	854.5	116.7
Westermost Rough	91.0	121.3	151.6	90.9
Seasonal Total (Ex. NV)	**26469**	**30233.2**	**20474.6**	**23093**

Project	Breeding season	Post-breeding season	Non-breeding season	Pre-breeding season
Annual Total (Ex. NV)				**100270**
Norfolk Vanguard East	**599**	**491**	**279**	**752**
Norfolk Vanguard West	**280**	**375**	**348**	**172**
Seasonal Total (Inc. NV)	**27368**	**31099.2**	**21101.6**	**24017**
Annual Total (Inc. NV)				**103586**

15. Natural England does not consider a single combination of displacement and mortality in their assessment of impact, instead advising presentation of the ranges from 0 to 100% as provided in this note, with a focus on 30% to 70% displacement and 1%-10% mortality. However, evidence is provided in Annex 1 in support of the use of a precautionary displacement rate of 50% with a 1% mortality rate for guillemot and razorbill. For the current assessment presented here application of this level of impact indicates that the baseline mortality rate for the relevant populations (North Sea BDMPS) would increase by less than 1% (Table 10).

16. Consequently, the potential cumulative annual displacement mortality for razorbill would not materially alter the background mortality of the population and would be undetectable. Therefore, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is **minor adverse**.

Table 10. Razorbill cumulative displacement matrix. Levels of mortality which would increase the baseline mortality by percentage thresholds indicated by shading: green <1%; orange >1% and <2%; pink >2% and <3%; clear >3%:

		Mortality (%)																			
		1	2	3	4	5	6	7	8	9	10	20	25	30	40	50	60	70	80	90	100
Displacement (%)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	2	21	41	62	83	104	124	145	166	186	207	414	518	622	829	1036	1243	1450	1657	1865	2072
	4	41	83	124	166	207	249	290	331	373	414	829	1036	1243	1657	2072	2486	2900	3315	3729	4143
	6	62	124	186	249	311	373	435	497	559	622	1243	1554	1865	2486	3108	3729	4351	4972	5594	6215
	8	83	166	249	331	414	497	580	663	746	829	1657	2072	2486	3315	4143	4972	5801	6629	7458	8287
	10	104	207	311	414	518	622	725	829	932	1036	2072	2590	3108	4143	5179	6215	7251	8287	9323	10359
	12	124	249	373	497	622	746	870	994	1119	1243	2486	3108	3729	4972	6215	7458	8701	9944	11187	12430
	14	145	290	435	580	725	870	1015	1160	1305	1450	2900	3626	4351	5801	7251	8701	10151	11602	13052	14502
	16	166	331	497	663	829	994	1160	1326	1492	1657	3315	4143	4972	6629	8287	9944	11602	13259	14916	16574
	18	186	373	559	746	932	1119	1305	1492	1678	1865	3729	4661	5594	7458	9323	11187	13052	14916	16781	18645
	20	207	414	622	829	1036	1243	1450	1657	1865	2072	4143	5179	6215	8287	10359	12430	14502	16574	18645	20717
	25	259	518	777	1036	1295	1554	1813	2072	2331	2590	5179	6474	7769	10359	12948	15538	18128	20717	23307	25896
	30	311	622	932	1243	1554	1865	2175	2486	2797	3108	6215	7769	9323	12430	15538	18645	21753	24861	27968	31076
	40	414	829	1243	1657	2072	2486	2900	3315	3729	4143	8287	10359	12430	16574	20717	24861	29004	33147	37291	41434
	50	518	1036	1554	2072	2590	3108	3626	4143	4661	5179	10359	12948	15538	20717	25896	31076	36255	41434	46614	51793
	60	622	1243	1865	2486	3108	3729	4351	4972	5594	6215	12430	15538	18645	24861	31076	37291	43506	49721	55936	62152
	70	725	1450	2175	2900	3626	4351	5076	5801	6526	7251	14502	18128	21753	29004	36255	43506	50757	58008	65259	72510
	80	829	1657	2486	3315	4143	4972	5801	6629	7458	8287	16574	20717	24861	33147	41434	49721	58008	66295	74582	82869
	90	932	1865	2797	3729	4661	5594	6526	7458	8390	9323	18645	23307	27968	37291	46614	55936	65259	74582	83905	93227
	100	1036	2072	3108	4143	5179	6215	7251	8287	9323	10359	20717	25896	31076	41434	51793	62152	72510	82869	93227	103586

1.2.1.1.3 Guillemot

17. Norfolk Vanguard East and Norfolk Vanguard West are located beyond the mean maximum foraging range of any guillemot breeding colonies. Outside the breeding season, guillemots disperse from their breeding sites. Large numbers are found throughout the North Sea in the nonbreeding season (defined as August to February). It was during this period that numbers peaked on the Norfolk Vanguard site with a mean maximum of 4,776 individuals (Table 11). The totals at risk on other North Sea wind farms are also presented in Table 11.

Table 11. Cumulative guillemot numbers on North Sea wind farms.

Project	Breeding season	Non-breeding season
Aberdeen	547.0	225.0
Beatrice	13610.0	2755.0
Blyth Demonstration	1220.0	1321.0
Dogger Bank Creyke Beck A	5407.0	6142.0
Dogger Bank Creyke Beck B	9479.0	10621.0
Dogger Bank Teesside A	3283.0	2268.0
Dogger Bank Teesside B	5211.0	3701.0
Dudgeon	334.0	542.0
East Anglia ONE	274.0	640.0
East Anglia THREE	1744.0	2859.0
Galloper	305.0	593.0
Greater Gabbard	345.0	548.0
Hornsea Project One	9836.0	8097.0
Hornsea Project Two	7735.0	13164.0
Hornsea Project Three	13374.0	17772.0
Humber Gateway	99.0	138.0
Hywind	249	2136
Inch Cape	4371.0	3177.0
Kincardine	632	0
Lincs and LID6	582.0	814.0
London Array I & II	192.0	377.0
Moray	9820.0	547.0
Neart na Gaoithe	1755.0	3761.0
Race Bank	361.0	708.0
Seagreen A	13606	4688
Seagreen B	11118.0	4112
Sheringham Shoal	390.0	715.0
Teesside	267.0	901.0
Thanet	18.0	124.0
Thanet Extension	49.0	837.0
Triton Knoll*	425.0	746.0
Westermost Rough	347.0	486.0
Seasonal Total (Ex. NV)	116985	95515
Annual Total (Ex. NV)		**208513**
Norfolk Vanguard East	**2931**	**2197**

Project	Breeding season	Non-breeding season
Norfolk Vanguard West	**1389**	**2579**
Seasonal Total (Inc. NV)	**121305**	**100291**
Annual Total (Inc. NV)		**221596**

18. Natural England does not consider a single combination of displacement and mortality in their assessment of impact, instead advising presentation of the ranges from 0 to 100% as provided in this note, with a focus on 30% to 70% displacement and 1%-10% mortality. However, evidence is provided in Annex 1 in support of the use of a precautionary displacement rate of 50% with a 1% mortality rate for guillemot and razorbill. For the current assessment presented here application of this level of impact indicates that the baseline mortality rate for the relevant populations (North Sea BDMPS) would increase by less than 1% (Table 12).

19. Consequently, the potential cumulative annual displacement mortality for razorbill would not materially alter the background mortality of the population and would be undetectable. Therefore, the magnitude of effect is assessed as negligible. As the species is of medium sensitivity to disturbance, the impact significance is **minor adverse**.

Table 12 Guillemot cumulative displacement matrix. Levels of mortality which would increase the baseline mortality by percentage thresholds indicated by shading: green <1%; orange >1% and <2%; pink >2% and <3%; clear >3%:

		Mortality (%)																			
		1	2	3	4	5	6	7	8	9	10	20	25	30	40	50	60	70	80	90	100
Displacement (%)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	2	44	89	133	177	222	266	310	355	399	443	886	1108	1330	1773	2216	2659	3102	3546	3989	4432
	4	89	177	266	355	443	532	620	709	798	886	1773	2216	2659	3546	4432	5318	6205	7091	7977	8864
	6	133	266	399	532	665	798	931	1064	1197	1330	2659	3324	3989	5318	6648	7977	9307	10637	11966	13296
	8	177	355	532	709	886	1064	1241	1418	1595	1773	3546	4432	5318	7091	8864	10637	12409	14182	15955	17728
	10	222	443	665	886	1108	1330	1551	1773	1994	2216	4432	5540	6648	8864	11080	13296	15512	17728	19944	22160
	12	266	532	798	1064	1330	1595	1861	2127	2393	2659	5318	6648	7977	10637	13296	15955	18614	21273	23932	26592
	14	310	620	931	1241	1551	1861	2172	2482	2792	3102	6205	7756	9307	12409	15512	18614	21716	24819	27921	31023
	16	355	709	1064	1418	1773	2127	2482	2836	3191	3546	7091	8864	10637	14182	17728	21273	24819	28364	31910	35455
	18	399	798	1197	1595	1994	2393	2792	3191	3590	3989	7977	9972	11966	15955	19944	23932	27921	31910	35899	39887
	20	443	886	1330	1773	2216	2659	3102	3546	3989	4432	8864	11080	13296	17728	22160	26592	31023	35455	39887	44319
	25	554	1108	1662	2216	2770	3324	3878	4432	4986	5540	11080	13850	16620	22160	27700	33239	38779	44319	49859	55399
	30	665	1330	1994	2659	3324	3989	4654	5318	5983	6648	13296	16620	19944	26592	33239	39887	46535	53183	59831	66479
	40	886	1773	2659	3546	4432	5318	6205	7091	7977	8864	17728	22160	26592	35455	44319	53183	62047	70911	79775	88638
	50	1108	2216	3324	4432	5540	6648	7756	8864	9972	11080	22160	27700	33239	44319	55399	66479	77559	88638	99718	110798
	60	1330	2659	3989	5318	6648	7977	9307	10637	11966	13296	26592	33239	39887	53183	66479	79775	93070	106366	119662	132958
	70	1551	3102	4654	6205	7756	9307	10858	12409	13961	15512	31023	38779	46535	62047	77559	93070	108582	124094	139605	155117
	80	1773	3546	5318	7091	8864	10637	12409	14182	15955	17728	35455	44319	53183	70911	88638	106366	124094	141821	159549	177277
	90	1994	3989	5983	7977	9972	11966	13961	15955	17949	19944	39887	49859	59831	79775	99718	119662	139605	159549	179493	199436
	100	2216	4432	6648	8864	11080	13296	15512	17728	19944	22160	44319	55399	66479	88638	110798	132958	155117	177277	199436	221596

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Annex 1

Are guillemots and razorbills displaced from operating offshore wind farms?

1. Garthe and Hüppop (2004), and Furness et al. (2013), identified guillemots and razorbills as among the seabird species most likely to be displaced by offshore wind farms. However, those assessments were based on knowledge of seabird ecology, and without evidence from operational offshore wind farms. Dierschke et al. (2016) reviewed evidence for displacement obtained from studies published up to 2016 that compared seabird abundances within and outside European offshore wind farms post-construction with baseline data from before construction. Studies at twenty different operational offshore wind farms found guillemots and/or razorbills present at twelve of these in large enough numbers for robust statistical analysis of avoidance or attraction, while in a further four cases the evidence was poor and inconclusive.

2. Guillemots were strongly displaced from five, weakly displaced from two, unaffected at two, strongly attracted at one, and at two others there was weak evidence of slight attraction. Razorbills were strongly displaced from two, weakly displaced from three, unaffected at two, not significantly attracted at any, but there was weak evidence of slight attraction at one. Large auks (guillemots or razorbills combined) were weakly displaced at two sites additional to those included for the single-species results, and there was weak evidence of slight displacement at another site.

3. The evidence is clear; guillemots and razorbills do tend to avoid offshore wind farms, but avoidance is incomplete and highly variable among sites. This very variable behavioural response may differ depending on ecological conditions. For example, birds may be displaced more strongly at times of year when they are not constrained, whereas during periods of stress they may be unwilling to be displaced. Variability in displacement may also be a response to configuration of wind farm sites. For example, Leopold et al. (2013) suggested that the lower displacement of alcids from Egmond aan Zee (OWEZ) than from the nearby Princess Amalia was likely to be due to closer spacing of turbines at Princess Amalia than at OWEZ, allowing more alcids to utilize the site with less dense turbine spacing.

How strong is the displacement effect?

4. Overall, Dierschke et al. (2016) concluded that the mean outcome across all offshore wind farms was ‘weak displacement’ for guillemot and for razorbill. They defined this, where the change in density was statistically significant, as less than an average of 50% reduction in density post-construction compared to pre-construction data.

5. The strongest displacement effect was reported for Thorntonbank and Bligh Bank offshore wind farms in Belgian waters (Dierschke et al. 2016). This study, used a Before-After-Control-Impact (BACI) design with monthly boat-based surveys using

Distance correction and spanning several years of data collection post-construction (Vanermen et al. 2016). There was a significant reduction, of 68%, in guillemot density but no significant reduction in razorbill density (but a non-significant reduction of 55%) post-construction at Thorntonbank (including a buffer zone of 0.5 km outside the wind farm). There was no significant reduction in guillemot or razorbill density in a buffer zone from 0.5 km to 3 km from the outer turbines (but they report a non-significant reduction of 24% in guillemot density and a non-significant increase of <10% in razorbill density in the outer buffer zone).

6. At Bligh Bank, there was a significant reduction, of 75%, in guillemot density and a significant reduction, of 67%, in razorbill density (including a buffer zone of 0.5 km outside the wind farm). In a buffer zone from 0.5 km to 3 km from the outer turbines at Bligh Bank there was a smaller (49%), but statistically significant, reduction in guillemot density, but no significant difference in razorbill density (but a non-significant reduction of 32%) between pre-construction and post-construction. Vanermen et al. (2014) reported displacement of guillemots at Bligh Bank as 71%, and showed that razorbill numbers at Bligh Bank increased between pre-construction and post-construction. This is largely the same data set as reported above in the analysis by Vanermen et al. (2016), so it is interesting to note the different conclusion regarding razorbill, which suggests variation across years in relation to ecological conditions.

7. From data collected during boat-based surveys over three years at alpha ventus in the German North Sea, Welcker and Nehls (2016) found that alcids (guillemots and razorbills combined) showed a significant avoidance of the wind farm. They reported a reduction in density of about 75% inside the wind farm plus a buffer of 300 m, compared with densities more than 2.5 km from the wind farm. The data indicated progressively less avoidance with increasing distance up to 2.5 km, after which no effect was apparent. The decrease in density from zero to 2.5 km from the wind farm was about 30% on average, so considerably less than seen within the wind farm.

8. From data collected by boat-based surveys, with Distance correction, in a large area during pre-construction and post-construction periods for two offshore wind farms, Leopold et al. (2013) reported statistically significant reduction in density of guillemots at Princess Amalia and OWEZ. Razorbill densities were significantly reduced at Princess Amalia, but there was no significant reduction at OWEZ, and the non-significant trend was close to zero for that site. Lindeboom et al. (2011) concluded that for these sites the counts in and around the wind farm indicated no marked avoidance by guillemots or razorbills. The percentage displacement of guillemots was not quantified by Leopold et al. (2013), but appeared to be between 30% and 70% at the wind farms. Percentage displacement of razorbills was not

quantified but appeared to be between 30% and 70% at Princess Amalia, but between 0% and 20% at OWEZ. There was no evidence of significant displacement from the buffer zone surrounding these two wind farms. Both species showed similar, or slightly stronger, displacement from an area used as a ship park within the survey area, indicating that alcids may show at least as strong displacement from other large structures in the marine environment as from offshore wind turbines (Leopold et al. 2013).

9. Fox et al. (2006) and Petersen et al. (2006) state that there was no statistically significant change in guillemot density within Horns Rev 1 wind farm post-construction compared to pre-construction. However, they did report that flying guillemots tended to show macro-avoidance, some birds flying around the wind farm rather than through it. There was a small and non-significant decrease in relative abundance of guillemots within the wind farm area, suggesting a possible small and incomplete displacement effect.

10. No displacement of guillemots was found as a consequence of construction of Thanet offshore wind farm, while displacement of razorbills was statistically significant but at a level of less than 50% (Ecology Consulting 2012, Percival 2013).

11. APEM (2016) showed that guillemot density decreased in the vicinity of London Array offshore wind farm during construction, but there were not sufficient post-construction data to reach a clear conclusion as to the extent of avoidance post-construction, although post-construction distribution appeared to indicate partial displacement.

12. At North Hoyle offshore wind farm, guillemot numbers within the wind farm increased by 55% post-construction compared to pre-construction (PMSS 2007).

13. At other sites, displacement of guillemots and/or razorbills was either less than 50%, or was highly variable so difficult to detect statistically and difficult to quantify, as summarised for each site by Dierschke et al. (2016).

14. Since that review, Vallejo et al. (2017) have reported on studies of guillemot numbers across an area of 360 km^2 that included Robin Rigg offshore wind farm. They concluded that relative abundance of guillemots remained similar within the Robin Rigg offshore wind farm across all development phases, and that their data show no significant displacement of guillemots by that wind farm. Indeed, guillemot relative density within the wind farm was marginally, but not significantly, higher post-construction than pre-construction.

15. It is clear that guillemots and razorbills are incompletely displaced from some offshore wind farms, and apparently not displaced at all from some. On average,

displacement results in densities within offshore wind farms that are about 50% of the density in the wider area around these sites. In some cases there is some slight displacement of birds from a buffer zone surrounding the outer turbines, but this also seems to vary among sites. The buffer zone where densities are reduced is generally less than 2 km wide, and in most cases appears to be no more than 500 m wide. Where it has been measured, guillemot and razorbill density increases across the buffer zone, with distance from the turbines, up to the 'background' density. Displacement results in densities within 2 km of the offshore wind farms being reduced by less than 30%, and most studies report no significant reduction in density within the buffer zone.

16. An evidence-based, but still precautionary, assessment of displacement of alcids by offshore wind farms might assume that alcid densities would be reduced inside offshore wind farms by 50% relative to densities in the surrounding area, and by 30%, on average, across a 1 km buffer zone surrounding the wind farm. There are very few examples where displacement is greater than this, and many cases where it is much less.

Is there evidence for habituation of guillemots and razorbills to offshore wind farms?

17. If guillemots and razorbills habituate over time to the presence of offshore wind farms, then habitat loss might be negligible in the long term. In relation to Thorntonbank and Bligh Bank offshore wind farms, Vanermen et al. (2012) concluded 'During recent surveys in 2012, good numbers of auks were encountered inside the wind farm. From an ecological point of view, the presence of auks is very interesting, and we wonder if these self-fishing species are already habituating to the presence of the turbines, and if they will profit from a (hypothetical) increase in food availability'. It has already been seen that cormorants and shags have habituated to offshore wind farms and have learned to aggregate at these sites where they can roost on turbine railings and surfaces, and forage on the aggregations of fish that occur around turbine foundations and scour protection (Dierschke et al. 2016). However, the generally modest increases in fish abundance within wind farms that have been detected so far seem to be mainly for benthic fish such as gadoids and blennies (Bergstrom et al. 2013, van Hal et al. 2017, Stenberg et al. 2015). It remains to be seen whether abundance of pelagic-feeding fish such as sandeels, sprats and young herring that are preferred prey of guillemots and razorbills also regularly show an increased abundance or local concentration within offshore wind farms.

18. Leopold and Verdaat (2018) found some evidence for auks habituating to Luchterduinen offshore wind farm and suggested a methodology to assess the extent of such habituation. They reported that '*on relatively many occasions, birds of*

both these species [guillemots and razorbills] were seen to dive within the wind farm (as well as just outside). We could see no difference in behaviour of these birds inside and outside the wind farm. Diving birds often dived several times in succession, indicating that diving was not a panic reaction in response to them suddenly seeing a working turbine…. This has important implications for how we must judge wind farm effects on seabird ecology. If birds refrain from foraging within wind farms, the entire footprint of any wind farm is lost as feeding habitat. If, on the other hand, birds that are found within the wind farm forage normally, the amount of habitat loss would merely be the footprint of the wind farm times the reduced density of the species involved…. It is also possible that seabirds could learn to (or are already learning to) exploit the new habitat of offshore wind farms to their advantage…. Our observations suggest that they may be on this track: both guillemots and razorbills are now feeding in offshore wind farms'.

19. Evidence for guillemots and razorbills habituating to the presence of operational offshore wind turbines is very limited, but there are hints, as cited above, that this may be occurring. Further evidence of alcid behaviour at operational offshore wind farms would be desirable, potentially following the protocols proposed by Leopold and Verdaat (2018).

What are the likely consequences of displacement for individuals?

20. Displacement could influence individual guillemots or razorbills if offshore wind farm barrier effects or habitat loss result in a change in the bird's energy budget. Under some circumstances, though not all, displacement could increase energy costs, or could result in decreased energy intake. The former could arise if birds had to fly more to avoid offshore wind farms or to reach more distant foraging areas. The latter could arise if displacement was to an area of lower quality habitat where food capture rates were lower, or if displacement resulted in an increase in guillemot or razorbill density on the sea, with a consequent increase in intra-specific competition. Alternatively, displacement may have no effect on individuals if birds are displaced into equally good habitat so that their energy budget is unaffected, or if birds could buffer any impact on energy budget by adjusting their time budget (for example by spending a higher proportion of the time foraging rather than resting, in order to compensate for an increase in energy budget).

21. Density-dependent competition for food could apply in this species during the nonbreeding season if densities of guillemots and razorbills were high enough to deplete their food resource (small fish such as sprats, young herring, sandeels) or to result in reduced catchability of fish due to disturbance competition between foraging alcids. Then if birds had to increase effort to obtain their food requirements, they could potentially reach a limit where they had no more time available to

increase foraging effort further, or were at an energy ceiling that would not permit them to work harder without incurring a loss in body condition (Drent and Daan 1980).

22. Little is known about nonbreeding season energy budgets of guillemots and razorbills, but this information is key to understanding the possible consequences of displacement. If individuals are in relatively good condition during the nonbreeding season and spend only a small proportion of their daily activity budget in foraging, they may have the capacity to buffer against any additional energetic expense of displacement and barrier effects. On the other hand, if individuals have to work to capacity (either in terms of time allocated to foraging or in terms of physiological limits to energy expenditure), then there would be no scope for buffering any additional costs resulting from displacement or barrier effects.

23. Searle et al. (2017) developed an individual-based model to assess impact of displacement and barrier effects for breeding guillemots and razorbills (and puffins and kittiwakes) in the Forth-Tay region during the chick-rearing period. No equivalent model has yet been developed for guillemots and razorbills in the nonbreeding season, due at least in part to the lack of necessary data for the nonbreeding season. Searle's model was tested against scenarios with offshore wind farms in the Forth-Tay region close to the breeding colonies. In the tested scenarios, birds were commuting several times a day from the colony to foraging areas, so potentially affected by barrier effects and displacement; the models assumed a 60% displacement from wind farms plus a buffer zone of 500 m around each wind farm, and a 100% barrier effect (all birds flying around rather than through wind farms).

24. For breeding guillemots, scenarios with offshore wind farms placed relatively close to the study colony (Isle of May) resulted in additional adult guillemot mortality of between 0.003% to 0.31%, depending on modelled food abundance and distribution. For breeding razorbills, scenarios with offshore wind farms placed relatively close to the study colony (Isle of May) resulted in additional adult razorbill mortality of between 0.08% and 0.17%, depending on modelled food abundance and distribution. These scenarios do not apply to birds in the nonbreeding season, but strongly suggest that impacts of displacement and barrier effects for guillemots and razorbills have a very small impact on adult survival, even when tested in scenarios with multiple offshore wind farms placed close to colonies between nest sites and foraging grounds.

25. In the context of overwinter survival, it is relevant that in many seabird species, including alcids, most mortality occurs during winter (e.g. Coulson et al. 1983, Reynolds et al. 2011). This may be caused by a variety of factors, such as winter storms (Anker-Nilssen et al. 2018). However, there is also evidence that seabirds

tend to be heavier in winter than during the breeding season (e.g. Coulson et al. 1983). It is inferred from this that most seabirds have relatively little difficulty in finding enough food during the nonbreeding season so can achieve higher body condition that buffers against short periods of adverse weather conditions. For example, puffins are 20-30% heavier in winter than in summer as a result of storing fat during the nonbreeding season, and the same is true of guillemots (Harris et al. 2000, Anker-Nilssen et al. 2018). An implication is that their body condition may not be greatly affected by plausible levels of displacement or disturbance, since these birds are capable of maintaining high body weight through winter.

26. This leads to consideration of when during the nonbreeding season there may be critical periods that influence survival (bottlenecks). One possibility for alcids is during autumn when birds moult and become temporarily flightless. At this time it is critical that birds are in an area where there is a reliable food supply. There have been ‘wrecks’ of guillemots and razorbills in autumn that have sometimes been suggested as having been related to flightless auks being unable to find food. Many autumn ‘wrecks’ seem to involve mainly juvenile birds, suggesting that the mass mortality of guillemots or razorbills in autumn affects large cohorts of inexperienced young birds that fail to find suitable feeding areas early in life. However, in some cases autumn ‘wrecks’ have involved mostly adult birds, in heavy flight feather moult, suggesting that a spatial mis-match between distributing of moulting adults and a reliable food supply can cause mortality events.

27. Another possible ‘bottleneck’ occurs during mid to late winter, when prey depletion through the nonbreeding season may have occurred and daylength is short and storms may make feeding difficult. Guillemots and razorbills are able to feed at night (for example, guillemots overwinter in the Arctic in places where there is continuous dark for many weeks through mid-winter, and they are able to find food and maintain high body weights during that period). However, it is thought that guillemots and razorbills mainly feed during daylight hours and tend to rest during the night. We are aware of only one study that has collected data on foraging activity of guillemots through the winter by deploying time-depth recorders (TDRs) (Daunt et al. 2007).

28. Daunt et al. (2007) deployed TDRs and geolocators on breeding adult guillemots at the Isle of May in July 2005. Data were recovered from 13 of these birds in June 2006. They had moved into the central North Sea in late summer and autumn, during which period foraging effort (in terms of time spent diving) was low. The birds spent winter (November to March) in the southern North Sea. Foraging effort was slightly higher in November-March than it had been in autumn, though lower in January-February than in November-December. Birds returned to the vicinity of the

Isle of May from late February. Foraging effort peaked in March, but in April was the lowest of the year. These data are from just one nonbreeding period (2005-06), so it is impossible to assess how representative the data are for other years. The data suggest that if there is a ‘bottleneck’ when foraging is most challenging, that may occur during winter, or possibly in early spring when birds first return to the vicinity of the colony. However, the mean number of dives per 24 hrs varied only between a minimum of 160 during April and a maximum of 280 during March. Diving was rarely recorded at night during April to November, but during December to March dives at night represented about 30% of the total. This also suggests that birds may have been working harder in December to March.

29. Mean foraging depth, and dive duration, showed little seasonal variation, averaging around 20 m and 70 s respectively, but nocturnal dives tended to be much shallower and shorter than daytime dives. Overall, birds spent about 6 to 7 hours in foraging each day, and this showed little seasonal variation apart from being lower in autumn. These figures suggest that there may be considerable flexibility in guillemot time budgets that would potentially allow birds to increase foraging effort if required. It would be very helpful to have data on these nonbreeding season foraging metrics for guillemots and razorbills for a number of years in order to assess whether foraging effort reaches an upper limit in some years, perhaps relating to abundance of food fish stocks or to other environmental factors. But these preliminary data suggest some flexibility and headroom in foraging effort during the nonbreeding period. That tentative conclusion matches general understanding of bird time budgets; larger birds generally spend a smaller percentage of their time in foraging than do smaller birds. This is the case at all times of year, and was demonstrated a long time ago for breeding seabirds.

30. Studying seabirds breeding at the Farne Islands, Northumberland, Pearson (1968) showed that breeding guillemots spent 16% of daylight hours foraging, compared to 4-8% by shags, 37% by puffins, 43-57% by kittiwakes and 54-103% by Arctic terns. This also suggests that, compared to small birds, larger birds, such as guillemots and razorbills, are likely to have more ‘spare’ time that could be put into foraging if necessary. Studies of breeding guillemots in Shetland further demonstrate the potential for flexibility of foraging effort in this species; in 1991, when sandeel abundance was moderate, guillemots that were ‘off-duty’ (not incubating or brooding) spent 478 minutes during daylight resting at the nest site per day. In 1990, when sandeel abundance was extremely low, guillemots that were ‘off-duty’ spent 28 minutes during daylight resting at the nest site per day. Foraging trips in 1990 lasted more than twice as long as in 1991, although the increased effort was unable to compensate for low food availability and chick mortality at the colony in 1990 (22%) was eleven times higher than in 1991 (Uttley et al. 1994). Smout et al. (2013)

and Kadin et al. (2016) also found that guillemots can increase foraging effort in years of poorer food supply or quality around the colony.

31. For guillemot, Horswill and Robinson (2015) recommend use of baseline age of first breeding at 6 years old, adult (4th year and older) survival 0.939, 2-3 year survival 0.917, 1-2 year survival 0.792 and juvenile (first year) survival 0.56. The much lower survival of juveniles suggests that if there were impacts of displacement from offshore wind farms then those might be most likely to arise among juvenile birds rather than adults. Adult and immature survival rates are influenced by the amount of oil pollution, by weather conditions (survival is lower with warmer sea temperatures and higher winds), and by abundance of prey fish (Harris and Bailey 1992, Sandvik et al. 2005, Votier et al. 2005, 2008).

32. For razorbill, Horswill and Robinson (2015) recommend use of baseline age of first breeding at 5 years old, adult (third year and older) survival 0.895, immature survival (0-2 years) 0.63. The much lower survival of immatures suggests that if there were impacts of displacement from offshore wind farms then those might be most likely to arise among immature (especially juvenile) birds rather than adults. Adult survival rates are influenced by weather conditions (survival is lower with warmer sea temperatures and higher winds), and by abundance of prey fish (Sandvik et al. 2005).

33. The annual mortality of adult guillemots is around 6% per annum and that of adult razorbills is around 10% per annum, and this will include any mortality caused by existing human impacts such as oil pollution, hunting, fishing bycatch and disturbance, as well as ‘natural’ mortality. Given that all offshore wind farms in UK North Sea waters combined represent an extremely small fraction of potential foraging habitat of guillemots and razorbills within UK North Sea waters, it would seem appropriate to assess the plausible additional mortality caused by offshore wind farm displacement, barrier effects and increased ship traffic as also being extremely small in relation to the total annual mortality, given that this total annual mortality already includes any impact of existing (baseline) human impacts. In that context, to suggest that displacement from an offshore wind farm might increase mortality by 5% or more for all individuals that are displaced seems inconsistent with a total annual mortality of guillemot adults of only 6% or razorbill adults of 10%, when that already includes all impacts from existing human activities throughout the entire year.

What are the likely consequences of displacement for the population?

34. Sutherland (1996) and Newton (1998) pointed out that for migrant birds, such as guillemots and razorbills, population change following habitat loss in their nonbreeding area would depend on the relative strength of density-dependence in

the breeding area and in the nonbreeding area. If the population was regulated by density-dependent competition for breeding resources then habitat loss in the nonbreeding area may be unimportant. Goss-Custard et al. (1997) also pointed out that nonbreeding season habitat loss would only result in a decrease in a waterbird population if the population was subject to density-dependent competition for resources and population size was at carrying capacity of the environment.

35. Evidence strongly indicates that guillemots and razorbills, as other alcids and seabirds in general, are limited by competition for safe breeding sites, either through limitations in food resource surrounding the colony (Crespin et al. 2006, Elliott et al. 2009, Jovani et al. 2016, Sandvik et al. 2016), or through limitation in suitable nest sites (Kokko et al. 2004, Mitchell et al. 2004, Pontier et al. 2008) or a combination of these (Furness and Birkhead 1984, Birkhead and Furness 1985, Wakefield et al. 2017). This would suggest that their population size will be limited by breeding habitat suitability and may not be limited by wintering habitat. Loss of wintering habitat might, therefore, have little or no impact on guillemot or razorbill numbers unless habitat loss was so extensive that nonbreeding season habitat became a limiting factor for the population because their density increased so much that interference competition or prey depletion became a driving factor.

36. Guillemots and razorbills breeding at colonies in the North Sea mostly remain within the North Sea through the nonbreeding period, but these are joined by some birds from Norwegian Sea and Barents Sea colonies (Anker-Nilssen et al. 2000, Wernham et al. 2002, Cherenkov et al. 2016). There are at least 1.5 million guillemots in the North Sea in the nonbreeding period (Blake et al. 1984, Furness 2015). These birds are distributed across the entire North Sea (Camphuysen 2002, MERP), though with slightly higher densities in areas near to colonies and over some shallow sand banks where foraging may be most profitable, and low densities in the small area near Norway where the North Sea is particularly deep (MERP). Guillemots can dive to at least 100 m depth, so can access the sea bed across almost the whole North Sea, but probably prefer to forage in water less than 40 m deep (Daunt et al. 2007). The North Sea is about 750,000 km^2, so the mean density of guillemots is around 2 birds/km^2. This is consistent with the latest mapping of guillemots, which found 2 to 3/km^2 in shallower parts of the North Sea in January, but <1/km^2 in the deepest areas off Norway (MERP). All constructed, consented and proposed offshore wind farms in the North Sea plus 2 km buffers around these sum to approximately 5,000 km^2. If 50% of guillemots were displaced from all offshore wind farms and a buffer area of 2 km around these, and guillemot density was the average of 2 birds per km^2, then 5,000 birds would be displaced. Assuming that all displaced birds remained in the North Sea in places away from offshore wind farms, that would increase the density of guillemots in the rest of the North Sea from 2/km^2 to 2.007/km^2. Even if

offshore wind farms all occupied higher quality habitat so that guillemot density was twice the North Sea average and so twice as many birds were displaced, the density elsewhere would only increase to 2.013/km^2. It is difficult to imagine any biological mechanism that would result in detectable increase in density-dependent competition as a result of such a small increase in mean density of the population. A similar calculation for razorbill would reach this same conclusion. By comparison, the main food fish of guillemots and razorbills fluctuate in abundance much more dramatically. For example, the southern North Sea sandeel has varied between a spawning stock biomass of 76,000 tonnes and 996,000 tonnes (a 13-fold range of densities), while recruitment of young fish in that stock has varied from 25.4 billion to 520.3 billion (a 20-fold range of densities) in different years (ICES 2018). Fluctuations in food abundance seem likely to be much stronger drivers of demography than changes in guillemot and razorbill densities, and that is consistent with the observation that there is a significant relationship between guillemot and razorbill survival rates and food abundance (Sandvik et al. 2005).

37. Probably the most likely consequence is that displacement will have impacts on guillemot and razorbill populations that are too small to detect. Even though there are now many offshore wind farms in the southern North Sea, the total area of these represents a very small fraction of the habitat used by nonbreeding guillemots and razorbills throughout the southern North Sea, so that cumulative habitat loss is very small. The increase in density of guillemots and razorbills caused by displacement away from offshore wind farms will therefore be extremely slight at the regional or biogeographic scale. However, this can be put into some context by looking at the impact of habitat loss for estuarine wader populations that feed on mudflats during the nonbreeding season. Estuarine populations of waders are known to deplete prey in mudflats through the nonbreeding season, and are known to be affected by interference competition. So impacts on estuarine waders are likely to be greater than impacts on populations of birds that are not at the carrying capacity of their nonbreeding habitat.

38. It is known that many shorebirds that feed on mudflats are subject to strong interference competition and prey depletion (Goss-Custard et al. 2006). Estuarine habitat loss caused by barrages at Cardiff Bay and Rhymney resulted in an increase in mortality of 3.17% of displaced redshanks, a species known to be subject to strong density-dependent competition for food in winter due to both prey depletion and interference (Goss-Custard et al. 2006). Oystercatchers are also known to be strongly susceptible to interference competition in winter on tidal mudflats. At Oosterschelde, Netherlands, two-thirds of the tidal mudflat area was destroyed by coastal engineering works (the Delta Works). There was no difference in oystercatcher winter adult survival or in movement rates before and after this

habitat loss, although survival was reduced in severely cold winters compared to mild winters (Duriez et al. 2009). A study of the consequences of saltmarsh habitat loss for individually colour marked dark-bellied Brent geese followed the fate of displaced geese for 13 years after loss of saltmarsh habitat (Ganter et al. 1997). Displaced birds moved more often to less preferred sites that were not filled to capacity than did control birds. However, no significant differences in subsequent survival or fecundity of displaced birds could be found compared to control birds, although there may have been a slight but not statistically significant trend towards displaced birds performing less well than controls (Ganter et al. 1997). The researchers concluded that ‘if alternative sites are available there may be no significant fitness consequences to forced dispersal’ (i.e. displacement).

39. Based on our understanding of their winter feeding ecology and susceptibility to density-dependent competition, any effect of displacement of guillemots and razorbills would be expected to be less than seen in redshanks, and would be unlikely to be greater than seen in oystercatchers or dark-bellied Brent geese.

40. Despite the uncertainty about impacts on nonbreeding guillemots and razorbills, the available evidence suggests that the most likely result is that there will be little or no impact on adult survival, and that any impact would probably be undetectable at the population level. However, data from further tracking studies, as pioneered by Daunt et al. (2007), on the time-activity budgets of nonbreeding guillemots and razorbills, and on their movements within winters, would allow stronger conclusions to be reached.

41. To conclude this section, we must acknowledge that the impact of displacement of guillemots and razorbills by offshore wind farms is uncertain. However, we do know that natural mortality of adult guillemots and razorbills (including impacts of existing human activities) is very low (6% and 10% per annum respectively), and that displacement of guillemots and razorbills by offshore wind farms is likely to be incomplete, and may reduce with habituation, and that offshore wind farms may in the long term increase food availability to guillemots and razorbills through providing enhanced habitat for fish populations. This suggests that impacts of displacement from offshore wind farms are unlikely to represent levels of mortality anywhere near to the 6% or 10% total annual mortality that occurs due to the combination of many natural factors plus existing human activities. In general, seabirds achieve higher body condition during the non-breeding season than they do while breeding, and the ecology of guillemots and razorbills suggests that density-dependent competition is highly likely during the breeding season, but is less likely to occur in the nonbreeding period. On that basis, it is unlikely that displacement by offshore wind farms would result in an additional mortality exceeding 1% of

displaced birds, and any impact is more likely to be close to zero. Assuming that 1% of displaced birds might die as a consequence of displacement would appear to be highly precautionary. In addition, strong evidence for density-dependent limitation of breeding numbers of guillemots and razorbills suggests that a small increase in winter mortality would have little influence on the size of the guillemot and razorbill populations because they are likely to be at carrying capacity set by breeding habitat suitability.

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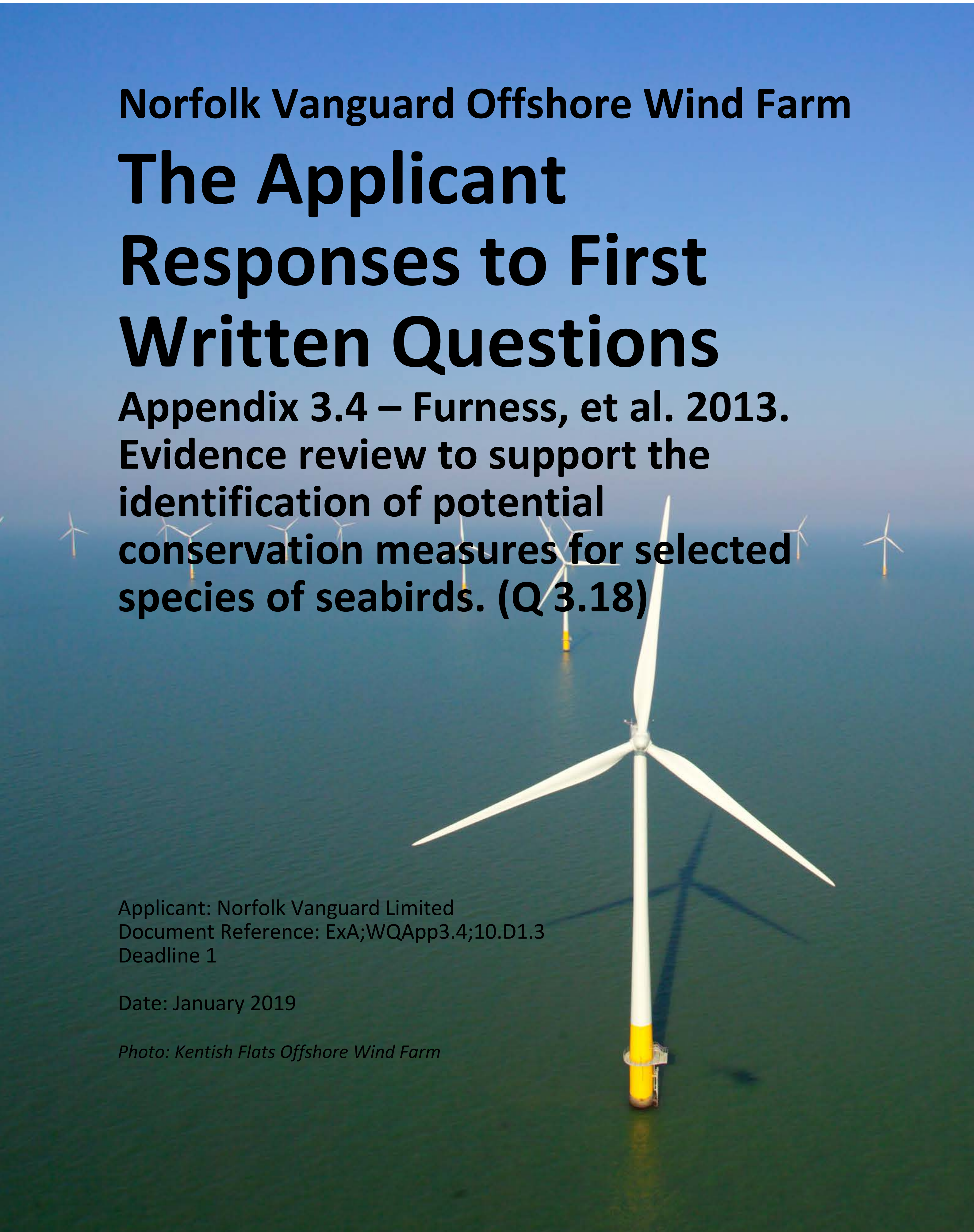

Norfolk Vanguard Offshore Wind Farm

The Applicant Responses to First Written Questions

Appendix 3.4 – Furness, et al. 2013. Evidence review to support the identification of potential conservation measures for selected species of seabirds. (Q 3.18)

Applicant: Norfolk Vanguard Limited
Document Reference: ExA;WQApp3.4;10.D1.3
Deadline 1

Date: January 2019

Photo: Kentish Flats Offshore Wind Farm

EVIDENCE REVIEW TO SUPPORT THE IDENTIFICATION OF POTENTIAL CONSERVATION MEASURES FOR SELECTED SPECIES OF SEABIRDS

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ACRONYMS

AEOI	Adverse effect on integrity (of an SPA)
AOBs	Apparently Occupied Burrows (census unit, roughly equivalent to 'breeding pairs')
AONs	Apparently Occupied Nests (census unit, roughly equivalent to 'breeding pairs')
AOSs	Apparently Occupied Sites (census unit, roughly equivalent to 'breeding pairs')
B_{lim}	Limit of stock biomass below which fishing should be closed to protect stock survival
DEFRA	Department for Environment, Food and Rural Affairs
DDT	Dichlorodiphenyltrichloroethane
EU	European Union
FAO	Food and Agriculture Organisation (of the United Nations)
HRA	Habitat Regulations Appraisal
ICES	International Council for the Exploration of the Seas
IROPI	Imperative reasons of Overriding Public Interest
JNCC	Joint Nature Conservation Committee
MANOVA	Multivariate Analysis of Variance
MIEU	Major Infrastructure and Environment Unit (of DEFRA)
MPA	Marine Protected Area
NAO	North Atlantic Oscillation
Natura 2000	Network of SPAs and SACs in EU member states
NGOs	Non-governmental organisations
NNR	National Nature Reserve
PCBs	Poly-chlorinated biphenyls
PFOW	Pentland Firth – Orkney Waters strategic area for development of wet renewables
POPs	Persistent Organic Pollutants
RSPB	Royal Society for the Protection of Birds
SCR	Seabird Colony Register
SNCBs	Statutory Nature Conservation Bodies
SPA	Special Protection Area
SPEC	Priority species (of birds) of European conservation concern, as defined by the EU
SST	Sea Surface Temperature
WECs	Wave Energy Converters
WWT	Wildfowl and Wetlands Trust

EXECUTIVE SUMMARY

MacArthur Green has been commissioned by CEFAS to produce an '*evidence review to support the identification of potential conservation measures for selected species of seabird* (from here on referred to as 'the Project'). The focus of the evidence review was on identifying measures that could be implemented either at protected sites or elsewhere with a view to informing considerations around the mitigation or compensation of predicted impacts from offshore marine developments. The review did not include mitigation measures that could be implemented at the site of offshore marine developments.

The evidence review identifies measures that could theoretically be applied to conserve seabirds where there are concerns about the potential impacts from marine developments and considers the evidence that is available to support the consideration of these measures. The measures identified raise a range of legal, administrative and policy issues. Their use in relation to any specific project or programme of marine development needs to be considered in a subsequent step based on the design of the project or programme and its specific environmental context.

The main factors affecting seabird population trends in the British Isles are food abundance, fisheries, predatory mammals at seabird colonies, and climate change. Both productivity and survival can be affected, with impacts on survival being especially influential but variation in productivity being high for all the species of concern except northern gannet (which has consistently high productivity).

Breeding populations of many seabird species increased during the 20th Century. However, in the last few years (particularly since 2000) populations of most seabirds in the UK have declined (including Manx shearwater, Arctic skua, lesser black-backed gull, herring gull, great black-backed gull, and kittiwake). Breeding populations of red-throated divers, Sandwich terns, common terns, common guillemots and razorbills have remained stable though have declined in Scotland and increased in England and Wales. Great skuas have declined at large colonies but increased at small colonies. Gannet numbers have continued to increase. Future trends are likely to be continued decreases in the numbers of most species. Given the EU objective to ban discarding by fishing vessels it is likely this will particularly affect those scavenging seabirds currently dependent on fishery discards.

Several management options to increase seabird productivity or survival, or both, have been identified in this evidence review. However, the scope for effective management varies among species. The most cost-effective, evidence-based management options for the species discussed in this report are:

- Provision of nest platforms for **red-throated divers**;
- Eradication of alien invasive mammal predators on islands with **Manx shearwater** colonies;
- Supplementary feeding of breeding pairs of **Arctic skuas**;
- Cessation of culling of breeding **lesser black-backed gulls**, **herring gulls**, and **great black-backed gulls**, and predator-proof fencing around mainland colonies of these gulls subject to fox predation;
- Closure of sandeel and sprat fisheries in UK waters to increase productivity and survival of **kittiwakes**, **common guillemots**, **razorbills, and Atlantic puffins**;
- Predator-proof fencing to exclude foxes from affected **Sandwich tern** colonies and engineering (every few years if necessary) to reduce risk of tidal flooding where this is a problem; and,
- Eradication of mink from islands with **common tern** colonies and deployment of predator-proof nesting rafts.

For gannets, scope to increase productivity or survival appears very limited, but includes a reduction in seabird by-catch by fisheries. The review also identified as a theoretical option that ending the legal harvest of chicks at Sula Sgeir would make a small contribution. For great skuas, any practical measures to increase productivity or survival appear very limited, although closure of sandeel and sprat fisheries may have a small beneficial effect, while supplementary feeding at colonies would be likely to be effective, but rather impractical. While presenting these single-species management options, we consider that interventions should aim to ensure functioning ecosystems as the highest priority. We therefore point out that some measures that could be implemented may benefit many seabird species rather than just a single species (e.g. closure of sandeel fishing), and we recommend consideration of the fact that great skua numbers (and those of some other scavenging seabird species such as herring gull, great black-backed gull and lesser black-backed gull) have been artificially elevated by human actions in the past (especially discarding by fisheries), and so numbers at SPAs at the time of designation represent elevated rather than sustainable population sizes, so desirable target population sizes may require further consideration by SNCBs.

SPECIES	**OPTION**
Red-throated Divers	Provision of nest platforms
Manx shearwater	Eradication of alien invasive mammal predators on islands with Manx shearwater colonies.
Arctic skuas	Supplementary feeding of breeding pairs.
Lesser black-backed gulls, herring gulls, and great black-backed gulls	Cease culling of breeding lesser black-backed gulls, herring gulls, and great black-backed gulls, and predator-proof fencing around mainland colonies of these gulls subject to fox predation.
kittiwakes, common guillemots and razorbills	Closure of sandeel and sprat fisheries in UK waters to increase productivity and survival of kittiwakes, common guillemots and razorbills.
Sandwich tern	Predator-proof fencing to exclude foxes from affected Sandwich tern colonies and engineering (every few years if necessary) to reduce risk of tidal flooding where this is a problem.
Common tern	Eradication of mink from islands with common tern colonies and deployment of predator-proof nesting rafts.
Gannets	Scope to increase productivity or survival appears very limited, but a reduction in seabird by-catch by fisheries would make a small contribution as might ending the legal harvest of chicks at Sula Sgeir.
Great skuas	Any practical measures to increase productivity or survival appear very limited, although closure of sandeel and sprat fisheries may have a small beneficial effect, while supplementary feeding at

	colonies would be likely to be effective, but rather impractical. Reduction in seabird by-catch by fisheries may also be useful for this species, but evidence base for this is very limited.

1. INTRODUCTION

MacArthur Green has been commissioned by CEFAS to produce an '*evidence review to support the identification of potential conservation measures for selected species of seabird'*(from here on referred to as 'the Project'). The focus of the evidence review was on identifying measures that could be implemented either at protected sites or elsewhere with a view to informing considerations around the mitigation or compensation of predicted impacts from offshore marine developments. The review did not include on-site mitigation measures that could be implemented at the site of developments. In the case of offshore wind these have been addressed by a recent and complementary Defra R&D report (Cook et al. 2011), but we do note in particular that recent studies show that most seabirds fly low over the sea so that raising turbine heights could potentially reduce collision risk for several species (Cook et al. 2012).

The assessment of adverse impacts on populations of certain bird species is potentially the most imminent barrier to the consenting of offshore wind energy development. The Habitats and Wild Birds Directives Marine Evidence Group and the Defra Major Infrastructure and Environment Unit (MIEU) have identified the need to collate information that would deepen technical and legal understanding of what might constitute compensation or mitigation (to reduce or avoid AEOI) measures for the impacts of birds from offshore wind. The evidence review identifies measures that could theoretically be applied to conserve seabirds where there are concerns about the potential impacts from marine developments and considers the evidence that is available to support the consideration of these measures. The project has dealt solely with technical aspects and makes no judgement about the legal position in respect of any of the measures identified.Their use in relation to any specific project or programme of marine development needs to be considered in a subsequent step based on the design of the project or programme and its specific environmental context.

The project has informed work by the MIEU to agree an outline definition of what constitutes mitigation and compensationby enabling a greater practical understanding of what measures are feasible and the effectiveness of each potential measure will feed into further consideration by the MIEU of the legal viability of particular options.

Given the potential for uncertainty in the interpretation of the terms 'mitigation' and 'compensation', we have minimised the the use of those terms in this report, and instead discuss management measures that aim to increase the survival rates or productivity of seabird populations.

2. AIMS AND OBJECTIVES

The CEFAS tender document (Ref: C5718I) for this project set out the following aim for this project : '*collate information to support the identification and development of potential measures that could be used to compensate or mitigate for the impacts of offshore developments on key bird species where there is an imminent consenting risk (i.e. where, on the basis of available evidence, assessments of proposed developments may predict an adverse effect on the integrity of sites designated for their protection or it may not be possible to ascertain that there will be no such effect)*'.

This was done by carrying out an evidence review that collates information to support the identification and development of measures to enhance survival and reproduction of key seabird species which may be affected by offshore wind farm developments where there is an imminent consenting risk (as defined above). The initial results of the evidence review (the draft report)were presented, by MacArthur Green, at a workshop involving regulators, advisers, industry and NGOs in May 2013. The main focus of the workshop was to test and understand the measures proposed. The findings were then revised (final report) taking into account the outcomes of the workshop where appropriate.

To inform the focus of the work, an initial brief review is presented based on up-to-date literature, on the most important factors that adversely affect seabird survival and productivity. This initial review considers all species of seabirds with a focus primarily on the populations in the British Isles, but considers literature from all parts of the world where that is informative. Following on from that initial review, the focus is on the key seabird species.

The key seabird species present on SPAs which are considered to be most at risk of adverse effects as a result of offshore wind farms developments are:

- Red throated diver *Gavia stellata*,
- Northern gannet *Morus bassanus*,
- Manx shearwater *Puffinus puffinus*,
- Arctic skua *Stercorarius parasiticus*,
- Great skua *Stercorarius skua*,
- Lesser black-backed gull *Larus fuscus graellsii*,
- Herring gull *Larus argentatus*,
- Great black-backed gull *Larus marinus*,
- Black-legged kittiwake *Rissa tridactyla*,
- Sandwich tern *Sterna sandvicensis*,
- Common tern *Sterna hirundo*,
- Common guillemot *Uria aalge*, and
- Razorbill *Alca torda*
- Atlantic puffin *Fratercula arctica*.

3. REPORT STRUCTURE

This report considers factors affecting seabird populations, especially within the British Isles (Section 4), introduces the use of population modelling to quantify the impact of changes in survival rates and productivity on seabird population trends (Section 5), and considers implications of life history ideas for management (Section 6).

For each of the key seabird species the following are considered in turn; the status and ecology of the species in the British Isles, recent and likely future population trends, factors affecting survival rates, breeding success of monitored populations in the British Isles, and the most promising management options that could increase survival rates or productivity of each species.

The report is structured to present evidence for each individual species separately in Sections 7 (red-throated diver) to 19 (razorbill). While there is considerable repetition across these species accounts sections, each one comprises a full account for that species in order to minimise the need for frequent cross-referencing within the report, and since it seems likely that only one or at most a few of the single species accounts will be of particular concern at some future date and should therefore each be complete in itself. Section 20 presents recommendations for management actions. We note that while the demography of seabirds means that measures to increase survival rates are likely to have a more powerful influence on population trend than measures to increase productivity, the evidence base to support decision making in relation to measures that affect survival is very weak, whereas there is a very strong evidence base to support decision making in relation to measures to increase productivity. Therefore, any evidence-based approach must focus on measures affecting seabird productivity even though these are likely to be less powerful than measures that increase survival rates.

4. GENERAL REVIEW OF FACTORS AFFECTING SEABIRD POPULATION SIZES WITH PARTICULAR REFERENCE TO THE BREEDING SEABIRDS OF THE BRITISH ISLES

4.1 Introduction

This review considers the consensus views in the scientific literature as to what factors primarily affect the trends in seabird population size, with particular reference to seabirds in the British Isles. This overview is intended as an introduction to the topic, before considering specific factors that might provide opportunities to carry out management actions that would recompense potential impacts of offshore wind farms on demographic trends in seabird populations.

Studies of seabirds have identified a large number of factors that affect seabird demography, and hence influence population trends, in the British Isles. These were listed by Mitchell et al. (2004) as:

- historic exploitation and persecution in the British Isles;
- current exploitation and persecution in the British Isles;
- exploitation and persecution of British and Irish seabirds while they are abroad;
- bycatch in nets;
- bycatch on longlines;
- collision with wind turbines;
- mammalian predation;
- avian predation;
- avian diseases and natural toxins (including botulism, puffinosis, ticks, red tides);
- food availability;
- effects of fisheries on food availability (through depletion of fish stocks, increases of non-target fish stocks and through provision of offal and discards);
- effects of offshore development on food availability;
- effects of refuse management on food availability;
- effects of farming on food availability;
- loss of nesting habitat;
- provision of new nesting habitat;
- oil pollution;
- persistent organic pollutants;
- heavy metal pollution;
- plastic pollution;
- global climate change;
- large-scale atmospheric and oceanographic events (e.g. El Niño, North Atlantic Oscillation); and,
- short-term weather events (e.g. storms).

Some of these factors affect only certain kinds of seabirds depending on their particular ecology, while some affect most or all species. Some have major impacts on demography of many species, while some have little or no detectable impact on most species.

Compared to most other kinds of birds, adult seabirds are exceptionally long-lived, with low reproductive output and late maturity (many species do not start to breed until three to eight years old, and many lay only a single egg which has a relatively low probability of survival to adult status). These demographic features of seabirds lead to low inherent population growth rates and generally poor abilities to recover from factors which reduce populations, particularly if these result from additional adult mortality. Consequently, any attempt to recompense for increased adult mortality by enhancing reproductive output will be constrained by both the limited extent to which low rates of reproductive output can be increased and its smaller relative contribution to population growth.

Thus, any management action that reduces mortality of adults is likely to have a greater influence on seabird population trajectory than action aimed at enhancing reproductive output, although the latter may still be effective even with long-lived seabirds with especially low reproductive rates (Finkelstein et al. 2010).

Compared to other types of birds, seabirds might be expected to have populations that remain relatively stable over periods of years or decades, because adult survival rates tend to be high, reproductive output tends to be low, and birds do not start to breed until several years old. So there is little scope for rapid increase in numbers, and unless mortality rates increase above 'normal' levels, numbers would not be expected to decrease rapidly. Nevertheless, many seabird breeding populations in Britain and Ireland have changed in numbers very considerably over the past 100 years, with a general trend for most species to have increased from 1900 to about 1990 to 2000, but in many cases to have declined since reaching a peak in breeding numbers late in the 20th Century; methods to census breeding seabirds are well developed, and counts of colony size are now fairly accurate for most species, though are less reliable for nocturnal and burrow-nesting seabirds (Mitchell et al. 2004, Forrester et al. 2007, Mitchell and Daunt 2010, Foster and Marrs 2012).

Identifying causes of population increase or decrease is difficult, as the response time of seabird populations tends to be slow. Any factor reducing breeding success, for example, will not be evident in terms of breeding numbers for several years, and may be delayed further by the buffering effect of a pool of non-breeders waiting to recruit into the breeding population (Klomp and Furness 1992). Furthermore, processes of emigration and immigration can also buffer local or even regional scale impacts on seabird demography. To compound this problem, counts of breeding numbers of seabirds at individual colonies do not necessarily reflect changes in numbers in the region as a whole as birds may move between colonies for various reasons such as local impacts of predators (Jennings et al. 2012), although such movements, which are typical of terns, may not occur in some other seabird species. Since national surveys of breeding seabird numbers tend to be made only every 10 to 15 years, and include a considerable inaccuracy in survey data, changes in breeding numbers may not become evident until several decades after the factor causing the change had its effect. Furthermore, in most cases, changes in population size are normally defined as changes in numbers of breeding pairs rather than from changes in total numbers in the population. These two metrics are not necessarily closely correlated. For example, breeding numbers could decline dramatically because increasing proportions of the population choose not to breed, while total numbers in the population may remain relatively stable. Such patterns appear to occur especially in terns where birds may opt not to attempt to breed when conditions are poor (Monaghan et al. 1989), but may apply to many other seabird species when conditions are extreme (Mavor et al. 2006).

Mitchell et al. (2004) suggested that the factors that most affected seabird populations in Britain and Ireland in the past were historic exploitation and persecution, exploitation of British and Irish seabirds abroad (i.e. outside the breeding season when the birds migrate), mammalian predation (especially involving introduced alien mammals), food availability, and fisheries. They also suggested that current population trends were most likely to be influenced by food availability, fisheries and climate change. Seabird researchers across nine nations identified the 20 highest priority global research questions regarding recent seabird declines and grouped them into six categories: population dynamics, spatial ecology, tropho-dynamics, fisheries interactions, response to global climate change, and management of anthropogenic impacts (Lewison et al. 2012). These six categories are all consistent with, or at least related to, the factors affecting seabirds covered in this review.

In this review, evidence is presented for each of the main factors affecting seabird populations, with particular reference to populations in the British Isles but taking global examples when these are informative and provide particularly clear examples. This initial introduction considers all species of seabirds, and not just the subset considered to be potential targets for management action to recompense impacts of offshore wind farms.

4.2 Exploitation and persecution

Mitchell et al. (2004) suggest that persecution and exploitation of seabirds during the 19th Century and the cessation of this during the 20th Century is '*likely to be at least partially responsible for the increases in most seabirds species observed between the 1930s and the mid-1980s in Britain*'. The same authors also suggest that current levels of persecution and exploitation in Britain can affect rates of population change of some species; culling of large gulls, shooting of great cormorants *Phalacrocorax carbo* under licence to protect freshwater fisheries (see also Smith et al. 2008), illegal shooting of skuas in some parts of Scotland, and the traditional harvest of northern gannet chicks on Sula Sgeir may all have reduced population growth rates or increased declines of particular populations. However, these effects appear to be less influential and also much more local than the effects of factors such as changes in food supply or climate change. Exploitation of British breeding seabirds in winter when they are in their wintering areas, or during migration, may influence breeding numbers of roseate terns *Sterna dougallii* in Britain (Mitchell et al. 2004). Although large numbers of auks have been shot on the coast of Norway and these include birds from British colonies, there is no evidence to indicate that this harvesting has affected numbers at British colonies (Mitchell et al. 2004). Overall, exploitation and persecution seem unlikely to be significant factors affecting current seabird numbers in the British Isles, except locally in a few cases.

4.3 Food supply

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

Mitchell et al. (2004) conclude that seabird demography in Britain and Ireland is '*strongly affected by the availability of food*'. Many breeding seabirds feed primarily on small schooling pelagic fish. These fish are important food because they tend to be abundant, available in the upper layers of the sea, have a high energy density, and are relatively small so are easy for seabirds to catch and swallow. In many different parts of the world and for many different kinds of seabirds, breeding success shows a strong sigmoidal correlation with the abundance of their preferred prey fish (Cury et al. 2011). Cury et al. (2011) identified a threshold of one third of the long-term maximum prey biomass of forage fish abundance. Below this stock biomass levelmany seabird species suffer from reduced and more variable productivity which is likely to lead to population decline if sustained. Food availability of preferred prey species varies across the oceans. For example, comparison of flight time data from geolocation loggers on wintering northern gannets suggested that food availability at this time is much more varied in the north of the range (Bay of Biscay, Celtic sea) but more consistent off Western Africa (Garthe et al. 2012). In waters around the British Isles, sandeels *Ammodytes marinus,* sprats *Sprattus sprattus,* and juvenile herring *Clupea harengus,* are preferred food for most seabirds in summer (Mitchell et al. 2004). In winter, a few deep-diving seabirds continue to feed on sandeels but because sandeels spend most of the winter buried in the sea bed, they are unavailable to most seabird species; seabird diets in winter tend to be more diverse than in summer, with less focus on small pelagic fish. In northern waters such as around Shetland, there are no sprats and no juvenile herring, so seabirds depend strongly on sandeels in summer and tend to move away from the area to spend the winter elsewhere. In coastal areas further south there may be sprats and young herring as well as sandeels, providing seabirds with a more stable prey base.

Several species' breeding success in Shetland, including that of the black-legged kittiwake (Votier et al. 2008), Arctic tern and Arctic skua*,* shows strong correlation with sandeel stock biomass (Furness, 2002). A potential minimum acceptable stock biomass (known as B_{lim}) has been put forward for seabirds (specifically kittiwakes and Arctic skuas) as a total stock biomass of 30,000 tonnes of sandeels in the Shetland stock (Furness, 2007). Below this, sandeel-dependent seabirds would be predicted to suffer from reduced breeding success, and hence potentially from population declines. One such example from the North Sea caused breeding failures for a variety of seabirds in 2004, after sandeel landings (reflecting stock biomass) decreased by over 50% between 2003 and 2006 (Frederiksen et al. 2006). In 2005 at North Sutor, North Scotland, black-legged kittiwakes suffered from complete breeding failure. Along with other species' low levels of breeding success, this decline was suspected to be linked to a shortage in sandeel abundance at the time (Mavor et al. 2005). On the Isle of May, there was a positive correlation between seabird breeding productivity and the size of sandeel prey (both in terms of individual fish size and stock biomass) (seen for Atlantic puffin, shag, common guillemot, razorbill and kittiwake; Frederiksen et al. 2006). Another study on the Isle of May kittiwake colony in relation to the opening and closure of the Danish sandeel fishery operating locally showed that breeding productivity was significantly reduced while the fishery was active, reducing the availability of the kittwake's preferred prey (Frederiksen et al. 2008).

Overall it is evident that particular species of seabirds in northern Scotland, namely black-legged kittiwakes, Arctic terns, Arctic skuas and Atlantic puffins have a strong prey preference for sandeels and hence are vulnerable to changes in their abundance. A few species of seabirds on the other hand appear unaffected by sandeel stock biomass: gannet breeding success in northern Scotland shows no correlation with the availability of sandeels. Although they will feed on sandeels when available, when sandeel abundance is low they switch to alternative prey such as adult herring or mackerel, fish that are too large for most other seabird species to swallow.

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

The impact of food availability on adult survival is potentially more influential on population trend than the impact on breeding success. Similar relationships between food supply and seabird population size have been reported further afield. For example, off South African's Western Cape, swift tern *Sterna bergii* population numbers track the abundance of their prey; anchovy *Engraulis capensis* and sardines *Sardinops sagax* (Crawford 2009). In Japan, the availability of anchovy *Engraulis japonicus* and sandeel *Ammodytes personatus* prey was recorded to affect seabird chick diet, growth rate and breeding success of the rhinoceros auklet *Cerorhinca monocerata*, Japanese cormorant *Phalacrocorax filamentous* and black-tailed gull *Larus crassirostris* (Watanuki et al. 2012). In the Firth of Forth, common tern breeding numbers at individual colonies show rather different trends over recent decades, but the regional population size correlates with the abundance of sprats (their main breeding season food) in the area (Jennings et al. 2012). The variable dynamics of individual colonies seems to be driven by predation impacts and presence of gulls, with terns moving between colonies in response. Overall breeding numbers in the region varied much less than numbers at individual colonies. On the east coast of England and Scotland, large kittiwake colony "clusters" were associated with aggregations of sandeels whereas on the west coast, these predator-prey dynamics were not as apparent. The study showed that regional variation in prey abundance has a stronger impact on kittiwake populations than local prey depletion (Frederiksen et al. 2005, Fauchald et al. 2011). Harris et al. (2010) used geolocators to identify wintering areas of breeding adult puffins from the Isle of May. Mortality rates of puffins at this colony had increased and the authors concluded that increased mortality related to reduced food availability in the wintering areas used by these birds in the North Sea, and changes in their migration behaviour. This, and other studies, indicate a complex relationship between food, survival and productivity. Birds unable to find sufficient food during winter may not be in adequate body condition in spring to sustain successful breeding, so that food shortage may have complex 'carry-over' effects on productivity as well as migration behaviour and overwinter survival.

Not only are seabird populations affected by the amount of prey available, some can be vulnerable to changes in the prey age-class structure. This highlights how specific some seabird's reliance can be upon a relatively unpredictable food source. An unproductive breeding season for Atlantic puffins at St Kilda in 2006 was correlated with a low availability of the appropriate age class of sandeel (Mavor et al. 2006). The breeding success of common guillemots, razorbills and shags in the North Sea has shown a positive correlation to sandeel growth rates (Burthe et al. 2012). In Shetland and on the Isle of May, a positive correlation between 0-group sandeel abundance and adult survival of kittiwakes is apparent (Oro and Furness, 2002, Wanless et al. 2007). Food abundance can affect a wide range of demographic parameters and even such biometrics as egg size. Decreases in puffin egg size at colonies in Norway and Scotland have been related to effects of reduced food fish abundance (Barrett et al. 2012).

Food availability can affect the foraging ranges of seabirds, with decreased levels of food increasing the distance that birds will travel to feed. A study of breeding northern gannet colonies around the UK concluded that there was a positive correlation between population size and mean foraging trip duration (Lewis et al. 2001). Larger populations of seabirds increase competition for food and hence at larger colonies, birds will have to travel further to obtain food, depleting energy stores and potentially leaving nests unattended during the breeding season.

Great skua migratory routes were studied to identify changes over time and Scottish skuas were recorded to winter off northwest Africa, much further south than previously thought. It was suggested that this migration further south could be linked to an increase in fishery discards and increasing pelagic fish stocks in the area (Magnusdottir et al. 2012). The survival of Scottish adult common guillemots has been correlated for colonies which share wintering areas, suggesting that some environmental factor present at these shared wintering sites is affecting adult guillemot survival (Reynolds et al. 2011). Perhaps the most likely common factor affecting adult survival is food availability.

4.4 Fisheries

Fisheries can affect seabird populations in several ways, either directly, as for example through bycatch of seabirds in fishing gear and though provision of discards eaten by scavenging seabirds, or indirectly though influences on ecosystem functioning and fish community composition. Fisheries for the small pelagic fish that seabirds tend to target as preferred food can reduce food availability and so can cause breeding failures of seabirds (Wagner and Boersma 2011). However, fisheries for predatory fish may alter food web structure such that small pelagic fish stocks increase, and so trawl fisheries reducing predatory fish biomass can benefit seabird populations.

Fisheries can also alter food availability to seabirds through the provision of offal (fish guts) and discards (whole fish rejected as beyond quota or too small or not worth taking to market and thrown back at sea). Most discards tend to come from bottom trawl fisheries. This supply of food that would otherwise be inaccessible to seabirds (because the fish involved are generally much too big to be swallowed by the smaller species which are able to dive to the sea floor) can increase numbers of scavenging seabirds. However, reductions in the amounts of offal or discards can then lead to large scavenging seabirds such as great skuas, great black-backed gulls, herring gulls and lesser black-backed gulls to predation on smaller seabird species, adding a novel predatory impact onto small seabird populations in their vicinity (Votier et al. 2004).

4.4.1 Fishery bycatch

Seabird mortality through bycatch from fisheries can have a strong impact on seabird populations in certain parts of the world. In the UK, long-line fishing has been a cause of bycatch mortality in gannets, great skuas and northern fulmars (Dunn and Steel, 2001), while set gill nets have drowned large numbers of auks in the past. However, according to Mitchell et al. (2004) changes in food availability may be involved in declines in fulmar numbers in Shetland, and the role of longline mortality is unclear. Fisheries bycatch was also a named potential factor causing the decline in adult survival of Yelkouan shearwaters *Puffinus yelkouan* between 1969 and 1994 in Malta, (Oppel et al. 2011). The decline of common guillemots recorded during the 1980s at a Low Arctic colony in Newfoundland was considered to be associated with bycatch drowning from gillnets which overwhelmed any impact from climate change (Regular et al. 2010). The by-catch of albatrosses and petrels in the North Pacific and Southern Ocean is currently a major problem caused by long-line fisheries (Tasker et al. 2000). This is an example where major gains can be made by mitigation measures to reduce bycatch mortality. However, although bycatch mortality of seabirds due to fisheries can be an important issue in some parts of the world, it is apparently only a minor influence for seabird populations breeding in Britain. According to Mitchell et al. (2004) '*studies of bycatch mortality in Britain showed that large numbers of auks may be caught and drowned in these nets but the rates were insufficient to cause local population declines*' and '*mortality in nets outside British waters during winter was insufficient to cause population declines of auks in Britain and Ireland*'. There is a small amount of evidence to show that gannnets and great skuas occasionally get caught as fisheries bycatch. While not likely to contribute, in and of itself, to any population decline due to the (apparently) small numbers being caught, there is the potential to minimise this threat to adult survival within European waters through the EC-PoA for Seabirds, which is specifically addressing the issue of seabird bycatch in European waters and among European fleets. While such efforts are to be welcomed, as any reductions in adult mortality reduces pressure on declining populations, reductions in fisheries bycatch (for this species at least) could not be recommended as sufficient compensatory mitigation for the impacts of OWFs as the evidence base on the magnitude of this bycatch is deficient. However, future quantification of bycatch may indicate that there is potential to compensate by measures that reduce the bycatch. Recent research suggests that the bycatch of some UK seabirds may be much higher than previously thought. For example, it is estimated that the Gran Sol fishery accidentally caught 1,331 gannets per year during 2006/07 (BirdLife 2009), while recent unconfirmed reports from West Africa (a core winter ground for some UK seabirds including gannets and great skuas) indicate large numbers being caught by fisheries, with some of these being found in refrigerated containers destined for the Far East http://seabirds.net/posts/2013/02/13/evidence-for-massive-bycatch-in-chinese-fisheries/ .

4.4.2 Fishing of pelagic prey fish stocks

Although there are many examples of seabird breeding success relating to pelagic fish abundance, there is often dispute as to how much the abundance of pelagic fish is determined by fishing and how much variation is due to natural factors. The presence of sandeel fishing on the Wee Bankie, E Scotland, has been clearly correlated with low and variable kittiwake breeding success and reduced adult survival, whereas before the fishery opened (in 1990), breeding success at the Isle of May colony was much higher (Frederiksen et al. 2004, Scott et al. 2006). However, this relationship was complicated as there was also an influence of sea temperature (i.e. global climate change impact) in addition to the influence of presence or absence of a sandeel fishery. After closure of the sandeel fishery in this area in 1998, breeding success of kittiwakes within the fishery area improved

(Frederiksen et al. 2004) and matched that in a control (unfished) area (Frederiksen and Wanless 2006). This one case study therefore provides evidence that closing a fishery can improve food supply for seabirds. However, although there was also evidence of an improvement of breeding conditions for Sandwich terns, there was no significant change in breeding success of other species (such as auks) (Frederiksen and Wanless 2006).

In Shetland, although the relationships between sandeel stock biomass and seabird breeding success are clear, it is uncertain whether the decline in sandeel abundance in the late 1980s and since 2000 is due to fishing impacts or whether it is due to natural factors or to climate change. There is, for example, some evidence to suggest that the declines in sandeel abundance in the Shetland stock may have been influenced by top down predation impacts due to recovery of adult herring biomass in the area in the late 1980s and after 2000 (Frederiksen et al. 2007). Examples where impacts of the fishery on pelagic fish can be more clearly seen include the collapse of seabird populations in Namibia and parts of South Africa following depletion of sardine and anchovy stocks by overexploitation (Pichegru et al. 2010a, Cury et al. 2011) and the mass mortality of seabirds in Peru following depletion of the anchoveta *Engraulis ringens* stock by fishing (Wagner and Boersma 2011). The fact that closing fisheries around African penguin colonies can lead to rapid recovery of breeding success and numbers (Pichegru et al. 2010b) also provides clear evidence of a fishery impact on seabird prey abundance in that ecosystem. Changes in breeding numbers of seabirds resulting from changes in fish abundance driven by fisheries can be dramatic. For example, African penguin numbers fell by over 90% when sardines were overfished in Namibia (Cury et al. 2011), Atlantic puffin breeding numbers at Røst, Norway, fell by 50% after the herring stock was depleted by fishing (Gjøsæter et al. 2009, Cury et al. 2011). Richerson et al. (2010) showed by modelling seabird-fishery interactions that a 20% reduction in sandeel harvest from Shetland could potentially double breeding success of Arctic terns in Shetland, by allowing sandeel stock biomass to recover from the depletion caused by a fishery. However, in reality there has been no commercial sandeel fishery at Shetland since 1990, and stocks of this species have not recovered, indicating that such theoretical predictions do not necessarily match up with empirical observation. Other factors may subsequently have prevented the sandeel stock from recovering to levels that were present before the fishery of the 1970s and 1980s. Bertrand et al. (2012) showed that seabird foraging can be affected by fishery depletion of pelagic fish within a few days, with seabirds being forced by fishery depletion of local stocks to travel further to search for food.

4.4.3 Discards and offal

Although there are efforts currently in place to reduce the amount of fish being discarded from vessels at sea (Bicknell et al. 2013), scavenging seabirds still consume large quantities of discarded fish, and offal (60-80% of roundfish discards and 70-95% of offal discards) (Furness et al. 2007). Great skua colonies of the northern North Sea have seen rapid population growth from early in the 20th Century up to the end of the century, and this has been correlated to increases in fishery discards from the 1940s to the 1980s (Votier et al. 2004). From a study on great skua populations around Shetland it was clear that these birds relied heavily on fishery discards as a large component of their diet (Votier et al. 2008). Similar dependence on fishery discards is apparent in large colonies of great black-backed gulls and possibly in several other gull species (Mitchell et al. 2004). As reducing fishery discards remains a top priority for FAO's policy for Responsible Fisheries (Furness 2003, Bicknell et al. 2013), it seems reasonable to predict that unless these seabird species can switch to alternative diets, their current breeding success is unlikely to be sustained.

Wintering seabirds off the Western coast of Africa (Sahara and Mauritania) have also been reported to have a close association with fishing trawlers, feeding on the discards. However, hydrography (specifically cold water upwelling) there has perhaps an equally important role to play in seabird success (Camphuysen and Van der Meer, 2005). During a study to investigate the winter migration patterns of northern gannets, it was revealed that they often stay in areas of elevated discard availability. Off western Africa, 88.8% of wintering gannets were found to be associating with fishing vessels (Kubetzki et al. 2009). As gannets are adapted for diving and catching live fish prey, it is not surprising that they are switching to discards as a food source when availability of pelagic fish is low. It has been suggested, however, that fishery discards are a poor substitute for higher energy anchovy and sardine, which gannets would preferably feed on in favourable conditions. Between 1986 and 2006, anchovy and sardine availability off the west coast of South Africa decreased from ca.84% to ca.35%, correlated with a decline in growth rate of cape gannet *Morus capensis* chicks (Mullers et al*. 2009)*. This decline in anchovy and sardine availability can be linked to the activities of a purse-seine fishery, which in 2007 took 41% of the food needed for the Malgas Island cape gannet colony (72,000 birds) (Okes et al. 2009). Although northern gannets apparently switch from feeding mainly on discards in winter to feeding more on pelagic fish while breeding, even during the chick-rearing period almost all gannets seem to forage around trawlers at times (Votier et al. 2013).

A study of Balearic shearwater *Puffinus mauretanicus* diet pre-incubation and during the breeding period showed an interesting link to the energy differences in prey species. During the pre-incubation period, breeding adults were recorded to feed mainly on demersal (discarded) fish, whereas during incubation, had a tendency to switch their foraging effort to higher energy fish such as anchovies and pilchards *Sardina pilchardus*, (this was particularly prevalent in the female adults), (Navarro et al. 2009). This prey-switching behaviour in the shearwaters could be linked to natural (seasonal) or unnatural (caused by fisheries) variations in the availability of their preferred prey. However, the increased switching amongst females would suggest that this move away from discard feeding was linked to feeding their young since discards are lower in energy (Mullers et al. 2009). Österblom et al. (2008) put forward the “junk-food hypothesis” affecting marine top predators such as seabirds. This hypothesis suggests that it is also the quality, not just the quantity of food which affects seabird population trajectories. Given that the EU Common Fisheries Policy is currently being rewritten and that it is intended that the EU will introduce a ‘no discards’ policy, scavenging seabirds in the British Isles are very likely to face a drastic decrease in food availability (Votier et al. 2013). This could result in reduced breeding success and declines in breeding numbers of great skuas, great black-backed gulls, herring gulls, lesser black-backed gulls and possibly of northern fulmars and northern gannets (Garthe et al. 1996, Furness 2003, Bicknell et al. 2013).

4.5 Climate change

Although impacts on seabirds from climate change are mostly indirect, such as increased sea surface temperature (SST) altering the marine food web from plankton upwards, they can still have strong impacts on the breeding success of seabirds (Frederiksen et al. 2013). According to Sydeman et al. (2012), seabirds are responding to climate change across the globe. When predicting climate change impacts on seabirds it is apparent that no single factor acts alone on populations, all factors (such as food availability, warming oceans or nest desertion rates) are interconnected (Heath et al. 2009). Climate change models have predicted that by the end of this century, great skua and Arctic skua*,* two widespread seabirds in Northern Scotland will no longer breed in the UK (Mitchell and Daunt 2010). Measurements using the NAO index have estimated that 29 species of North Atlantic seabirds are significantly affected by climate (Sandvik 2012).

One such example is the kittiwake which has suffered breeding success declines with increasing Sea Surface Temperature (SST), as warming oceans are also correlated with a decrease in large *Calanus* copepod abundance (this being a key food for sandeels) (Frederiksen et al. 2007). Kittiwakes in the West Atlantic are also thought to be sensitive to deteriorating environmental conditions (changes in sea temperature affecting zooplankton affecting food fish abundance), leading to population declines, (Frederiksen et al. 2012); these deteriorating environmental conditions can only become more exaggerated with climate change given the adverse effect of warming sea temperature on copepods and hence on higher trophic levels. Further north, in a much colder climate, kittwakes of the Bering shelf region however may experience short-term demographic benefits from ocean warming effects (Satterthwaite, 2012). In Nunavut, Canada, although the reproductive success of thick-billed murres (Brunnich’s guillemots) *Uria lomvia* has not yet been reduced by an increasing SST and declining ice cover, a declining rate of energy supply to chicks has been observed and this suggests that reproductive success could be affected in the foreseeable future if this trend continues (Smith and Gaston, 2012).

Auks of the North Sea (common guillemots, razorbills and Atlantic puffins) have been recorded to be breeding later as a response to climate change (Wanless et al. 2009), probably because of an effect of reduced prey availability during the laying and incubation period. One of the more indirect effects of warming oceans due to climate change is the effect upon the marine food web and most importantly at the bottom of the web, plankton. With a decrease in large *Calanus* copepods correlated to increasing SST, many small fish (such as sandeel) will have a limited food source, which as discussed above, can have seriously detrimental impacts on seabird's breeding success. It is not only kittiwakes that are affected by climate change; there is evidence of effects on many other species, but not all effects are negative. In the Alaska Gyre, the abundance of 15 species of seabirds was tested against seasonal oceanic trends to predict the potential effects of climate change in the future. Overall, there was an increase in seabird abundance (9 out of 15 species), apparently a result of an increase in forage zooplankton and a lengthening of the growing season (Thompson et al. 2012).

The extent of climate change effects on seabirds will differ across the globe. Great skuas experience heat stress above certain temperatures causing increased rates of nest desertion. Above 16°C, 10% of great skua territories were deserted by both parents leaving chicks vulnerable to predation and/or starvation (Oswald et al. 2008) at a study site in Foula, Shetland. At this same site, heat stress was recorded to have a "critical" effect on chick survival above 14°C. During a seabird review throughout the UK climate change, along with prey availability, was found to correlate with the breeding success and survival rate of kittiwake, shag, Arctic skua, Arctic tern, common guillemot and Atlantic puffin, with infanticide being recorded in guillemot colonies as a critical impact (Mitchell and Daunt, 2010). As for climate change affecting migration routes and wintering areas of seabirds, Fort et al. (2012) found that winter hotspots for northern gannets could be dictated by the origin of the bird, suggesting genetic control of migration pathways, which may influence their susceptibility to climate change.

Little auks *Alle alle* of the North Atlantic have the ability to offset potential climate change impacts (i.e. ocean warming) through plasticity of their foraging behaviour (Grémillet et al. 2012). Kittiwakes of the North Sea on the other hand have been reported to correlate their laying dates with NAO and SST in the prebreeding period, whereas common guillemots made only minor adjustments to their breeding schedule even with large-scale climate changes (Frederiksen et al. 2004). Rising sea levels linked to climate change are another concern for seabird colonies on low-lying islands or exposed coastlines, with the potential for the nest habitats of ground-nesting birds to be washed away (Mitchell and Daunt, 2010).

Anticipated climate warming in the 'Green Belt' of the south-eastern Bering Sea is predicted to reduce availability of prey for planktivorous predators (in this case least auklets *Aethia pusilla* (Dorresteijn et al. 2012). Another predicted effect of climate change is an increased frequency of hurricanes and storms. Along the US Atlantic Ocean seaboard, modelling indicates that rises in numbers of hurricanes has the potential to increase the risk of extinction of black-capped petrel *Pterodroma hasitata* (Hass et al. 2012). The impacts observed from these studies are relatively strong, affecting seabirds at a population level and the effects are long-term with predictions for even stronger effects in the future. Climate change is therefore a relatively important factor affecting seabird populations (Frederiksen et al. 2013). In contrast to the effects of fisheries however, climate fluctuation mainly affected reproduction which is typically a "low elasticity" demographic trait and, as such, has lower impact on population growth rates compared with impacts on adult survival, a "high elasticity" trait (Barbraud et al. 2012).

4.6 Predation

Predation from 'natural predators' of seabirds (such as eagles, peregrines and other birds of prey) tends to have only a small impact on breeding success or survival rates of seabird populations, and tends to occur at sustainable levels over the long term. Human impacts can alter this balance where populations of predators increase. For example, great skuas and large gulls have increased in response to provision of large quantities of fishery waste, but may then switch to killing smaller seabirds when fishery management changes discard availability (Votier et al. 2004). Predation from introduced mammals, however, can have extremely serious unsustainable impacts on seabird populations, especially on smaller seabirds, as mammals tend to only attack seabirds that are smaller in size than they are (Towns et al. 2011). For example, the presence or absence of brown rats in Orkney and Shetland is the "*single most important influence*" on storm-petrel breeding distribution (de Leon et al. 2006). Evidence of this comes from a survey reporting that European storm petrels were restricted to rat-free and low-disturbance islands (de Leon et al. 2006).

It is estimated that 75% of threatened island birds are at risk from introduced species (Phillips 2010). For example, mammal predators were responsible for the extinction of three seabirds endemic to New Zealand: a species of penguin *Megadyptes waitaha*, Scarlett's shearwater *Puffinus spelaeus* and the southern merganser *Mergus australis* (Towns et al. 2011). Since European expansion in the 16th century, invasive mammalian predators have spread through UK Overseas Territories (e.g. Bermuda, Gibraltar, Falklands, Cyprus). These invasions have caused seabird extinctions and population declines which are ongoing across the islands (Hilton and Cuthbert 2010). Similar devastating impacts have frequently been seen throughout the world where alien mammals have been introduced onto islands with seabird colonies (Towns et al. 2011). For example, Pascal (1980) estimated that on Kerguelen Island sub-Antarctic Indian Ocean, cats killed 1.2 million seabirds each year during the 1970s.

Sooty terns nested in 'great abundance' on Raoul Island in the Kermadec group. Cats reached the island in the early 19th century, followed by brown rats after a shipwreck in 1921. In 1967, 80,000 adult terns were still present, but by 1994, these had been reduced to 1,300 and carcasses of terns killed by cats littered the beach. By 1997, the tern population had been extirpated (Peck et al. 2008, Towns et al. 2011). However, there are a few examples of cases where very large seabird populations have coexisted with rats for many decades without obvious impacts (Quillfeldt et al. 2008), possibly because rat numbers in these cases are simply too low relative to the huge numbers of seabirds in the colonies. This may explain why it took over 100 years for cats to eradicate sooty terns from Raoul Island. Similarly, cats on Ascension Island reduced sooty tern numbers from more than one million pairs in the 1940s to about 150,000 in the late 1980s (Nogales et al. 2004).

Brown rat and American mink are two introduced predatory species that have caused whole colony extinctions of terns, gulls, storm petrels, Manx shearwater and Atlantic puffin at many sites in the British Isles (Mitchell and Daunt 2010). The Shiant Islands of the Outer Hebrides experienced declines in nesting seabirds in the early 1990s which was correlated to the presence of ship (black) rats, (Key et al. 1998, Stapp 2002), although those rats and the seabirds appear to have co-existed on the Shiants since 1900 (Brooke 1972). Now that the predatory impacts of invasive mammals are well known, regulations are in place regarding the accidental transport and release of such animals, to reduce the chances of them reaching currently "safe" islands (Manchester and Bullock 2000).

In the Azores, the introduction of mammalian predators led to the disappearance of Procellariform seabirds from the main islands, all except the Cory's shearwater *Calonectris diomedea,* the largest abundant seabird of that archipelago (Fonataine et al. 2011). In combination with low natal philopatry, high predation pressure on the East Limestone Island colony of ancient murrelets *Synthliboramphus antiquus* in British Columbia has the potential to eradicate this seabird species from the island (Gaston and Descamps 2011). In the British Isles, foxes, badgers, stoats, American mink, otters, rats, and feral cats have all been identified as major influences on productivity of some seabirds. Not all impacts are from alien mammals. Otters, an endemic mammalian predator resident to Scottish coastlines, were the cause of a very low breeding success rate of Arctic terns in NE Scotland in 2005 (Mavor et al. 2006). Many areas that used to be predator-free nesting habitat have been invaded by mammal predators, in some cases naturally, but in most cases as a consequence of human influences that have increased populations of these mammals or have assisted their colonisation of seabird islands. Cats and foxes have been described as 'superpredators' because they kill adult seabirds as well as taking eggs and chicks. They may kill large numbers of seabirds that they then store for subsequent meals, and their populations on remote islands often require rodents to be present to sustain them through periods when seabirds are absent (Towns et al. 2011). American mink fit into this classification too. Rodents have been categorised as 'mesopredators' (Towns et al. 2011). Their impacts are predominantly on seabird breeding success rather than on adult survival, and can increase where superpredators such as cats have been eliminated, allowing mesopredators such as rats to increase (Rayner et al. 2007, Le Corre 2008).

There are numerous examples around the world of devastating impacts of mammal predators on seabirds, and more recently of the extirpation of mammals to restore seabird habitat and endangered populations. These examples include eradication of rats (Gaze 2000, Stapp 2002, Lock 2006, Towns et al. 2006, Appleton 2007, Bell 2007, Howald et al. 2007, Luxmoore 2007, Mitchell and Ratcliffe 2007, Patterson 2007, Stoneman and Zonfrillo 2007, Swann et al. 2007, Zonfrillo 2001, 2007, Jones et al. 2008, Mulder et al. 2009, Ratcliffe et al. 2009, Bellingham et al. 2010, Capizzi et al. 2010, Howald et al. 2010, Phillips 2010, Brown et al. 2011, Dunlevy et al. 2011, Kawakami and Aoyama 2011, Mulder et al. 2011, Oppel et al. 2011), mink (Craik 2007), feral cats (Veitch 2001, Keitt et al. 2002, Nogales et al. 2004, Dowding et al. 2009, Ratcliffe et al. 2009, Bellingham et al. 2010, Kawakami and Aoyama 2011, Oppel et al. 2011), hedgehogs (Jackson 2001), mice (Wanless et al. 2007, Angel et al. 2009), rabbits (which can be predators as well as keystone species affecting habitat) (Bried et al. 2009) and other mammals (Donlan et al. 2002, Carrion et al. 2011), and the use of predator exclusion from colonies such as by deployment of predator proof fencing (Young et al. 2012). Veitch and Clout (2002) review eradication of 138 populations of seabird predators in ten countries (many of these being eradication of Arctic fox populations in Alaskan islands and eradications of various species from small islands off New Zealand). The ability to eradicate invasive alien mammals has improved with experience and development of techniques (e.g. Eaton and Ogilvie 2009, Gsell et al. 2010). Eradications have been successfully completed on increasingly larger islands, with rat eradication on islands up to 11,300 ha (Campbell Island, sub-Antarctic New Zealand), cat eradication on islands up to 29,000 ha (Marion Island, sub-Antarctic South Africa), fox eradication on islands up to 90,574 ha (Attu, Aleutian Islands, Alaska United States) (Dunlevy et al. 2011).

In cases where seabirds have been extirpated by predators, seabird restoration programmes have been developed to restore populations after their habitat has been cleared of the invasive predator population (Rauzon et al. 2002, Parker et al. 2007, Rauzon 2007, Miskelly et al. 2009, Jones 2010a,b, Lavers et al. 2010, Jones et al. 2011, Jones and Kress 2012). Jones and Kress (2011) define seabird restoration as '*efforts to actively restore seabirds through direct management interventions rather than allowing seabirds to passively recover following the removal of disturbance factors such as invasive mammals*'. They recognise two forms of seabird restoration; chick translocation, and social attraction.

Chick translocation is labour-intensive, expensive, and is successful only for species with particular life-history traits (in particular chicks that fledge independently of parents and do not receive any parental protection or feeding after fledging). These include Manx shearwater and Atlantic puffin, for example. The idea is that chicks subsequently return several years later to breed where they fledged; this works because most seabirds show very strong fidelity to their natal colony. Chick translocation can be effective (see for example Miskelly and Taylor 2004, Bell et al. 2005, Miskelly et al. 2009); a meta-analysis by Jones and Kress (2011) identified 5 successful projects out of 5 using this method alone, and a success rate of 70% for 59 projects using a combination of 2 or 3 of chick translocation, acoustic attraction and decoys (the difference in success rates not being statistically significant due to the small sample size using chick translocations alone).

Social attraction aims to lure adult birds to restoration sites where there is no colony at the time. Models of adults, sound recordings, mirrors, artificial nests or burrows may be used in some combination to encourage birds to join what appears to be an active colony. Social attraction has been used particularly successfully with auks and terns (Jones and Kress 2011). An attempt to start a new Australasian gannet colony at Young Nick's Head, in New Zealand by social attraction was successful, but attempts using the same method with Australasian gannets at Mana Island, New Zealand, and with northern gannets in Nova Scotia and in Quebec failed (Jones and Kress 2011). Jones and Kress (2011) suggest that the key feature affecting the outcomes at these sites was that the Young Nick's Head site was in an area frequented by large numbers of potential colonists from a nearby colony, while the unsuccessful attempts were at sites far from source colonies. After rats had been eliminated from the island, an attempt was made using chick translocation and acoustic playback to attract Manx shearwaters to re-colonize Cardigan Island in Wales from which they had been extirpated by brown rats, but this failed, apparently due to high numbers of gulls deterring shearwaters from re-establishing (Gummer 2003).

However, costs of bringing seabirds back to an island from which they were extirpated by alien predators is usually much greater than the cost of removing the predators, and the probability of success is less than 1, so it would be very much better to remove predators before seabird colonies are completely abandoned (Jones et al. 2011). Kress and colleagues successfully used social attraction methods to encourage Atlantic puffins and terns to re-colonise islands in the Gulf of Maine (Kress 1983, 1992, Kress et al. 2008). The seabird restoration methods developed in Maine have since been used globally to restore at least 49 species of seabirds on 89 islands in 14 countries (Jones et al. 2011). Jones and Kress (2011) suggest that a typical restoration project for a seabird in a developed country may cost around £500,000 per annum over a project lasting at least 5, possibly 10 years on average. They point out that the successful project restoring Atlantic puffins to Eastern Egg Rock in Maine took 35 years of sustained effort to establish a population of 100 pairs of puffins.

Globally, there have been positive responses for at least 45 species of seabirds following removal of mammalian predators, so the effectiveness of predator removal in allowing seabird populations to recover is very clearly established (Towns et al. 2011). For example, eradication of cats on Ascension Island in 2004 resulted in an increase in breeding numbers of sooty terns by over 50,000 pairs in less than three seasons, and resulted in immediate natural recolonisation of the island by masked boobies and brown boobies (Hughes et al. 2008, Ratcliffe et al. 2009). Removal of mink from islands in the Baltic Sea led to increases in breeding numbers of Arctic skuas, Arctic terns and common gulls (Nordstrom et al. 2003). Numbers of breeding terns, black guillemots, eiders and gulls all increased on islands in Argyll where mink were trapped (Craik 1997, 2007). Evidence for the benefits of predator control can even be found in some cases where there has been no detectable impact on breeding seabird numbers. For example, controlling or eradicating rats resulted in white-chinned petrel breeding success at Crozet increasing from 16% to 50% (Jouventin et al. 2003). Control of rats on Lavezzi Island doubled breeding success of Cory's shearwaters (Pascal et al. 2008).

While invasive alien mammals have been eradicated from 12 relatively small islands around the British Isles, there are at least 80 islands or island groups around Britain where rats are present (Ratcliffe et al. 2009). Eradications that have been carried out including brown rats on Canna Scotland (Bell et al. 2011), brown rats on Ailsa Craig (100 ha, Scotland) (Zonfrillo 2001, 2002, 2007), brown rats on Handa (100 ha Scotland) (Stoneman and Zonfrillo 2005), brown and black rats on Lundy Island (500 ha England) (Bell 2004, Appleton et al. 2006), and brown rats on Ramsey Island (256 ha Wales) (Bell et al. 2000). ,

Numerous British islands of major importance for their seabird colonies have populations of invasive alien mammals, including several SPAs for seabirds:–

- Hermaness SPA has brown rats;
- Noss SPA has had feral cats, although it has been suggested that these have now died out and indeed storm petrels have returned to breed there in the last few years (M. Bolton pers. comm.);
- Shiant Islands SPA (one of the biggest puffin colonies in UK) has black rats;
- Foula SPA (the biggest great skua colony in the world) has feral cats and hedgehogs;
- Rum NNR and SPA (the biggest Manx shearwater colony in Scotland) has brown rats, red deer and possibly feral cats; and,
- Imperial Dock Lock SPA (the biggest common tern colony in Scotland) has brown rats and American mink.

In contrast, in Mexico, 12 species of invasive alien mammals have been eradicated from 31 islands (total of 51,000 ha) providing conservation gains for 227 colonies of seabirds (Aguirre-Munoz et al. 2008). Britain and Ireland lag far behind several other countries in terms of removing alien mammals, possibly because so many islands in Britain have been invaded for such a long time that these populations are now treated as part of the environment rather than a problem that can be solved; an issue also in some other countries (Ruffino et al. 2009). However, in many parts of the world, conservation managers recognise that eradication of alien mammals represents one of the most cost-effective approaches to protecting seabird populations (Dunlevy et al. 2011). Dunlevy et al. (2011) also identify a mindset among managers in those countries where eradication programmes are well developed, that while removal of invasive species is a necessity, monitoring the recovery of native species after eradications is an optional luxury. This can lead to reluctance to support new initiatives if the outcomes of previous financial outlays are not clear. This has encouraged a shift to evidence-based conservation, where the removal of introduced alien predators has been shown to have significant benefits (Dunlevy et al. 2011).

In the UK, DEFRA (2007) recognized the threat that invasive species pose to native biodiversity and drafted a strategy that requested prioritization of remedial management. Ratcliffe et al. (2009) provided exactly that required prioritization for eradications of rats on British islands that would maximize benefits for seabird conservation. They followed guidelines of Falklands Conservation (2008) in assuming a maximum swimming distance of 300 m for brown rats and black rats, and assessed ‘eradication units’ defined as islands at least 300 m from the mainland and at least 300 m from adjacent ‘eradication units’. They also considered reintroduction risk, based on the premise that eradications would be less cost-effective if there was a high likelihood that rats would recolonize the island subsequently. Of the 12 eradications carried out in the UK, all were considered successful, but two of these sites were recolonized by rats from adjacent areas (Isles of Scilly and Looe Island). Recolonization risk increases with the size of the human population (de Leon et al. 2006, Ratcliffe et al. 2009). On this basis, they excluded from further consideration all islands with resident human populations exceeding 100 people. Ratcliffe et al. (2009) assumed a cost of eradication of £440 per hectare, based on the cost for eradication on Canna, probably the most expensive rat eradication so far carried out in the UK.

Mammals are obviously not the only predators of seabirds. Skuas feed on other seabirds; Arctic skuas mainly on young chicks but great skuas can take larger seabirds too. Their predominant prey is sandeels but in response to declines in sandeel availability and in the face of reduced rates of fishery discarding, great skuas have resorted to feeding more on other seabirds (Votier et al., 2007). To emphasise the extent of predation from skuas on other seabirds, it was recorded in 2005 that only 2 out of >300 pairs of Arctic terns fledged as a result of great skua predation and bad weather in North Ronaldsay (Mavor et al. 2006). Common terns are also strongly influenced by local predation, mainly by large gulls (Jennings et al. 2012). In North America, increasing populations of large gulls during the 20th century have been linked to declines of terns, with several tern colonies wiped out by gull depredations. Since 1997 almost all colonies of roseate terns in the United States have been managed to control predation by gulls (Whittam and Leonard 1999). Great skuas fed less on other seabirds at larger colonies than at small colonies (Votier et al. 2007). An interpretation of this finding was that at larger colonies, competition for other seabirds as prey was so intense that most of the skuas would feed on fish instead, even though foraging range would be wider.

As discussed above, following climate change, increasing temperatures (and hence a projected decline in sandeel stocks), seabird predation by great skuas (particularly affecting kittiwakes, Arctic skuas and Larus gulls in Orkney and Shetland) may well be subject to increase in the future. However, on a scale of factors affecting seabird populations, avian predators are a natural source of predation that tends to be sustainable, and alone (without the cumulative effect of climate change and fisheries management affecting skua and gull predation), have a relatively weak impact on seabird populations. Nevertheless, increased populations of gulls can have damaging effects on tern colonies in particular, and control of gull numbers near to tern colonies of conservation concern has been an effective method to increase tern breeding success (Whittam and Leonard 1999, Donehower et al. 2007).

4.7 Disturbance

Human disturbance of wild animals is often a concern. As for seabirds, ground-nesting species are obviously of greater disturbance potential than are cliff-nesters or burrow-nesters. At Alness Point, N. Scotland, a long term decline in common gull *Larus canus* numbers has been recorded, with human disturbance being the main factor reducing gull success; all nests failed in 2005 (Mavor et al. 2006). Kittiwakes and common guillemots at St. Abb's Head, East Scotland have shown reduced nesting success and even nest failure linked to human disturbance (Beale and Monaghan, 2004, 2005). In Orkney and Shetland, the presence of human visitors also appears to affect the distribution of storm petrels (de Leon et al. 2006). A study of Cassin's auklets (*Ptycoramphus aleuticus* - a ground-nesting seabird) off the coast of Mexico on West San Benito Island, has recorded human disturbance effects from a seabird colony less habituated to visitors. At this site, disturbed auklet chicks showed a lower mean peak mass than the control group of chicks left undisturbed. This showed that under strict experimental conditions, the growth rate (measured in mean peak mass) decreases with increasing disturbance. This considered, human disturbance is a relatively low impact factor on most seabird populations, being a significant hazard mainly for ground-nesting species frequently found on mainland sites (such as little tern). Although disturbance in studies cited above shows a negative correlation with both nesting success and chick growth rate, the effect of disturbance at a population level was weak compared to other factors affecting seabird success, although disturbance impacts should be considered at the site level for particularly vulnerable species such as terns and gulls at mainland colonies, red-throated divers nesting at lochans close to roads providing vehicle access for tourists. Disturbance is also a significant issue in the offshore environment, where boat traffic and offshore wind farms can disturb seabirds (which species such as divers and scoters being considered especially vulnerable).

4.8 Parasites and disease

Botulism is thought to have caused reductions in breeding numbers of large gulls where these birds regularly scavenge on refuse tips where botulism can develop (Mitchell et al. 2004). Puffinosis can cause death of large numbers of Manx shearwater chicks, with up to 4% dying in some years from this disease. However, the scale of mortality is thought not to affect breeding population trends (Mitchell et al. 2004). According to Mitchell and Daunt (2010), the detrimental impact of parasites on seabirds is increasing. However, a lack of research on parasitology of seabirds makes it difficult to gauge the importance of parasitic impact on seabird populations. The colonial nature of seabird breeding however represents an ideal niche for tick infestations. Tick infections have been recorded to reduce nestling condition, growth rates and survival of seabirds (Muzaffar and Jones 2004). Ticks as parasites will rarely kill their hosts, but can have a detrimental impact on their seabird hosts (especially when carrying viruses) which could impact the population dynamics long-term. Desertion of portions of kittiwake colonies during their breeding season has been associated with abnormally high levels of ectoparasite infestations (Boulinier and Danchin, 1996). However, these impacts are mostly not substantial enough to cause large-scale population changes among seabird colonies, although in some cases they might influence local population trends (Mitchell et al. 2004). Therefore parasites can be considered a relatively low-impact factor of seabird population change.

4.9 Oil Pollution

Whether from offshore oil platform leakages, oil spills at sea or general waste disposal, oil pollution remains a factor that affects seabirds and can kill large numbers of seabirds, especially auks. However, Mitchell et al. (2004) concluded that '*effects of large oil spills are relatively short-term and localised, with no evidence of wide-scale, long-term effects on auk population trends*'. Seabirds are described by Boulinier and Riffaut (2008) as the '*emblematic victims*' of oil pollution. It is thought that after an oil spill, when it affects adult seabird survival, population declines will occur the following year but that after the decline, populations generally recover again (Piatt and Roseneau, 1999).

Off the Dutch coast, quick post-spill recoveries have been recorded in gulls. All birds in the study managed to clean their plumage within a few weeks of the spill, in enough time to establish territories and breed that season (Camphuysen, 2011). However, a study on the effect of oil spills on adult common guillemots showed that major oil pollution incidences doubled their winter mortality rate (Votier at al. 2005).The Exxon Valdez oil spill of 1989 in Alaska caused population declines in affected common guillemot colonies and delayed breeding phenology leading to low reproductive success (Piatt and Roseneau, 1999). The Prestige oil spill, NW Spain, caused a delayed but sublethal impact on local seabirds, e.g. damage to kidneys and liver (Perez et al. 2010). In December of 1999, the Erika oil tanker spill in the Bay of Biscay resulted in the wreck of 80,000 seabirds being washed ashore, with 80% of these birds recorded as common guillemots (Riffaut et al. 2005).

These oil spills obviously had a large impact on local seabirds, particularly on common guillemots, but as the spills were rare events, alongside evidence for self-cleaning properties of some seabird species, populations had future opportunity to recover. Oil spills around the Scottish coastline are especially rare, however oil spills around the Northern Isles between December 1978 and March 1979 caused the death of 7,735 birds which were found oiled and dead on Orkney and Shetland islands (Richardson et al. 1982). Although the rarity of oil spills around Scotland presents a case for that factor to be of relatively low importance on seabird populations, it also means that the coastline is unprepared to deal with potential spills and clean up procedures, which could aggravate already serious impacts (Heubeck et al. 2003). Mortality peaks have also been recorded for foraging seabirds around offshore oil platforms as a result of ingested oil or damaged feathers (Wiese et al. 2001).

As for oil discharges from ships, along the coast of Newfoundland, Canada, illegal oil discharges were the cause of 315,000 ± 65,000 common guillemot deaths each year (Wiese et al. 2004). This emphasises the impact that oil can have on seabirds in areas where oil pollution is much more common. Although the impact upon seabird populations from various forms of oil pollution can be severe and often lethal, impacts vary greatly between species (Boulinier and Riffaut 2008) and they are usually centred around certain hotspots where commercial shipping is intense and frequent. In the British Isles therefore, the relative importance of oil pollution as a factor affecting seabird populations, is relatively low because pollution events are relatively infrequent. Worldwide, oil pollution represents a more important impact upon seabirds, but despite this is generally considered not to have a significant long term impact on breeding seabird numbers.

4.10 Persistent Organic Pollutants and heavy metals

Persistent Organic Pollutants (POPs) include pesticides such as DDT, and industrial chemicals such as PCBs and flame retardants. Some of these POPs have oestrogenic effects on birds, or are carcinogenic, and most are toxic although toxicity varies considerably among compounds (Knudsen et al. 2007). POPs tend to accumulate in body fat and increase in concentration up the food chain, so reach highest concentrations in top predators such as skuas and large gulls (Sagerup et al. 2009).

Increased POP levels can show a positive correlation with corticosterone levels in seabirds, as shown in black-legged kittiwakes in Svalbard.

This is an important environmental finding as corticosterone stress hormone levels predict the level of response from seabirds to changing environmental conditions (e.g. SST, food availability, parental effort) (Nordstad et al. 2012). Another POP study on seabirds in the Arctic researched the effects of POPs on the immune system of glaucous gull *Larus hyperboreus* chicks. The findings revealed that multiple POP exposures on the chicks had a negative effect on their immune system (Sagerup et al. 2009). Again, in the face of changing environmental conditions, a weakening of the immune system represents a negative impact on the survival rate of gull chicks. However, very few studies have found harmful effects of POPs on seabirds on a scale that would be likely to affect seabird population size, except in a very few cases such as the impact of a massive pesticide spillage in the southern North Sea in 1967 which killed large numbers of gulls, terns and cormorants (Mitchell et al. 2004). In the highly polluted Great Lakes, POPs caused embryo mortality and population decline in cormorants, gulls and terns, but levels of these compounds in British seabirds are far below the levels reported in seabirds in the Great Lakes.

Heavy metals, especially mercury and cadmium, are also considered to be a hazard for seabirds, but as with POPs, there is very little evidence that these metals influence seabird demography in any detectable way in the British Isles, and impacts elsewhere seem generally to be difficult to detect and probably only very local in exceptional circumstances (for example, lead pollution from paint peeling off military buildings affects albatross chick survival on Midway Island; Finkelstein et al. 2010).

4.11 Plastics

Plastics have been distributed around the oceans in increasing amounts over the past 40 years (Yamashita et al. 2011), acting as a source of pollution and a cause of potential harm to marine animals that may ingest plastic or become entangled. A group of short-tailed shearwaters *Puffinus tenuirostris* accidentally caught in the North Pacific Ocean in 2003 were studied for the potential effects of ingested plastics. Each bird's stomach contained a mean mass of 0.23g plastic. The mass of ingested plastic in the birds was found to correlate with concentrations of lower-chlorinated congeners found in the birds' tissues (Yamashita et al. 2011). Ingestion of plastics can be toxic to seabirds and could become more of an issue with an increase in the numbers of vessels using our oceans and amounts of plastic being used throughout the world. In a study along the coastline of Rio Grande do Sul, southern Brazil, stomach contents were studied from several species of dead seabirds, looking for plastics. Seabird species included shearwaters, albatrosses and petrels. Pieces of nylon line (from fishing boats) accounted for 17% of the plastics found in the seabird's stomach contents (Colabuono et al. 2010) while most was industrial raw plastic pellets or broken fragments of user plastic. Large accumulations of plastic fragments in the gizzard could reduce hunger or food assimilation. According to Colabuono et al. (2010), plastics could be an additional source (alongside transfer up the food chain) of POP exposure to seabirds, as well as a hazard blocking the intestine and filling the gizzard with indigestible material. However, attempts to measure harmful effects of ingested plastics in seabirds have been largely unsuccessful, suggesting that plastics are unlikely at present to have any detectable effect on seabird demography. Amount of plastic in stomachs of northern fulmars from the North Sea is used in the OSPAR Ecological Quality Objective (EcoQO) for marine litter. The preliminary EcoQO defines acceptable ecological quality as the situation where no more than 10% of fulmars exceed a critical level of 0.1 g of plastic in the stomach. During 2003-2007, 95% of 1295 fulmars sampled in the North Sea had plastic in the stomach and the critical level of 0.1 g of plastic was exceeded by 58% of birds. The EcoQO is now also used as an indicator for Good Environmental Status in the European Marine Strategy Framework Directive (van Franeker et al. 2011). However, there is no evidence to demonstrate that these amounts of plastic affect fulmar demography. During winter 2012-13 there has been a pollution incident in southern England with large numbers of seabirds, mostly guillemots and razorbills, killed by plumage contamination with a slick of polyisobutene. This is not the first such incident (see Camphuysen et al. 1999) so this problem may be more common than has been recognised. While such pollution incidents receive considerable media attention, impacts of these winter mortality events are extremely difficult to detect in terms of breeding numbers at colonies.

4.12 Offshore wind farms

The European Union has set an ambitious target of generating 20% of energy consumption from renewables by 2020 (European Commission 2007). Offshore wind farms avoid most of the public opposition, visual intrusion, noise, environmental and land use conflicts that are increasingly evident with terrestrial wind farm developments, and also capture up to 50% more energy because of higher wind speeds and less turbulence over the sea (Kikuchi 2010). By January 2013, 1,662 turbines had been installed and grid connected at 55 offshore wind farms in ten European countries, with 52% of these turbines in UK waters and 65% of the European total in the North Sea (European Wind Energy Association 2013). European offshore wind energy capacity represents 90% of the world total offshore capacity, and over 10% of this capacity was installed during 2012, with a similar amount anticipated in 2013 (European Wind Energy Association 2013).

There is concern that birds collide with offshore wind farm turbine blades (Desholm and Kahlert 2005; Hüppop et al. 2006; Everaert and Stienen 2007). Seabirds are especially vulnerable to collision mortality because they are long-lived animals with deferred maturity and low reproductive rates. As such, their populations are strongly affected by factors that increase adult mortality, whereas populations of most terrestrial birds produce large numbers of offspring that can buffer impacts on survival rates (Exo et al 2003; Desholm 2009). Assessment of the vulnerability of different species of seabirds suggests that northern gannets *Morus bassanus* and large gulls are among the species with highest vulnerability in Europe (Garthe and Hüppop 2004; Furness et al. 2013). This relates in part to gannets and large gulls often flying high enough above the sea to be at risk of colliding with turbine blades, whereas many species of seabirds habitually fly too low over the sea to be at risk (Furness et al. 2013). Gannets also appear to be at higher risk than many other seabirds because when breeding they have especially large foraging ranges from colonies (Thaxter et al. 2012) so may frequently commute past offshore wind farm sites, and after breeding they migrate past offshore wind farms that lie between their breeding and wintering ranges (Kubetzki et al. 2009; Fort et al. 2012). Empirical evidence supports this view. Surveys of seabird numbers at proposed offshore wind farm sites in the UK record high numbers of gannets (WWT Consulting 2012).

Nevertheless, offshore wind farm developments will be localized and therefore not as widespread as the impacts that climate change or fisheries and food availability would have on seabirds. However, seabirds are at risk of collision mortality at offshore wind farms, and may also be affected by displacement (habitat loss) and barrier effects (increasing flight times and so energy costs) (Furness and Wade 2012). Conversely, there are fewer records of offshore collisions than onshore collisions (Boehlert and Gill 2010) and seabirds, wildfowl, and waders have been noted to avoid collisions with offshore turbines by altering their flight paths up to a few hundred metres around wind farms to avoid collisions (Exo et al. 2003), even at night time. In fact, out of all the ducks and geese recorded off Nysted wind farm, less than 1% flew close enough to the turbines during migration to be at any risk of a collision (Desholm and Kahlert, 2005). While the impact of offshore wind farms on seabird populations remains to be seen, there are concerns that collision mortality rates for some seabirds, such as northern gannets and gulls, which tend to fly at heights that make them relatively vulnerable to collisions with turbines (Furness and Wade 2012; Furness et al. 2013), may have detectable effects on population trends in these groups (Busch et al. 2013). Projected effects of Round 3 developments in UK waters are of particular concern given the relative novelty of offshore wind farms, the large scale of Round 3 developments, and uncertainty about impacts on seabirds, and the potential for cumulative impacts on seabird populations. These would most likely be detected in areas where vulnerable seabird species breed close to concentrations of offshore wind farms (such as with gannets and large gulls in south-east Scotland and east England). On the other hand, if offshore wind farms are closed to fishing, they may represent areas where fish populations could increase; whether this would benefit seabirds through higher densities of small pelagic fish remains to be seen, but is a possibility, and some preliminary evidence indicates that some fish populations do increase within offshore wind farm areas (Danish Energy Agency 2013).

4.13 Wave and tidal arrays

Impacts of wave and tidal arrays on bird populations remain to be seen, but are considered likely to be small and confined to a relatively small number of seabird species and populations (Furness et al. 2012). It is thought that diving birds will encounter a risk of entanglement, collision or blade strike with subsurface components (Boehlert and Gill 2010); subsurface components specifically of tidal turbines, as wave energy device structures will be situated mostly above the sea surface. Seabirds such as auks, divers, shags and cormorants dive deep below the sea surface to catch their prey hence any novel construction underwater has the potential to act as a barrier to their movements and a collision hazard.

There is also concern for seabirds during the construction and maintenance of new devices at sea, that boat traffic and disturbance will increase. Increase in boat traffic during the construction, decommissioning and maintenance of devices could flush auk species from hundreds of metres away (Langton et al. 2011). Divers have been reported to be especially sensitive to boat movements and therefore could be negatively impacted by an increase in boat traffic in the PFOW area during construction and maintenance of tidal stream and wave devices. For seabirds along the Oregon coastline, it has been predicted that stormy conditions such as high winds or poor visibility could increase collision rate with wave energy converters (WECs) and that continuous lighting present on any WECs could increase collision risk at night when birds could be attracted to the lights.

Alongside these potential negative impacts of the pending wave and tidal arrays, they also carry potential positive impacts to the local seabird colonies: modifications to water movements and turbulence could alter vertical movements of marine organisms and result in prey and predator aggregations (Boehlert and Gill, 2010). Langton et al. (2011) have also reported that fish move closer to structures after disturbance events and suggest that once tidal stream and wave energy devices are installed, this could increase the success of seabirds foraging around the new device structures. There is speculation that with rotating blades under the sea surface, there is potential for seabirds to collide with rotating blades as with onshore wind turbines. However, Faber Maunsell and Metoc (2007) believe that underwater, birds' moderately fast burst speed would enable escape from the path of tidal turbine blades. There still remains the risk of collision for diving birds underwater with newly installed turbines though, especially for those that actively forage underwater. It is important to emphasise here the novelty of these marine energy devices and hence the scarcity of available literature assessing their potential impacts.

It will be impossible to know the full extent of these device instalments upon seabird populations until they have been installed and the local area and seabird colonies surveyed. However, the likely effects of wave energy and tidal stream turbine arrays have been assessed by McCluskie et al. (2012) and by Furness et al. (2012). These two reviews reach broadly similar conclusions. Impacts of wave energy devices are likely to be substantially less than impacts of tidal stream arrays, and both technologies are likely to have less impact on seabirds than development of offshore wind farms. For example, displacement of seabirds by tidal stream turbine arrays or wave energy devices is likely to be substantially less than from offshore wind farms because wet renewable developments occupy much smaller areas than taken up by offshore wind farms (McCluskie et al. 2012).

Seabirds most likely to be adversely affected can be identified based on knowledge of seabird ecology. For wave energy devices, the main hazards to seabirds are possible displacement of sensitive species from foraging habitat and possible injury through collision with structures either above or below water. While in the past there has been a tendency to assume that displacement equals death, this approach is no longer considered appropriate, and the effects of displacement are more appropriately assessed through a model linking behaviour to demography (McDonald et al. 2012). More speculative impacts include the possibility that such devices may provide 'stepping stones' permitting alien mammal predators such as mink to extend their range, and the possibility that pollutants may enter the marine environment by leakage from these devices (McCluskie et al. 2012). Seabirds most vulnerable to impacts of wave energy devices appear to be divers (all species), as these birds are particularly sensitive to disturbance.

For wave energy devices, divers are the species considered to be at risk of significant displacement, and both red-throated divers and black-throated divers breed in SPAs that have connectivity with parts of PFOW. For tidal stream arrays, of those species considered to be at high or moderate risk of impacts at the population level, several breed in SPAs that have connectivity with parts of PFOW. These are razorbill, shag, common guillemot, great cormorant, red-throated diver, Atlantic puffin and black-throated diver. Given the likely locations of MPAs with black guillemot as a feature, these may not have connectivity with PFOW sites, although the population of this species in PFOW represents a significant proportion of the total Scottish population of this species, as do the populations of all of the other species considered to be at high or moderate vulnerability.

4.14 Conclusions

To conclude, a range of factors in the marine environment impact upon seabird populations, and of these, the ones that have affected seabird populations in the British Isles most in the past are human exploitation and persecution, mammal predation, food abundance, and fisheries. Those most likely to affect seabird populations at present are food abundance, fisheries, mammal predation and climate change, and those most likely to affect seabird populations in the future are also likely to be food abundance, fisheries, mammal predation and climate change, with cumulative impacts likely where these pressures occur together. This would suggest that the most promising candidates for management action might be strategies that would enhance local food supplies around seabird colonies (such as closing these areas to fisheries for preferred food fish such as sandeels, sprats and young herring), or reduce predation impacts (such as eradication of alien mammal predators from seabird breeding habitats). Other possible strategies could involve reduction in levels of human persecution (for the few species that are still subject to legal culling or harvesting), or to the provision of safe nesting sites/colony sites for seabirds where habitat availability may limit breeding distribution.

5. A POPULATION MODELLING APPROACH TO ASSESSING THE RELATIVE EFFICACY OF MANAGEMENT ACTION AFFECTING SURVIVAL VERSUS REPRODUCTIVE OUTPUT

5.1 Introduction

Seabird population sizes are driven by rates of survival and reproduction. Influences on survival and reproduction can be inherent (e.g. competition for resources), external (e.g. predation, provision of fishery discards) and due to management interventions. The relative magnitude of effects on a population from changes in demographic rates will depend on which rates (e.g. adult survival, juvenile survival, fledging success) are affected and by how much. Population modelling provides a means to both quantify the magnitude of a potential impact and also predict the degree of benefit which may be achieved through a management intervention. As with any modelling, the quality and robustness of the results obtained are very dependent on the quality of the data used. Some seabird species have been comparatively well studied (e.g. gannet) with the consequence that demographic rates have been estimated with a reasonable level of confidence and model predictions can be considered reliable. Other species have been much less well studied, and thus greater assumptions about rates of survival and reproduction need to be made and the consequent model outputs are less certain. Nonetheless, certain features of seabird life histories are common, and thus commonalities amongst population models mean that certain general traits can be identified. In the following section the population modelling method is detailed and summary outputs for each species presented.

5.2 Methods

The same age-structued stochastic population model structure was used for all the species modelled here; red-throated diver, Manx shearwater, northern gannet, Arctic skua, great skua, lesser black-backed gull, herring gull, great black-backed gull, kittiwake, Sandwich tern, common tern, common guillemot and razorbill. The modelling followed best practice methods, as described in WWT Consulting (2012). Demographic data used in the models are provided in Table 5.2.1. For those species with only limited data on survival rates (red-throated diver, Manx shearwater, Arctic skua, great skua, Sandwich tern, commion tern) a standard deviation of 0.05 has been used. For the remaining species direct estimates both of mean rates and the standard deviations were available in the literature. Breeding success was taken either from published literature or the JNCC seabird breeding data base (see sections 7.1.4, 8.1.4, 9.1.4 etc and annual reports (e.g. Mavor et al. 2008)).

Table 5.2.1. Demographic rates used in the seabird populations models. Sources provided in table footnote. See also sections 7.1.3, 8.1.3. 9.1.3 etc.

Species (ref.)	**Age at first breeding**	**Mean survival rates (standard deviation)**					**Fledglings/pr.**	**Brood size range (min-max)**
		Adult	Year 1	Year 2	Year 3	Year 4		
Red-throated diver[1,2]	5	0.84 (0.05)					0.635 (0.41)	0 - 2
Manx shearwater[2,3]	5	0.9 (0.05)					0.591 (0.26)	0 - 1
Gannet[4,5,6]	5	0.919 (0.012)	0.42 (0.079)	0.829 (0.031)	0.891 (0.031)	0.895 (0.031)	0.697 (0.035)	0 - 1
Arctic skua[2,7]	4	0.84				NA	0.522 (0.37)	0 - 2
Great skua[2,8]	6	0.9 (0.05)					0.664 (0.1)	0 - 2
Lesser black-backed gull[2,9]	5	0.91 (0.025)	0.819 (0.027)				0.517 (0.37)	0 - 3
Herring gull[10,11,12]	4	0.898 (0.017)	0.82 (0.07)			NA	0.42 (0.28)	0 - 3
Great black-backed gull[9,10,11,12,13,14,15,16]	5	0.93 (0.025)	0.82 (0.03)				0.74 (0.297)	0 - 3
Kittiwake[11,17,18]	5	0.876 (0.035)	0.79 (0.1)	0.79 (0.05)	0.79 (0.05)	0.79 (0.05)	0.65 (0.098)	0 - 3
Sandwich tern[2,19]	3	0.9 (0.05)			NA		0.656 (0.136)	0 - 2
Common tern[2,20]	3	0.88 (0.05)					0.721 (0.49)	0 - 3
Guillemot[10,21,22]	5	0.965 (0.01)	0.56 (0.014)	0.792 (0.03)	0.917 (0.017)	0.938 (0.017)	0.335 (0.113)	0 - 1
Razorbill[10,11,22,23,24]	4	0.9 (0.028)	0.937 (0.028)			NA	0.38 (0.085)	0 - 1
Atlantic puffin[10,11,25]	5	0.924 (0.01)					0.345 (0.11)	0 - 1

1 – Hemmingsson and Eriksson 2002; 2 – JNCC reports on seabird numbers and breeding success (http://jncc.defra.gov.uk/page-2143); 3 – Perrins et al. 1973; 4 - Wanless et al. 2006; 5 – Nelson 2002; 6 – WWT Consulting 2012; 7 – O'Donald 1983, Furness 1987, Phillips et al. 1998; 8 – Furness 1987, Ratcliffe et al. 2002; 9 - Wanless et al. 1996; 10 - Mavor et al. 2008; 11 - Robinson 2005; 12 - Maclean et al. 2007; 13 - Garthe and Huppop 2004; 14 - Calladine and Harris 1996; 15 - Reeves and Furness 2002; 16 - Poot et al. 2011; 17 - Coulson and White 1959; 18 - Frederiksen et al. 2004; 19 – Robinson 2010; 20 – del Hoyo et al. 1992-2006, Becker and Ludwigs 2004; 21 - Harris et al. 2007; 22 - Birkhead and Hudson 1977; 23 - Lloyd and Perrins 1977; 24 - Chapdelaine 1997; Harris et al. 1997.

The model was matrix based, run for a simulated period of 25 years and for each modelled scenario 10,000 simulations were performed from which the median growth rate was calculated. The relative impact on the population growth rate of increases in mortality of all age classes and increases in the rate of reproduction were modelled to estimate the increase required in the latter to offset a given increase in the former. Four levels of additional mortality were trialled: 0%, 1%, 5% and 10%. At each of these additional mortality rates, productivity was modelled sequentially across the range 100% to 120% (at 1% increments). Thus, the increase in productivity required to offset additional mortality of 1%, 5% and 10% could be estimated.

The additional mortality was applied to each age class in proportion to their presence in the population. In some cases, this may not be an appropriate assumption, as immature birds may be more vulnerable to collisions, due to their inherent lack of experience (as represented in their generally lower survival rates compared with adults). Equally, in some instances they may be less vulnerable (through travelling to different areas from adults and so potentially avoiding threats in a specific area). However, such detail is beyond the scope of this modeling. Since additional mortality would be likely to operate as a per capita rate, rather than an absolute number, the number removed from the population at each time step was proportional to the population size. Thus, additional mortality remained at the same proportional level relative to the population size throughout the simulation, whether the population increased or decreased.

Closed populations were assumed since there is no information on rates of exchange (i.e. immigration and emigration) between the breeding colonies being assessed. Similarly there is no information on which to base density dependent population regulation, hence the models were density independent. While this is clearly unrealistic in the longer term, for the benefits of short term modelling of small populations, the risks from violating this assumption were considered to be small. The populations were modelled on an annual time step, with one year age classes up to adults, which is a multi-age class for all individuals of this age and older. Only the final age class breeds and the models were based on a post-breeding census structure (i.e. each census of the modelled population occurs immediately after the breeding season).

Environmental stochasticity was modelled using the mean rates and the standard deviations as listed in Table 5.2.1. Survival rates were drawn from beta distributions, and brood sizes from stretched beta distributions (Morris and Doak 2002). These distributions were used as they generate random numbers from probability distributions which have characteristics appropriate to the demographic rates (i.e. survival rates between 0 and 1, and brood sizes which lie between pre-defined limits).

Demographic stochasticity on survival was modelled using a binomial process, whereby the number of individuals which survive from one time step to the next was estimated using a binomial function (Akcakaya 1991). Thus, the number of individuals alive at time t+1 is generated by a 'coin-toss' process, using the number of individuals alive at time t and the randomly generated survival rate for that time step (as described in the preceding point). The difference between environmental and demographic stochasticity can be thought of as follows; environmental stochasticity generates random values for the probability of survival from one time step to the next; and, demographic stochasticity generates random numbers of individuals which survive from one time step to the next for any given survival probability. Thus, environmental stochasticity models variable environments (e.g. weather effects) while demographic stochasticity models the effects of chance, which are increasingly important as the population size falls.

For each species an initial population size of 10,000 was used. While this bears little resemblance to the real population of most of these species, this parameter has virtually no effect on the results obtained. The population models used for this assessment were stochastic and density independent. It may be argued that the models would be more realistic if they reflected current trends. However, such arguments are based on the premise that the baseline model for any given species should generate predictions which match the recent trend in the population of interest. While this is a reasonable request, it presupposes that the underlying reasons for such trends have been studied and are well understood. This is rarely the case. Most population models are, of necessity, based on demographic rates derived either from different populations or at some time in the past (or often both). Indeed in many instances there are few data on which to base the trends themselves. Population change occurs due to a wide range of factors, some intrinsic (i.e. population regulation through competition for resources, often referred to as density dependence), some extrinsic (e.g. weather conditions), and these two also interact so that intrinsic effects may be greater during periods of unfavourable weather. Without knowing the main drivers of such changes (which is typically the case), simply modifying the survival or reproductive rates in order that the population model generates a prediction in line with the estimated population trend (which may itself be poorly known) without understanding what has really caused observed changes has the potential to render the model very unreliable as a predictive tool. In addition, some population change may be due to movements of individuals between locations (i.e. immigration and emigration), unrelated to change in demographic rates.

In such circumstances the most robust approach for modelling is to avoid the temptation to include density dependence, since this is often based on the premise that ‘it must be operating, therefore it should be included’, even if the mechanism is unknown. Furthermore, this highlights that the most appropriate means for considering model outputs is in terms of the relative outcomes between alternative scenarios. For example, an approporiate measure to consider is the *change* in the population growth rate predicted to occur as a result of a given impact, not the absolute rate of growth itself, which has a high likelihood of being inaccurate. Thus, the onus on the absolute reliability of the model is eased and instead focus is directed towards assessment of the relative magnitudes of a range of predicted impacts.

5.3 Results

The percentage increase in reproductive rate (fledglings per pair) required to offset increases in mortality of 1%, 5% and 10% is shown in Table 5.3.1.

Table 5.3.1. Percentage increase in reproduction required to offset additional mortality of 1%, 5% and 10%.

	Percentage increase in reproduction required to balance increase in mortality		
Species	**1% increase in mortality rates**	**5% increase in mortality rates**	**10% increase in mortality rates**
Red-throated diver	4.47	21.98	43.98
Manx shearwater	4.87	23.41	47.65
Northern gannet	7.18	38.2	79.02
Arctic skua	3.91	21.1	44.1
Great skua	4.68	22.91	47.15
Lesser black-backed gull	6.58	31.8	67.2
Herring gull	5.85	30.64	63.87
Great black-backed gull	4.64	23.6	48.07
Kittiwake	4.6	24.94	51.95
Sandwich tern	2.93	14.3	29
Common tern	2.6	12.69	26.12
Common guillemot	13.76	69.38	146.1
Razorbill	5.14	27.6	58.35
Atlantic puffin	7.08	36.1	75.73

The increase in reproduction required to offset a 5% increase in mortality varied across the species modelled between 13% (common tern) and 70% (guillemot), while for a 10% increase in mortality the increase in reproduction required was in the range 26% to 146% (same species). At the lower end of the range are the shorter lived birds (e.g. terns), for which population growth rates are relatively sensitive to changes in reproduction, though still more strongly affected by survival. At the upper end are the longer lived birds (e.g. guillemot, puffin and gannet) for which population growth rates are much more sensitive to changes in adult survival. The terns, and to a lesser extent Arctic skua, breed earlier than most of the other species. This reduces the relative impact of additional mortality on these species as their populations are better able to replace themselves (through earlier reproduction) than ones which commence breeding at a later age. Conversely, a greater relative increase was required for guillemot. This species has the highest adult survival rate and the lowest reproductive rate, and hence its population can be considered as the most reliant on having long lived adults which although they breed at a low rate, do so for a comparatively longer life span.

It should be stressed that the numbers in table 5.3.1 need to be treated as a guide to the relative efficacy of alternative management options, not as an indication of how the population of any given species will definitely respond to any particular scenario. This is because the models are only as reliable as the data used to parameterise them (for some species very few demographic data are available) and also make several important assumptions, such as continuation of the conditions under which the demographic studies were conducted (i.e. the period over which the rates have been estimated will be representative of future conditions) and an absence of trade-offs between rates (e.g. survival costs of increased reproduction). Nonetheless, the broad message which emerges from this modelling is that for many seabird species, a given percentage of additional mortality can be offset by a rather larger percentage increase in reproduction, but the ratio varies considerably among species.

6. LIFE HISTORY CONSIDERATIONS AND THEIR IMPLICATIONS FOR MANAGEMENT

Life history theory envisages trade-offs between components of fitness, such as survival and reproductive investment (Stearns 1992). Long-lived birds, such as seabirds, may be expected to respond to adverse environmental conditions by reducing their investment in current reproduction in order to increase residual reproductive value and hence maximize lifetime reproductive success. Based on such considerations, Cairns (1987) proposed a model of seabird responses to food supply in which slight reductions in food supply affect breeding adult activity budgets and diet selection, but not breeding success or adult survival rate. Moderate reductions in food supply would affect breeding success, but only severe reductions in food supply would affect adult survival rates, by which time effects on breeding success would be catastrophic. Essentially the Cairns model proposes that because of the overwhelming importance of adult survival in seabird demography, the trade-off between investment in survival and breeding success is skewed strongly towards investment in survival. A similar trade-off would apply in relation to other environmental challenges, such as predation. According to the Cairns model, breeding adult seabirds should avoid risks of predation even at the expense of their own eggs or chicks.

Monaghan et al. (1992) proposed that an example of this trade-off would be that seabirds should abandon their breeding attempt if their body condition fell below a threshold value below which their survival might begin to be compromised. Wernham and Bryant (1998) showed that Atlantic puffins apparently do reduce the quality of their offspring rather than compromising their survival. However, in contrast to this theory and the examples mentioned above, several studies have shown that the trade-off between survival and productivity is less one-sided than this. In the kittiwake in Shetland, although breeding success correlated with sandeel stock biomass as predicted, contrary to the prediction above, adult survival rates of kittiwakes varied considerably from year to year (annual values from 0.98 down to 0.53, with standard error of around 0.02 so highly statistically significant variations). Contrary to the Cairns predictions, kittiwakes did not refrain from breeding in poor conditions, and the adult survival rate correlated with sandeel abundance. Experimental studies with breeding kittiwakes found similar effects, where the trade-off between survival and productivity led to considerable impacts on survival rates as well as on breeding success (Golet et al. 1998, Golet and Irons 1999). Similarly, Davis et al. (2005) showed that both breeding success and survival rates of Arctic skuas were affected by food supply and could be manipulated experimentally, with adults using some supplementary food to increase chick survival as well as using some to increase their own survival.

These results are important in relation to management to increase seabird populations. They imply that management measures that are primarily directed at influencing breeding success are likely to have consequences for survival rates too, and vice versa. This is particularly true where changes in food supply are concerned, but also applies for other challenges such as predation risk. In particular, the often quoted concept that seabird populations are primarily driven by variations in adult survival and not by variations in productivity (Stahl and Oli 2006, Finkelstein et al. 2010) appears to be a flawed concept for several reasons. Firstly, modelling seabird population dynamics (Section 5 above) indicates that changes in seabird population size are influenced by changes in productivity as well as by changes in survival rates for some UK seabirds (particularly those that start breeding at a relatively young age and rear several chicks per season) including terns and Arctic skua (Table 5.3.1). Secondly, modelling of the benefits to population conservation of increasing chick survival in the Laysan albatross by Finkelstein et al. (2010) showed that management to reduce lead poisoning of chicks significantly influenced the predicted population trajectory even in this extreme example of a seabird with especially high adult survival, delayed maturity and low productivity. Thirdly, empirical evidence from studies of seabird populations tends to indicate that both productivity and survival rates vary in relation to environmental factors, and therefore that there is scope for influencing seabird demography through manipulation of both or either of these parameters.

For many seabird species, although there has been some kind of estimate of the adult survival rate, either from studies of individually marked breeding adults at colonies or through analysis of ringing recovery data, there is a lack of evidence on the quantitative impact of environmental variables on survival rates. In contrast, the evidence regarding impacts of environmental factors on breeding success is very good for many seabird species. This leads to a better opportunity to demonstrate the evidence base for management actions to influence productivity, though paradoxically for most seabird species this may have a less powerful influence on population trend than actions to increase survival. Finkelstein et al. (2010) make the important conclusion '*overgeneralizations about demography can stifle useful conservation actions and highlights the need to consider the population-level benefits from multiple management strategies*'.

Another trade-off that should be considered is that between chick starvation and chick predation risk. In many studies it is evident that rates of predation of seabird chicks increase when there is a food shortage. For example, breeding adult Arctic skuas spend longer away from the nest searching for food when sandeel stock biomass is low, and as a consequence more chicks are killed by neighbouring great skuas (Phillips et al. 1998). Studies of common guillemots on the Isle of May showed that when sandeel abundance was severely reduced, adults spent less time at the nest site, and rates of predation of common guillemot chicks increased (Mavor et al. 2005). Productivity reductions ascribed to predation can be ultimately due to food shortage; management that increased food supply or that reduced predation threat could potentially increase productivity, but management action reducing predation when the ultimate cause of low productivity is lack of food might simply result in more chicks starving to death. Such interactions need to be given careful consideration for particular species.

7. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT OPTIONS FOR RED-THROATED DIVER

7.1 Red-throated diver ecology

Red-throated diver is a Schedule 1 and Annex 1 species, SPEC 3 (Unfavourable conservation status (Vulnerable) but not concentrated in Europe. The European breeding population is thought to be around 7,200 to 10,500 breeding pairs (Hagemeijer and Blair 1997).

7.1.1 The species in the British Isles

About 935-1,500 pairs of red-throated divers breed in the British Isles, with 100% of these in north and west Scotland (Forrester et al. 2007). Red-throated divers nest on the shores of freshwater lochs or pools, laying a clutch of one or two eggs (mean clutch size 1.8; Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). They commute from the nesting site to feed on small marine fish (especially sandeels) in shallow coastal waters (Forrester et al. 2007). About 395 pairs breed on 10 SPAs within the UK (Caithness and Sutherland Peatlands, Foula, Hermaness Saxa Vord and Valla Field, Hoy, Lewis Peatlands, Mointeach Scadabhaigh, Orkney Mainland Moors, Otterswick and Graveland, Ronas Hill – North Roe and Tingon, and Rum), representing an estimated 42% of the UK breeding population (JNCC web site).

Adult survival rate has been estimated to be around 0.84 (Hemmingsson and Eriksson 2002), but age of first breeding is thought to be at least 5 years old (Forrester et al. 2007), which would suggest that adult survival rate may well be higher than the one published estimate. UK breeding birds move to the coast after breeding, and overwinter at sea off sheltered, often estuarine, coasts. According to BirdLife International the maximum diving depth is 9 m and since this species often feeds on fish close to the seabed this may limit its foraging areas, although it regularly occurs in water up to 30 m deep in winter (http://seabird.wikispaces.com/Red-throated+Diver). Forrester et al. (2007) summarised the main threats to red-throated divers as disturbance making eggs and chicks vulnerable to predation by gulls and skuas, mink and otter predation of nests, lack of sandeels around Shetland, oil pollution, drowning in fishing nets, disturbance, displacement and collision mortality caused by offshore wind farms.

UK birds may winter from Orkney to the Atlantic coast of France (Forrester et al. 2007). Large numbers of red-throated divers arrive in British waters in autumn and remain until spring. These are thought to include birds from Greenland, Iceland and Scandinavia, and possibly from further east. A revised Great Britain wintering population estimate was compiled using data primarily from systematic line transect surveys by aircraft over marine nearshore areas conducted during 2001-2006. 17,116 (13,198-21,034, 95% confidence interval) red-throated divers were estimated to winter around Great Britain. The largest numbers were found off southeast and east Britain (59.3% of the total was between Flamborough Head, Yorkshire, and Dungeness, Kent), with large concentrations off the English south coast (10.9%), north Wales and Liverpool Bay (9.8%), and eastern Scotland (6.1%) (O'Brien et al. 2009). The Firth of Forth SPA holds 88 red-throated divers in winter. The Outer Thames holds over 6,000 red-throated divers in winter, the largest concentration in UK waters (Webb et al. 2009).

7.1.2 Present, and likely future trends

Breeding populations of red-throated divers in Shetland, Orkney and mainland Scotland probably increased in the second half of the 20th century, but surveys have not been considered highly accurate (Forrester et al. 2007). There was a decline of about 35% in breeding numbers in Shetland (the largest breeding population in the UK) between 1983 (when the population was thought to be at an 'all-time high' of around 700 breeding pairs) and 1994 (Pennington et al. 2004). Since then, breeding numbers appear to have been approximately stable. Wintering numbers are still not very well known, but may have declined since the 1990s in the Moray Firth and NE Scotland where there were previously large aggregations feeding on the local sprat stock. There is no obvious reason to expect major changes in breeding or wintering numbers in the UK in the immediate future (Forrester et al. 2007).

7.1.3 Factors affecting survival rates

Although there is one published estimate for the adult survival rate of red-throated divers, it is in a rather obscure 'grey' publication that was based on ringing relatively small numbers of breeding birds on lakes in part of Sweden (Hemmingsson and Eriksson 2002), so may not apply to populations elsewhere, and is of somewhat uncertain accuracy. There are no quantitative data on how survival rates are affected by particular factors such as food abundance, predators, drowning in fishing nets, oil pollution, or weather conditions. Estimation of the extent to which management might alter survival rates is therefore extremely difficult.

7.1.4 Breeding success in the British Isles

In 1988-90 red-throated diver productivity in Shetland was mainly reduced by low abundance of sandeels (Walsh et al. 1992), with minor additional impacts from predation, flooding of nests, and disturbance by people. Studies in Shetland indicated that human disturbance facilitates predation of red-throated diver nests. Adults which leave nesting lochs on the approach of humans are more likely to fail to rear young, especially when predation pressure by gulls and skuas is high (Walsh et al. 1992). Evidence of the effect of reduced sandeel abundance (which was measured around Shetland by fisheries surveys) includes a reduction in the proportion of broods of two chicks in years of low sandeel stock (Walsh et al. 1992). In addition, during 1971-80 when sandeel stock biomass was high, 97% of fish carried to feed chicks at Foula, Shetland, were sandeels but this fell to 17% in 1988 and was again low in 1989, coinciding with low breeding success (Walsh et al. 1992). According to Walsh et al. (1993), improved productivity of red-throated divers in 1992 compared to 1988-90 may be a reflection of increased availability of sandeels in 1992 compared to 1988-90. Marked variations in success between areas, was thought likely to reflect variations in predation and human disturbance (Walsh et al. 1993). In 1997, otters took eggs, a chick and an adult on Eigg, while great skuas took chicks on Foula (Thompson et al. 1998), but overall productivity in 1997 at 0.67 in Shetland, 0.47 in Orkney, and 0.33-1 elsewhere, was close to the average for 1986-1996. In 1999 in Shetland, some divers failed due to heavy rain in late May causing flooding of nests and otters killed young at five sites on Yell, but mean productivity at 0.59 chicks per pair was almost the same as the average for 1986-1998 (0.6) (Upton et al. 2000). High productivity in Orkney in 1999 (0.75 chicks per pair) was attributed to improved food supply (Upton et al. 2000). In 2000, a severe storm in June destroyed many nests of red-throated divers in Shetland, especially on larger lochs and easterly shorelines, although birds in Orkney and elsewhere were apparently unaffected (Mavor et al. 2001). However, despite the storm, productivity in Shetland in 2000 (0.53 chicks per pair) was only slightly below the mean for 1986-1999 (0.6 chicks per pair). An otter was believed to have taken both clutches of pairs on Eigg in 2000 (Mavor et al. 2001). In 2001, two pairs of divers nested for the first time on artificial sites provided in 2001. Both were successful, raising three chicks on reservoirs where fluctuating water levels had previously caused nesting failure (Mavor et al. 2002). In 2001 on Rum, atypically high rainfall in early June was thought to have reduced productivity there (0-0.25 chicks per pair), while apparently an otter on Eigg not only depredated two clutches and caused a third to be abandoned, but also killed an adult red-throated diver on the nest (Mavor et al. 2002). In 2003, red-throated diver breeding success was similar to the long-term average. Pairs at two lochs where breeding has tended to fail due to falling water levels in summer, were provided with an artificial island and a raft. Both pairs were successful at these artificial nest sites (Mavor et al. 2004), suggesting that artificial nest sites may improve productivity at lochs where nesting on the shore is often unsuccessful. In 2004, many red-throated diver chicks in Shetland died of starvation, and some of these were scavenged by great skuas, giving the false impression that low productivity was due to predation (Mavor et al. 2005). However, at many sites chick weights were low for their age and there was no sign of predation, indicating that food shortage was the main problem. In 2005, red-throated diver breeding success in Shetland was reduced primarily by food shortage, but also by flooding and predation (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for red-throated divers as follows. Success was especially low at Hermaness (0.38), where most pairs failed at an early stage, possibly due to food shortage, and on Foula (0.33) where many chicks disappeared during the second half of July possibly due to predation. On Fetlar, twelve out of 28 pairs had clutches depredated and four of seven chicks lost were known to have been taken by either great skuas or otters. In Shetland, due to a wet winter and above average rainfall between March and July, water levels at nesting lochs remained high throughout the breeding season, with little fluctuation in levels which can cause problems in some years. A small proportion of pairs did not attempt to breed. Few sandeels were seen to be delivered to chicks; nearly all food items identified were saithe, which have a lower calorific value than sandeels, but it is still possible for divers to successfully raise chicks on this lower quality fish. In Orkney, six out of 21 pairs monitored on Mainland failed due to low water levels in dry weather. On Eigg, three pairs all failed due to predation by otters, an annual problem on that island (Mavor et al. 2008).

Breeding success can be improved by provision of nesting platforms, especially on lochs where there is fluctuation in water levels (e.g. reservoirs) or high risk of human disturbance or predator activity (http://seabird.wikispaces.com/Red-throated+Diver). The benefits of nesting rafts for divers have also been established by trials in Argyll (Hancock 2000, ap Rheinallt et al. 2007).

Table 7.1.1 Meta-analysis of main factors contributing to reduced productivity of red-throated divers at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Food shortage	12
Flooding of nests	8
Otter predation	6
Great skua predation	3
Gull predation	2
Drying out of lochs	2
Human disturbance	2

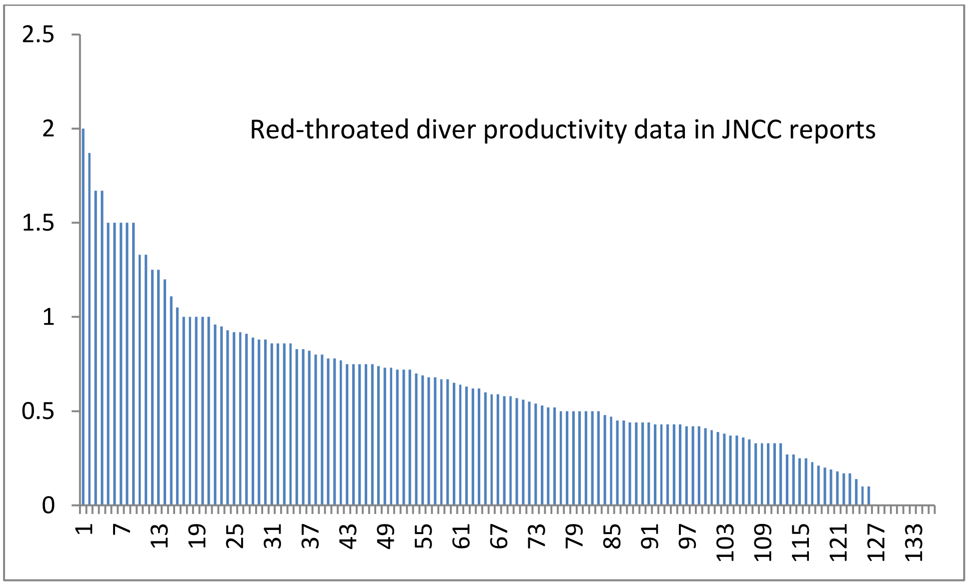

Figure 7.1.1. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of red-throated divers at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

7.2 Management options

	7.2.1 Nesting rafts	7.2.2 Closure of sandeel and sprat fisheries close to wintering areas	7.2.3 Closure of sandeel and sprat fisheries close to breeding areas	7.2.3 Closure of sandeel and sprat fisheries in all UK waters	7.2.4 Prevent oil spills
Evidence of success for this species	High C=High*	Low C=Low	Low C=Low	Low C=Low	Low C=Mod
Evidence of success for similar species	High C=High	Low C=Low	Low C=Low	Low C=Low	High C=Mod
Cost-effectiveness	High C=High	Uncertain C=Low	Uncertain C=Low	Uncertain C=Low	Uncertain C=Low
Feasibility	High C=High	Moderate C=Low	Moderate C=Low	Moderate C=Low	Low C=High
Practicality	High C=High	Moderate C=Low	Moderate C=Low	Moderate C=Low	Low C=High
Applies at SPA populations	Yes C=High	Uncertain C=High	Yes C=High	Yes C=High	Yes C=High

*Confidence in each score is listed as ‘High’ ‘Moderate’ or ‘Low’ based on amount and consistency of evidence

7.2.1 Provision of nest platforms on breeding lochs

There is strong evidence available that providing floating nesting platforms increases productivity considerably, especially on lochs where water levels fluctuate, predators are present, or human disturbance occurs. This also appears to be a management option that would be generally supported by the public, and where volunteer help and matched funding may be available.

A project was started in Argyll in 1976 to provide nesting rafts for breeding divers on two estates. This concept was then adopted by the RSPB for black-throated divers in Argyll and the Highlands, and by Forest Enterprise for red-throated divers in Argyll. Breeding success of red-throated divers in Argyll in 1973-78 (when there were no rafts) averaged 0.35 young reared per pair. Productivity of pairs provided with rafts increased to an average of 0.75 chicks reared per pair per year (ap Rheinallt et al. 2007). Rafts were found to eliminate the risks due to water level fluctuation, and reduce those due to human disturbance and natural predation from foxes and mink which were the main causes of failure in Argyll (Merrie 1996, ap Rheinallt et al. 2007). However, only 15% of red-throated diver pairs in Argyll were provided with rafts. Productivity of the closely related black-throated diver, which also nests in the same kind of nest sites as red-throated divers, was initially doubled on sites in Scotland where they were provided with nesting rafts (Hancock 2000). However, from 1985-2004, 56 breeding attempts at natural sites in Argyll fledged 19 young (0.34 per pair) and 60 breeding attempts at raft sites fledged 30 young (0.5 per pair) (ap Rheinallt et al. 2007) which represently slightly less than a doubling of productivity over a longer period, possibly because some rafts became less suitable over the years; maintenance of rafts or periodic replacement may be a way to maintain higher output over the long term.

In Shetland in 2001, two pairs of red-throated divers nested for the first time on artificial sites provided in 2001. Both were successful, raising three chicks on reservoirs where fluctuating water levels had previously caused nesting failure (Mavor et al. 2002). These pairs were deliberately selected for rafts because it seemed unlikely that they would ever nest successfully on lochs with large fluctuations in water height caused by human management.

Productivity of common loons (great northern divers) is also considerably increased when they are provided with suitable nesting rafts (Piper et al. 2002, de Sorbo et al. 2007).

In North America, the loon preservation committee puts out about 40 rafts each year for common loons (great northern divers) in New Hampshire, and strongly advocates the benefits of these rafts for increasing productivity in that state, protecting the birds from fluctuating water levels and reducing impacts of human recreational disturbance and nest predation by raccoons http://www.loon.org/nest-rafts.php. The Big Mantrap Lake Association also deploys rafts for nesting loons on Big Mantrap Lake Minnesota http://mantraplake.webs.com/loonnestingrafts.htm Their rafts are rather unattractive, with a metal mesh roof, whereas most diver rafts are designed to look like natural islets, but their design presumably helps to reduce predation risk, and they state that it achieves high productivity and is well used by their population of great northern divers. Because diver breeding lakes in North America freeze over in winter, they deploy rafts each spring when ice thaws, and remove them in the autumn. What appears to be an updated version of this raft is described in detail by De Sorbo et al. (2008) who also estimate the cost of materials to construct their design at around US$100 per raft for a raft that will require replacement about every 10 years. In Britain, diver rafts are left in place all year round, reducing the amount of volunteer effort required compared to the situation in North America. Maine Audubon Society provide guidance on construction and siting of diver nesting rafts http://maineaudubon.org/wp-content/uploads/2011/10/Loon-raft-plans.pdf as does Hancock (2000) for divers in Scotland.

Since there are about 395 pairs of red-throated divers breeding on ten SPAs in Britain, it would be necessary to deploy large numbers of nesting rafts in order to significantly increase productivity at the population level. This could best be achieved by selecting sites that have low breeding success because of known problems (fluctuating water levels, human disturbance, predation), since provision of nest rafts on sites where divers regularly nest successfully would not increase productivity. Normally, only one pair of divers nests on a particular loch, although there are a few exceptions to that generalisation. Lochs that have a small island and do not suffer from fluctuating water levels would not be improved by adding a raft; lochs without any islands or where water levels fluctuate strongly would be likely to show a large increase in diver productivity once a raft is available. This probably describes about 25% of red-throated diver nesting lochs, although that value will undoubtedly vary among regions depending on local conditions. The JNCC productivity monitoring data (Figure 7.1.1) shows a long ‘tail’ with poor productivity, suggesting that there is scope for many sites to be enhanced by provision of a nesting raft.

The cost of deploying 100 nesting rafts at suitably chosen red-throated diver lochs where productivity has tended to be low would depend on the logistics (such as distance of the loch from a road). However, typically it might be around £2,000 per site to install nesting rafts (David Okill Shetland Bird Club, Roger Broad RSPB, David Merrie Argyll Bird Club, pers. comms.). Rafts would require occasional maintenance checks, ideally once per year before the breeding season, and would probably require replacement every 10 to 20 years, depending on the exposure of the site (rafts on larger lochs tend to be more affected by wave action than on small lochs, and rafts on lochs sheltered from the wind last longer than those on exposed upland lochs (Roger Broad RSPB, David Merrie Argyll Bird Club, pers. comms.). Provision of nesting rafts at 100 of the least productive nesting lochs within the UK red-throated diver SPA suite could boost productivity at those 100 lochs from an average of around 0.3 to an average of around 0.7 chicks per pair (based on evidence outlined above in this section, and in section 7.1.4). This would be equivalent to boosting the productivity of the entire ca 400 pairs on SPAs by 0.1 chicks per pair (an additional 40 chicks per year). At many red-throated diver SPAs the sites which are least productive are well known as a result of monitoring. For example, Foula SPA has around 12 pairs of red-throated divers, but three of the sites are unsuccessful in most years; two of these are lochs used for the local pump-storage hydro-electric scheme which have severe fluctuations in water level and no islands, leaving incubating red-throated divers either stranded far from the water edge or flooded in most years, and one is the largest loch on the island, close to human disturbance, and which has no island so divers nest on the shore and are usually unsuccessful due to human disturbance. However, possibly a greater gain for the UK red-throated diver breeding population would be achived if nest platforms were distributed not just in SPA areas but across the entire UK population, targeting least productive sites which may often not be on SPAs.

7.2.2 Closure of sandeel and sprat fishing close to wintering area SPAs

Red-throated divers aggregate in sheltered bays and estuaries in winter, areas where they can feed in sheltered sea on small prey fish such as sprats and sandeels. Closure of fishing in areas close to SPAs for nonbreeding red-throated divers, or throughout UK coastal waters, should increase survival rates but evidence for this is lacking (because it would be difficult to gather). There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species. There are localised fisheries for sprats in UK waters, by Scottish fishermen off west Scotland (usually close to Mull), by English fishermen in the English Channel (Lyme Bay), and occasionally as a result of the activity of foreign industrial fishing fleets (ICES 2013). There have in the past been fisheries on sprats in the Moray Firth and in the Firth of Forth, fisheries which appear to have caused local depletion of those stocks and then been closed (Jennings et al. 2012). There has in the past been a fishery by Scottish fishermen on sandeels in Shetland which was closed in 1991 due to the depletion of that stock which has still not recovered, and the large industrial fishery for sandeels in the North Sea has moved around over the years to exploit different stocks within the North Sea, resulting in depletion of many of the distinct sandeel stocks in the northern North Sea, but continued exploitation by Danish fishermen of sandeel stocks in English waters (ICES 2010, ICES 2012). All of these sandeel and sprat stocks are likely to represent important preferred food of red-throated divers in UK waters in winter. Closure of sandeel fishing in UK waters would have low cost for UK fishermen, as almost the entire fishery is carried out by Danish fishermen. Closure of sprat fishing in UK waters would affect a small number of fishing vessels in west Scotland and in Lyme Bay, and limit potential development of sprat fisheries on stocks that are currently unfished (e.g. in Irish Sea, Clyde, Thames, Firth of Forth, Moray Firth).

7.2.3 Closure of sandeel and sprat fishing close to breeding area SPAs or all UK waters

There is strong evidence available that red-throated diver productivity is reduced when prey fish stocks are depleted. By analogy with evidence for some other species (e.g. Arctic skua, kittiwake) adult survival rates are also likely to vary with prey fish stock abundance. Red-throated divers feed in shallow sea within about 20 km of breeding areas. Closure of fishing for sandeels and sprats within 20 km of red-throated diver SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. Also, several of the red-throated diver SPAs are in Shetland and Orkney, regions where there is no fishing for sandeel or sprat (fishing for sandeels at Shetland was closed in 1991 due to depletion of the sandeel stock there and has never re-opened; the sandeel stock at Shetland remains depleted). Closure in all UK waters would include large areas of marine habitat not used by breeding red-throated divers, as their populations are in a limited range within north and west Scotland.

7.2.4 Reducing oil pollution

Red-throated divers are highly vulnerable to oil pollution when at sea, and during winter red-throated divers spend almost 100% of their time on the sea (small amounts of time in flight or underwater). So reducing risk of oil pollution would benefit diver over-winter survival as it would that of auks (see section 18.2.4 for example). However, the quantitative importance of oil mortality for red-throated diver populations is unknown, as is the quantitative effect of any reduction in amount of oil pollution in red-throated diver wintering areas.

8. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR MANX SHEARWATER

8.1 MANX SHEARWATER ECOLOGY

Manx shearwater is a SPEC 2 species (Unfavourable conservation status (localised) and concentrated in Europe. The world population is 340,000 to 410,000 breeding pairs (Mitchell et al. 2004), all located within Europe apart from a handful of pairs nesting in Canada.

8.1.1 The species in the British Isles

About 280,000 to 300,000 pairs breed in Great Britain, the Channel Islands and Isle of Man, and 27,000 to 61,000 pairs breed on Ireland. Manx shearwaters nest in burrows, predominantly in a small number of very large colonies, but with a number of small colonies scattered between the big ones. Virtually all colonies are in the Atlantic Ocean, Celtic Sea or Irish Sea, with the only colonies technically in the North Sea being very small colonies in Shetland. The adults only come ashore after sunset and departures occur before dawn. Adults at colonies during daylight are only those remaining within their burrow. The clutch size is a single egg. Manx shearwaters forage over considerable areas from their colonies, feeding predominantly on small shoaling pelagic fish.

According to JNCC, '*In the breeding season, the UK's SPA suite for Manx shearwater supports an average of 219,898 pairs. This amounts to effectively all of the British breeding population and most (nearly 83%) of the international population*' (http://jncc.defra.gov.uk/pdf/UKSPA/UKSPA-A6-7.pdf). These SPAs are Bardsey Island, Rum, Skomer Skokholm and Middleholm, and St Kilda. Copeland Island, Co. Down was added to the SPA suite for Northern Ireland in 2009.

Adult survival rate of Manx shearwater is around 0.9 (Perrins et al. 1973) but predation by great black-backed gulls at Skomer may add up to 2% to natural mortality rate of adults (Cramp and Simmons 1977-1994). Age of first breeding usually 5 or 6 years old (Cramp and Simmons 1977-1994). Manx shearwaters migrate from British colonies to winter over the continental shelf sea off South America. None remain in European waters in winter. According to Mitchell et al. (2004) the main threats to Manx shearwaters are introduced alien predators, especially brown rats and feral cats. Population declines and extinction have been recorded on islands that have been invaded by rats (through accidental introduction or otherwise). For example, the species became absent or extremely rare on the Calf of Man and on Lundy following the arrival of rats. Natural predators such as eagles, gulls and skuas have some impact, but generally not enough to cause population decline. Puffins may compete for burrows at some colonies, influencing local distribution. Climate change, especially heavy rainfall during incubation, represents a threat, as may fishery bycatch and impacts of fishing on pelagic fish abundance. Since there are relatively few Manx shearwaters breeding outside the British Isles, migration of birds from other populations through British waters is very limited, and almost all birds seen in British waters will be birds from British colonies.

8.1.2 Present, and likely future trends

Manx shearwaters were wiped out on the Isle of Man in the 1780s by brown rats that arrived there off a shipwreck, although a very few pairs recolonized the Calf of Man around 1999 (Mitchell et al. 2004). Most large colonies now remaining in the British Isles are on rat and cat-free islands, although there are several colonies where rats and other mammal predators are in the process of eradicating remaining populations. Over 90% of the British population of Manx shearwaters is on three islands: Rum in NW Scotland, Skomer and Skokholm (these two together with adjacent Middleholm representing a 'supercolony' in Pembrokeshire, Wales). Trends in these populations determine the overall population trend. However, Manx shearwaters breed at, at least 36 and possibly as many as 50, other colonies in Britain and Ireland (Mitchell et al. 2004). Trends in numbers breeding at the largest colonies are uncertain. There is some evidence suggesting that numbers may have increased on Skokholm (Mitchell et al. 2004). There is some slightly stronger evidence suggesting a decline in breeding numbers on Rum (Mitchell et al. 2004). At some smaller colonies, population change has been much clearer. Manx shearwaters were extirpated by brown rats on Canna in 2002 although the population had been between 1,000 and 15,000 pairs in the late 1970s (Mitchell et al. 2004). On the Isles of Scilly, there were 201 occupied burrows in 2000, whereas earlier counts were much larger (e.g. 900 pairs in 1974), so this population has probably been declining, possibly due to brown rats and feral cats (Walsh et al. 1995, Heaney et al. 2002, Mitchell and Ratcliffe 2007). On Lundy, only 154 responses to tape playback were obtained from over 7,000 potential burrows in May 2001, suggesting a major decline in breeding numbers there, possibly again relating to impacts of rats and feral cats (Mavor et al. 2002, Appleton 2007). In the Channel Islands, Manx shearwater colonies are thought to be on the verge of extinction due to brown rats and cats (Mitchell et al. 2004). On Rathlin Island County Antrim, Manx shearwaters have been extirpated from their main breeding areas and only a tiny number of pairs remain on ‘inaccessible’ ledges following introduction of ferrets to the island several years previously (Thompson et al. 1996, Mitchell et al. 2004). In Shetland, colonies on Foula, Horse of Burravoe and Yell were all thought to have been extirpated by mammal predators (especially feral cats), while the colony on Fetlar has been reduced almost to extinction, apparently also due to feral cats (Walsh et al. 1995, Pennington et al. 2004).

The trend for colonies to disappear due to mammal predators is likely to continue in the future, and the view that predation by brown rats on Rum may now be having a serious impact on productivity at that colony (Mavor et al. 2004, 2005, 2006) is of particular concern given that this island holds one of the two or three largest populations of this species in the world, is an SPA for this species, and is a National Nature Reserve.

8.1.3 Factors affecting survival rates

Although there are published estimates for the adult survival rate of Manx shearwaters which indicate an adult survival rate around 0.9 (Perrins et al. 1973), apart from the tentative suggestion that predation by great black-backed gulls may increase adult mortality by up to 2% (Cramp and Simmons 1977-1994), based on the estimate by Buxton and Lockley (2960) that great black-backed gulls on Skomer killed 2,500 Manx shearwaters a year on the island. Mylne (1960) also estimated that great black-backed gulls on Skomer were killing 5,000 to 10,000 Manx shearwaters a year on Skomer in the 1950s, which led to culling of gulls there, and reduction in numbers from over 300 pairs to about 40 pairs (Poole 1995). There are no quantitative data on how survival rates are affected by particular factors such as food abundance, other predators including alien mammals such as rats, drowning in fishing nets, oil pollution, weather conditions while breeding, or conditions on migration and in the wintering area off South America. However, there is evidence that mammal predators such as rats, mink and feral cats that become established on islands where there are Manx shearwater colonies can kill large numbers of adult shearwaters as well as taking eggs and chicks, and can in some cases cause breeding numbers to decline to extinction (Walsh et al. 1995, Mitchell et al. 2004, Lock 2006, Mavor et al. 2006, Luxmoore 2007, Patterson 2007, Swann et al. 2007). This implies the likelihood of a strong impact on adult survival rates. Estimation of the extent to which management might alter survival rates is therefore extremely difficult, but qualitative assessment is certainly possible.

8.1.4 Breeding success in the British Isles

In 1993 on Skomer, very wet weather in May was believed to have contributed to poor productivity as burrows were flooded (Walsh et al. 1994). In 1994, productivity on Canna was apparently around 0.75 chicks per occupied burrow, but fewer than 100 burrows there were occupied. Brown rats were known to be present on the island but were not thought to be affecting the shearwaters (Walsh et al. 1995). Colonies on Foula, Horse of Burravoe and Yell were all thought to have been extirpated by mammal predators (especially feral cats) (Walsh et al. 1995). In contrast, the small colony on Fetlar had recovered to 8 pairs in 1995 with no sign of cat predation that had affected that colony in previous years (Walsh et al. 1995). In 1995, occupancy of shearwater burrows on Canna was noted to be low (40%), and only 0.25 chicks fledged per occupied burrow, the lowest productivity recorded since 1982; there was evidence of predation by rats, with several partly-eaten chicks found (Thompson et al. 1996). On Rathlin Island in 1995, shearwaters have become confined to inaccessible grassy cliff ledges following introduction of ferrets to the island several years previously (Thompson et al. 1996). In 1997, a census at Canna indicated a decline in numbers from 1,000-1,500 pairs in the mid-1970s to 65 in 1997, attributed at least in part to depredations by brown rats, and possibly also feral cats; in addition, breeding success was only 0.07 chicks per pair for the few pairs attempting to breed there (Thompson et al. 1998). In Wales in 1997, many chicks on Skokholm drowned when their burrows flooded in exceptionally heavy rain, but breeding success was good on Skomer (0.64) and Bardsey (0.8) in the same season (Thompson et al. 1998). In 1998, flooding of burrows during heavy rain reduced productivity on Rum and on Lighthouse Island (Co. Down) (Thompson et al. 1999). In 1999, poor breeding success on Rum (0.56 chicks per occupied burrow) was attributed to predation by rats (Upton et al. 2000). On Lighthouse Island (Co. Down) wet weather resulted in chilling of many eggs and drowning of chicks (Upton et al. 2000). In 2000, all the shearwater burrows normally monitored on Canna had been abandoned as a result of the activities of brown rats (Mavor et al. 2001). There was a concern on Rum that numbers of fledglings were declining in the area where these are traditionally ringed on the surface of the colony, and that this decline may be due to activities of rats on Rum (Mavor et al. 2001). In 2001 on Skomer productivity was 0.43 chicks per nest, well below average apparently due to a thunderstorm on 6 July flooding many burrows (Mavor et al. 2002). On Lundy in 2001, signs of rat predation, including broken eggs and carcasses of adults, were apparent at several sub-colonies of the relatively small population on that island which is not regularly monitored (Mavor et al. 2002). On Bardsey in 2002, productivity was high (0.8) but carrion crows there took at least 80 eggs from exposed burrows (Mavor et al. 2003). In 2003 on Rum, breeding success in monitored burrows was higher than average, but there was evidence suggesting that the colony as a whole had reduced numbers of fledglings, and this was thought possibly due to predation by rats (Mavor et al. 2004). On Bardsey in 2003, a pair of carrion crows took eggs from 50 accessible shearwater nests (Mavor et al. 2004). In 2004, for the first time on Rum, eggs were eaten by rats in a small number of the monitored burrows (Mavor et al. 2005). This represents a change from the situation described by Thompson (1987) who found no evidence of rat predation on shearwater eggs on Rum in 1984 and 1985. On Canna, where rat predation led to the extirpation of the Manx shearwater colony in 2002, no shearwaters were found attending any burrows in 2004 and in 2005 one adult was found that had apparently been killed by rats (Mavor et al. 2006). On Bardsey, one pair of carrion crows took as many as 40 shearwater eggs from burrows (Mavor et al. 2005). In 2005, rats were again evident in the Manx shearwater colony on Rum, and took some eggs although the extent of depredation is unclear (Mavor et al. 2006). On Skomer, breeding success of shearwaters has been low since 2000; reasons for failures at 40 nests in 2005 were egg abandoned (13 nests), egg broken (8 nests), adults evicted (3 nests, two by puffins, one by another shearwater), burrow collapse (1 nest) and failed at egg or small chick stage (15 nests) (Mavor et al. 2006). These losses suggest food shortage as the likely main cause. Mavor et al. (2008) described the 2006 breeding season for Manx shearwaters as follows. Breeding success on Rum, at 0.59 chicks per egg laid, was well below the long-term average. The breeding season was exceptionally late and protracted, possibly in part due to very cold weather in April and May. On Bardsey, breeding success (0.74 chicks per occupied burrow) was lowest since monitoring began in 1996. Outwith the study plot, carrion crows targeted accessible nest chambers taking 29 eggs. Breeding success was also below average at Skomer, with many eggs abandoned during incubation, suggesting food shortage.

Table 8.1.1 Meta-analysis of main factors contributing to reduced productivity of Manx shearwaters at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Rat predation	12
Intense rainfall	6

Factor	Cases reported
Feral cat predation	4
Crow predation	4
Food shortage	2
Ferret predation	1

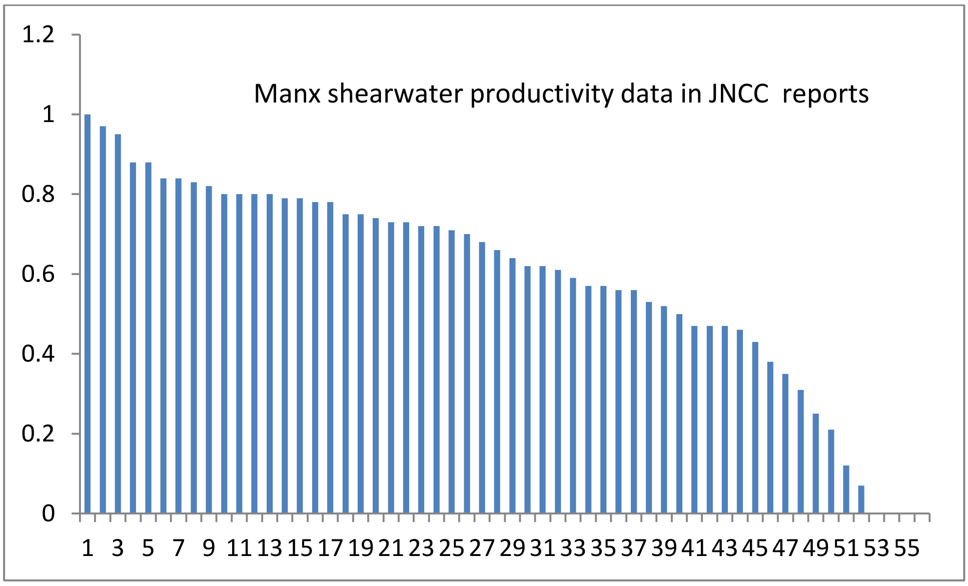

Figure 8.1.1. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of Manx shearwaters at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

8.2 Management options

	8.2.1 Eradicate rats	8.2.1 Eradicate feral cats/ferrets	8.2.2 Exclude large gulls from around colony	8.2.3 Closure of sandeel and sprat fisheries close to breeding areas	8.2.3 Closure of sandeel and sprat fisheries in all UK waters
Evidence of success for this species	High C=High*	Low C=Low	Moderate C=High	Low C=Low	Low C=Low
Evidence of success for similar species	High C=High	High C=High	Moderate C=High	Low C=Low	Low C=Low
Cost-effectiveness	High C=High	High C=High	Moderate C=High	Uncertain C=Low	Uncertain C=Low
Feasibility	High C=High	Moderate C=High	Moderate C=High	Moderate C=Low	Moderate C=Low
Practicality	High C=High	Moderate C=High	Low C=High	Moderate C=Low	Moderate C=Low
Applies at SPA populations	Yes (Rum) C=High	No C=High	Yes C=High	Yes C=High	Yes C=High

*Confidence in each score is listed as ‘High’ ‘Moderate’ or ‘Low’ based on amount and consistency of evidence

8.2.1 Eradication of alien mammals from islands with Manx shearwater colonies

There is strong evidence that productivity is considerably reduced at some Manx shearwater colonies where alien mammals have been accidentally introduced and become established. Survival rates have almost certainly also been reduced at these colonies although evidence of this is lacking because survival rates at those colonies have not been measured, but Manx shearwaters have apparently been extirpated from several islands by alien predators (Ratcliffe et al. 2009) and reduced in numbers at several other islands (Table 8.2.1). Keitt et al. (2002) estimated that feral cats on Natividad Island were killing about 200 shearwaters (of a different species) per year per cat, showing how high cat predation impact can be for shearwaters due to their physical vulnerability while on land. Species of alien mammals currently reducing productivity at Manx shearwater colonies monitored in the UK include brown rats, feral cats, ferrets, red deer, and may include American mink (Table 8.2.1). Sites where alien mammals are reducing productivity at present include one SPA, the island of Rum, which may hold the largest colony of this species in the world. This is a particularly concerning development, as brown rats have been present on Rum for a long time, and were thought to be having no impact on shearwaters when this interaction was studied in the 1980s (Thompson 1987). However, more recent studies indicate an impact on productivity and possibly on survival rates, and suggest that this may now be starting to cause a decline in shearwater numbers (Mavor et al. 2006). This is consistent with many other examples, where rats and seabirds co-existed for some time with little evidence of impacts, but subsequently rat impact increased until the seabird population was eradicated. This is what was described, for example, for the island of Canna (Swann et al. 2007). Ratcliffe et al. (2009) identified Rum as one of the ten top sites in the UK for rat eradication to benefit seabirds. They stated '*The co-occurrence of brown rats with a large proportion of the national Manx shearwater population on Rum is cause for concern. Research during the 1980s showed that rats were relatively rare in this high-altitude colony during the breeding season, and predation was negligible (Thompson and Furness 1991), but recent monitoring has found that predation rates on viable eggs and chicks have increased (A.D. Ramsay unpubl. data). Experimental control of rats in the worst affected colony on the mountain of Hallival is being considered to quantify the impact of rat predation on Manx shearwater productivity (A. Douse pers. comm.)*'. Since Manx shearwaters have become extinct at some sites due to alien mammal predation, eradication of alien mammals at those sites may also require reintroduction programmes to encourage Manx shearwaters to re-colonize those sites.

Table 8.2.1. The main present and recently occupied Manx shearwater colonies in the UK, ranked by estimated colony size in 2000, and the status of invasive alien mammals at these sites.

Colony	**Pairs (AOS) in 2000**	**Alien mammals present**	**Impacts**	**Population trend**	**SPA status**
Rum, Lochaber	120,000*	Brown rats	Yes	Declining	SPA
Skomer, Dyfed	101,800**	No	No	Stable?	SPA
Skokholm, Dyfed	46,200	No	No	Stable?	SPA
Bardsey, Gwynedd	10,000	No	No	Increasing	SPA
St Kilda, Western Isles	4,800	No	No	Stable?	SPA
Copeland, Co. Down	4,600	No	No	Increasing?	SPA
Middleholm, Dyfed	3,000	No	No	Increasing	SPA
Treshnish Isles, Argyll	1,283	No	No	Unknown	
Ramsey, Dyfed	950	(brown rats eradicated 2000)	No	Now increasing	
Eigg, Lochaber	250	Cats	No?	Increasing	
Sanda, Argyll	200	Mink	?	Stable?	
Isles of Scilly	200	Brown rats, cats	Yes	Depleted	

Lundy, Devon	166	Black rats and brown rats eradicated 2004, cats	Yes	Depleted	
Calf of Man	34	Brown rats, cats	Yes	Depleted	
Channel Islands	10	Brown rats, cats	Yes?	Depleted?	
Fetlar, Shetland	7	Cats	Yes	Depleted	
Foula, Shetland	0	Cats	Yes	Extirpated?	
Canna, Lochaber	0	(brown rats eradicated 2005), cats	Yes	Depleted or Extirpated	
Rathlin, Co. Antrim	0-10	Ferrets	Yes	Depleted	
Cardigan Island, Dyfed	0	Brown rats eradicated in 1968 but no natural recolonization by Manx shearwaters there up to 2000.	Yes	Extirpated	

*The population on Rum is now thought to be closer to 60,000 to 70,000 pairs and possibly declining, although 120,000 pairs is the official census from Seabird 2000 (Dr Andy Douse, pers. comm.)

**Perrins et al. (2012) suggest that the true figure for this colony may be 316,000 breeding pairs in 2011, but the 2011 census by tape playback may either indicate a large increase in numbers or a difference due to survey methodology.

Eradication of brown rats at Rum, Isles of Scilly, and Calf of Man, eradication of ferrets at Rathlin Island, eradication of mink at Sanda, and eradication of feral cats at Calf of Man, Fetlar, Foula, and Canna, would all be management measures that could allow increases in Manx shearwater productivity and survival, and in the longer term increases in breeding numbers at these sites, almost all of which have seen depletion of shearwater numbers. A LIFE-funded project to eradicate rats in the Isles of Scilly is currently underway. The eradication is focussing on those linked islands to which re-invasion is unlikely. Due to the linked nature of the remaining islands (all lie within rat swimming distance of each other) any subsequent eradication would have to embrace the entire archipelago, and may be beyond current feasibility limits (RSPB in litt.). Since feral cats and ferrets act as 'superpredators' the removal of their populations from islands should be especially targeted as it is likely to have a greater benefit than removal of rats alone. However, the logistics of eradicating rats are simpler than those of eradicating feral cats and local public approval of rat eradication is likely to be high, whereas killing cats is likely to be less widely supported by the resident human population or stakeholders (Ratcliffe et al. 2009). In particular, on islands such as Foula and Fetlar in Shetland, some cats are 'semi-feral' and will come into houses in winter but live on the cliffs feeding on seabirds in summer. Some of these animals will be considered to be domestic pets by residents rather than feral cats having serious impacts on seabird populations. For cat eradication projects to be successful in the long term on islands with human resident populations, there would be a need to control fertility of domestic cats and to limit future importation of domestic cats to neutered animals (Ratcliffe et al. 2009).

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island (Morgan 2012), Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for other British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

Eradication of feral cats can be more complex than eradication of rats, especially where there is a resident human population with pet cats, as experienced on Ascension Island (Ratcliffe et al. 2009). In that example, large numbers of pet cats were killed by the poison deployment (38% of 168 domestic cats were accidentally killed) despite efforts to keep domestic pet cats safe (which had all been neutered as part of the project). Nevertheless, that eradication successfully removed all of the feral cats and has resulted in re-establishment of seabirds that had been extirpated from the main island (Ratcliffe et al. 2009). This project cost £650,000 over three years, with 24,200 person-hrs of cat control work and 14,480 person-hrs of monitoring work on an island of 97 km^2 (9,700 ha so a cost of £67 per ha which is not far from the costs per unit area of rat eradication programmes on UK islands) but where fieldwork is made relatively easy by the climate, terrain and existence of roads (Ratcliffe et al. 2009). Cats have been successfully eradicated from many islands, the largest being Marion Island (290 km^2) in the sub-Antarctic Indian Ocean, where cats were eradicated in 2000, but this required a 15-year programme including shooting and poisoning and hunting with dogs and introduction of disease (feline panleucopaenia virus) to eradicate a population of around 3,400 cats at the start of the programme that were estimated to be killing over 450,000 seabirds per year (van Aarde 1980, Bloomer and Bester 1992, Bester et al. 2000). The second-largest island from which cats have been eradicated is Macquarie Island (120 km^2) in the sub-Antarctic which had a population of around 2,400 cats, and where the programme of trapping, hunting and deployment of the poison 1080, took 25 years before success was achieved around 2000 (Copson and Whinan 2001). Obviously these very long programmes are much more expensive, and the long timescale required can be attributed to relatively limited expertise in this type of work when those two projects started, and major logistical difficulties of working on uninhabited sub-Antarctic islands far from the mainland. Nogales et al. (2004) provide a useful tabulation of data about 48 islands where eradication of cats has been carried out and summary information on these eradication programmes, indicating that most programmes are now completed within 1-3 years, although none of the projects reviewed in that paper were in the British Isles. Although trapping, shooting, hunting with dogs, poisoning with baits and introducing disease have been the main methods used to eradicate cats, virus-vectored immune-contraception may hold promise in the near future (Courchamp and Cornell 2000).

There seems to be little or no literature on the eradication of alien populations of ferrets (such as the one on Rathlin Island), but this would appear to be equivalent to eradication of species such as American mink, which can be trapped out relatively easily from small areas at low cost (Craik 2007), although eradicating mink from large areas is much more challenging. Efforts by Scottish Natural Heritage to eradicate mink from the Western Isles have been costly and protracted. Rathlin Island lies somewhere between these two extremes.

8.2.2 Exclusion of large gulls from Manx shearwater colonies

Great black-backed gulls were identified as increasing adult mortality by up to 2% (Cramp and Simmons 1977-1994), based on the estimate by Buxton and Lockley (2960) that great black-backed gulls on Skomer killed 2,500 Manx shearwaters a year on the island. Mylne (1960) also estimated that great black-backed gulls on Skomer were killing 5,000 to 10,000 Manx shearwaters a year on Skomer in the 1950s, which led to culling of gulls there, and reduction in numbers from over 300 pairs to about 40 pairs (Poole 1995). There is no evidence of gulls killing Manx shearwaters on Rum (Furness 1988). Small numbers are killed by great black-backed gulls on St Kilda, and at that colony a few are also killed by great skuas (Furness, pers. obs.). There seem to be no major problems with shearwaters being killed by gulls at other colonies in the UK (Mitchell et al. 2004; data in 8.1.3). It would, therefore, appear that removal of gulls that had been a problem for Manx shearwaters has already been carried out at colonies where this was perceived to be a major issue. There might be limited scope for removal of large gulls that can be identified as killing Manx shearwaters at colonies, but the scope for this to significantly improve shearwater survival rates seems limited.

8.2.3 Closure of sandeel and sprat fishing close to breeding area SPAs or all UK waters

Evidence indicates that Manx shearwater productivity can be reduced by food shortage around breeding colonies, although this is less influential and less frequent than predation impacts. Given that Manx shearwaters forage at considerable distances from colonies (Langston 2010 indicates a foraging range averaging 172 km), effective closures would need to be over considerable distances from Manx shearwater SPAs. This would appear to be much less practical and less effective than eradication of alien mammal predators. However, if closures to sandeel and sprat fisheries in UK waters were to be implemented, this would be likely to have a small net benefit to Manx shearwater productivity, and possibly also to survival rates.

9. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR NORTHERN GANNET

9.1 Northern gannet ecology

Northern gannet is a SPEC 2 species (Unfavourable conservation status (localised) and concentrated in Europe. The world population is around 390,000 breeding pairs (Mitchell et al. 2004), with most of these breeding in Europe apart from 77,700 pairs in six colonies in Canada.

9.1.1 The species in the British Isles

About 230,000 pairs breed in Great Britain, the Isle of Man and Channel Islands, and 33,000 pairs on Ireland (Mitchell et al. 2004). Gannets tend to breed in a small number of large colonies, and show reluctance to establish new colonies, though when new colonies are founded they tend to grow rapidly in size through immigration in the early years (Mitchell et al. 2004). The clutch size is a single egg. Gannets feed predominantly on pelagic fish, but can take sandeels when these are abundant, or larger fish such as adult herring and mackerel. In winter gannets feed extensively on fishery discards. Discards appear to be less frequent in the breeding season diet, although tracking data indicate that breeding adults will often feed behind trawlers even when rearing chicks (Votier et al. 2013).

The UK's SPA suite holds about 197,000 breeding pairs of gannets, representing around 98% of the UK breeding population (http://jncc.defra.gov.uk/pdf/ukspa/ukspa-a6-10.pdf). These are on ten SPAs (Ailsa Craig, Fair Isle, Firth of Forth Islands, Flamborough Head and Bempton Cliffs, Grassholm, Hermaness Saxa Vord and Valla Field, North Rona and Sula Sgeir, Noss, St Kilda, and Sule Skerry and Sule Stack). Adult survival rate is 0.92 (Wanless et al. 1996).

Gannets are believed to start breeding from age 5, although data on this are limited and come only from the Bass Rock colony (Nelson 1978, 2002). Non-breeding by adults that have nested before is thought to be very infrequent (Nelson 1978, 2002, WWT Consulting 2012). Gannets from colonies in the British Isles tend to winter from the southern North Sea to the continental shelf off West Africa, with younger birds travelling further south on average. Tracking suggests that adult gannets may be wintering on average further south now than they did a few decades ago, possibly in response to changes in fisheries in the North Sea and off West Africa (Kubetzki et al. 2009, Garthe et al. 2012).

Few threats to gannets have been identified, partly because their numbers have continued to increase up to the present. Gannets accumulate relatively high levels of pollutants compared to other European seabirds, but there is no evidence of toxic impacts, some gannets are killed by oil pollution, by entanglement in fragments of fishing net or on fishing lines, but none of these are thought to represent major threats. Bycatch in fisheries may be a greater problem than is currently recognised (BirdLife 2009), but there is inadequate data to assess this fully. The ability of gannets to swallow large fish as well as small ones, their extremely long foraging range, and their aggressive nature and large size giving them a dominant position in mixed-species feeding groups, makes them relatively insensitive to fluctuations in abundance of any particular fish species. Gannets from some other populations may visit waters around the British Isles. Norwegian gannets appear to winter further north than British gannets (Fort et al. 2012) and many of them winter in the North Sea, but their population is very small (3,850 pairs) compared to that of the British Isles (less than 2%). Icelandic gannets may winter west of the British Isles (WWT Consulting 2012), but their population (25,400 pairs) is relatively small compared to that of the British Isles (about 10%).

9.1.2 Present, and likely future trends

The 2003/04 national census of gannets found nearly 261,000 pairs of gannets in 24 colonies in Britain and Ireland, with three new colonies founded since the previous national census in 1994/95 (Wanless et al. 2005). The rate of increase between 1994/95 and 2003/04 (1.2% per annum) was considerably less than in previous decades, suggesting that the population may be approaching carrying capacity after 100 years of sustained and well-documented increase (Wanless et al. 2005). However, there is no evidence yet of any density-dependent decrease in gannet productivity (Mavor et al. 2005, 2008, WWT Consulting 2012). Changes in the Common Fisheries Policy, and especially the likely phased introduction over the next few years or decades of a zero discarding policy is likely to reduce food supply for gannets (Votier et al. 2013), which may be especially important for gannets in winter when availability of pelagic fish in British waters tends to be lower and gannets then feed more extensively on discards (Garthe et al. 1996). Whether this change will affect gannet numbers is uncertain. Gannets are the dominant scavenger behind fishing vessels and can swallow larger fish than any other UK seabirds (Hudson and Furness 1989), and have exceptionally long foraging range, so they are likely to be less affected than other scavenging seabirds. Changes in discarding may simply encourage a higher proportion of gannets to overwinter off west Africa rather than off Britain (Garthe et al. 2012). Most probably, future increases in gannet breeding numbers are likely to occur at a slower rate, and numbers may possibly stop increasing or possibly even decline.

9.1.3 Factors affecting survival rates

Only one study has investigated survival rates of northern gannets, and that analysis was based on ring recovery data (Wanless et al. 1996). Rather surprisingly, no studies have colour ringed gannets for survival analysis. Wanless et al. (1996) estimated adult survival rate to be 0.919 (0.915 to 0.922), 4th year survival 0.895 (0.889 to 0.900), 3rd year survival 0.891 (0.886 to 0.896), 2nd year survival 829 (0.821 to 0.836) and 1st year survival 0.424 (0.410 to 0.439). The data suggested a decline in survival rate in 1990 to 2000 compared to 1960 to 1990, but there were too few data to determine if this suggestion was a real change or an artefact of small sample size in the recent time period. The study found no clear evidence for differences in survival rates between colonies and did not investigate whether annual variations were caused by specific environmental factors (partly because the data set is not large or robust enough for more detailed analysis than was carried out). Estimation of the extent to which management might alter survival rates is therefore extremely difficult.

9.1.4 Breeding success in the British Isles

In Shetland in 1988-90 when sandeel abundance fell to low levels, gannets switched from feeding primarily on sandeels while breeding to feeding on herring and mackerel, and maintained high breeding success (Walsh et al. 1991). Productivity at Ailsa Craig in 1993 was only 0.53, an exceptionally low value for an established colony, and this appeared to be caused by extremely cold weather (snow and ice) in May causing many birds to fail at the egg stage (Walsh et al. 1994). In 1994, lower than normal productivity at Troup Head (0.5 chicks per pair) was attributed to disturbance or predation at that colony (Walsh et al. 1995). Despite severe weather in 1997, there was no evidence that gannet productivity was adversely affected (Thompson et al. 1998). In 1999, mean productivity at monitored colonies was 0.66 chicks per pair, very slightly below the average for 1986-1998 (0.67), with many nests at Fair Isle washed away by a severe storm in May, and breeding success at Ailsa Craig reduced by disturbance by helicopters flying close over the colony (Upton et al. 2000). In 2000, at Hermaness the productivity was 0.57, the lowest on record there, possibly due to the severe storm in June 2000 (Mavor et al. 2001). However, there was no impact of this storm at Noss (0.73 chicks per pair) and the mean for 2000 for all monitored colonies (0.66) was close to the long-term mean (1986-1999, 0.69). A landslide at Hermaness destroyed several hundred gannet nest sites between 1999 and 2003, causing a reduction in breeding numbers at that colony in contrast to increases continuing elsewhere (Mavor et al. 2004). A new colony on Sule Skerry, which held 15 nests in mid-July 2003 lost the contents of 5 nests due to depredation by great black-backed gulls (Mavor et al. 2004). Mavor et al. (2008) commented that breeding success at newly formed gannet colonies tends to be slightly lower than at established colonies, because colonizing birds tend to be less experienced breeders than birds at established sites. Otherwise, breeding success tends to be consistently high (around 0.68-0.71) at all established colonies in all years, with only small variations between years and between sites. Mavor et al. (2005) stated that the northern gannet’s ability to travel hundreds of miles when foraging, coupled with a diet less reliant on sandeels, ensured that productivity remained high compared with other Shetland seabirds. In 2004, gannets at Shetland colonies fed mainly on mackerel (Mavor et al. 2005).

A licenced harvest of chicks at Sula Sgeir has been carried out traditionally each summer, with around 2000 gannet chicks killed there each year by the men of Ness in Lewis (Beatty 1992, Murray 2008).

Table 9.1.1 Meta-analysis of main factors contributing to reduced productivity of Gannets at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	**Cases reported**
Storms	3
Great black-backed gull predation	1
Helicopter disturbance	1
Cold weather	1
Landslides	1
Human disturbance	1

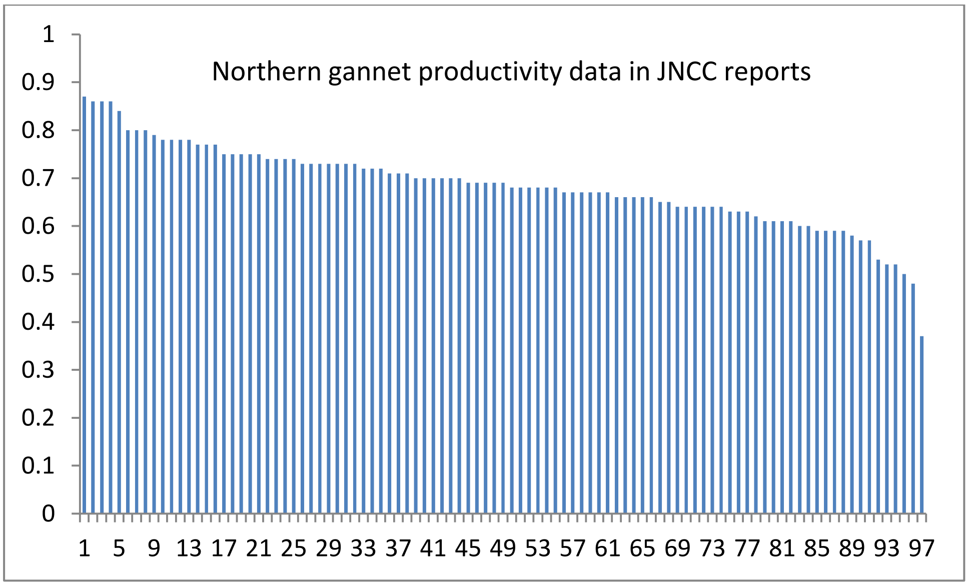

Figure 9.1.1. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of northern gannets at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

9.2 Management options

	9.2.2 End harvest of chicks	9.2.3 Encourage establishment of new colonies	9.2.4 Reduce bycatch in fisheries
Evidence of success for this species	High C=High*	Low C=Low	Low C=Low
Evidence of success for similar species	High C=High	Moderate C=Low	Low C=Low
Cost-effectiveness	High C=Low	Low C=Low	Low C=Low
Feasibility	Low C=High	Moderate C=Low	Low C=Low
Practicality	Low C=High	Low C=Low	Low C=Low
Applies at SPA populations	Yes C=High	No C=High	No C=High

*Confidence in each score is listed as ‘High’ ‘Moderate’ or ‘Low’ based on amount and consistency of evidence

9.2.1 Management to increase survival rates

No management options have been identified that would have a strong probability of increasing adult and/or immature survival rates. One possibility might be reductions in fishing for herring and mackerel, which are important preferred food of gannets. Management measures that increased stock sizes of these fish species might increase overwinter survival and improve body condition of gannets, but there is no evidence available to test whether or not this would be the case, and it is possible that there would be no effect. The fact that rates of increase of gannet breeding numbers did not noticeably decline during the years when herring and mackerel stocks in UK waters were depleted by fishing in the 1960s-1980s, would tend to suggest that gannet survival rates are, at least at present and in past decades, not closely related to abundance of these fish stocks. There is, therefore, no clear evidence base to support management decisions to attempt to increase gannet survival rates.

9.2.2 End harvest of gannet chicks on Sula Sgeir

At almost all colonies, gannet productivity varies very little among colonies or among years, and is normally high. So there is almost no scope for increasing productivity of gannets. The one clear exception to this is Sula Sgeir, where there is a licenced harvest of around 2000 gannet chicks per year by ‘The men of Ness’. Gannet numbers on Sula Sgeir have increased less than at other colonies, indicating that this harvest has apparently affected the rate of colony growth. The harvest of 2000 chicks per year also probably affects productivity of unharvested nests through the human disturbance involved. Ending this harvest would increase productivity at that colony. However, it is unlikely that any other measures could significantly increase gannet productivity or survival at other sites. Figure 9.2.1 shows a plot of the increase in breeding numbers of gannets at different colonies in relation to the size of each colony at the 1969 census. It is clear from this graph that Sula Sgeir is an ‘outlier’. The rate of growth of that colony is much lower than the rate of growth predicted from the data from other gannet colonies. This reduced performance is most likely to be due to the lower productivity at Sula Sgeir and the associated disturbance of birds by the gannet harvest.

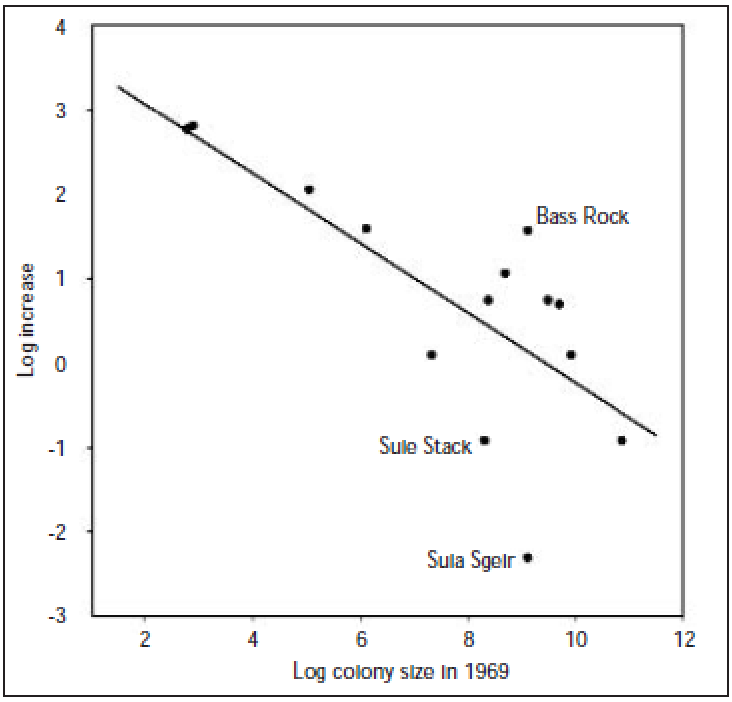

Figure 9.2.1. Rate of increase of gannet colony sizes in recent decades in relation to breeding numbers in 1969 (both axes plotted on log scales). The plot shows the colony at Sula Sgeir to be far below the expected rate of population growth compared to other colonies. From Wanless et al. 2005.

Ending the harvest of gannet chicks (gugas) at Sula Sgeir would increase productivity at that colony by at least 2000 chicks per year, and would be likely to result in more rapid growth of breeding numbers there. However, such a measure may not be acceptable for cultural reasons as this harvest is an important part of the local culture in north Lewis (Murray 2008).

9.2.3 Encourage establishment of new colonies

It might be possible to encourage gannets to form new colonies at locations where the species does not currently breed that are some distance from existing colonies. Birds would be likely to be able to exploit local fish resources more efficiently where they did not have to travel long distances from their colony to feeding areas, and where nesting numbers were smaller so reduced competition. Behavioural attraction methods developed in Maine have since been used globally to restore at least 49 species of seabirds on 89 islands in 14 countries (Jones et al. 2011). Jones and Kress (2011) suggest that a typical restoration project for a seabird in a developed country may cost around £500,000 per annum over a project lasting at least 5, possibly 10 years on average. They point out that the successful project restoring Atlantic puffins to Eastern Egg Rock in Maine took 35 years of sustained effort to establish a population of 100 pairs of puffins. An attempt to start a new Australasian gannet colony at Young Nick's Head, in New Zealand by social attraction was successful, but attempts using the same method with Australasian gannets at Mana Island, New Zealand, and with northern gannets in Nova Scotia and in Quebec failed (Jones and Kress 2011). So it is uncertain whether northern gannets could be encouraged to colonise new sites, and the cost of attempting to stimulate colonisation would be quite high.

9.2.4 Reduce bycatch in fisheries

There is too little data on bycatch rates to be able to assess whether reducing bycatch would significantly increase gannet survival rates, but it has been estimated, for example, that the Gran Sol fishery accidentally caught 1,331 gannets per year in 2006/07 (BirdLife 2009). If such large numbers are killed as bycatch in several fisheries in the wintering areas used by gannets, then reducing bycatch rates could represent a cost-effective compensation measure. There is also a lack of evidence regarding bycatch rates of gannets in EU fisheries, but the current belief is that numbers killed as fishery bycatch in EU waters are probably relatively small.

10. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR ARCTIC SKUA

10.1 Arctic skua ecology

The Arctic skua has a circumpolar breeding distribution, between 56°N and 82°N, on tundra and coastal moors. The world population is probably between 85,000 and 340,000 pairs, with 15,000 to 35,000 of these in the NE Atlantic (Mitchell et al. 2004).

10.1.1 The species in the British Isles

Seabird 2000 estimated 2,100 pairs breeding in Great Britain, with all of these in N and W Scotland, especially Orkney and Shetland (Mitchell et al. 2004). Arctic skuas nest on the ground in a shallow scrape, mostly on moorland close to colonies of terns, kittiwakes and/or auks, from which they steal fish. Most pairs lay two eggs, and the mean clutch size is 1.8 (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). The UK SPA suite includes 780 pairs of Arctic skuas (24% of the British population) nesting on 7 sites; these are Fair Isle, Fetlar, Foula, Hoy, Papa Westray, Rousay, and West Westray (http://jncc.defra.gov.uk/pdf/UKSPA/UKSPA-A6-79.pdf). Adult survival rate (in the absence of illegal shooting) has been estimated at 0.84 (O'Donald 1983, del Hoyo et al. 1992-2006), 0.89 (Furness 1987), and 0.90 (Phillips and Furness 1998), and the mean age of first breeding is 4.5 years (O'Donald 1983). British Arctic skuas migrate to spend the winter in the South Atlantic; none remain in British waters overwinter (Wernham et al. 2002). Threats to British Arctic skuas include depletion of sandeel stocks, climate change (as this species is at its southerly breeding limit in Scotland), territorial conflicts with and depredation by great skuas (Forrester et al. 2007). Migrations of Arctic skuas from higher latitudes (Scandinavia, Iceland, Faroes, the Arctic tundra) brings birds past the British Isles in autumn (when moderate numbers linger in the North Sea and elsewhere during migration, stealing fish from terns and small gulls in particular) and in spring (the latter migration tending to be rapid and off the west of the British Isles) (Forrester et al. 2007).

10.1.2 Present, and likely future trends

Arctic skua breeding distribution in Britain, concentrated mainly in Shetland, Orkney and the Western Isles, has hardly changed over the last 150 years (Mitchell et al. 2004). Breeding numbers increased somewhat between 1969 and 1988, but the trend in recent years has been for large declines throughout the breeding range (Figure 10.1.1), but especially in Shetland and Argyll. The Arctic skua consequently moved from the 'Green' list directly to the 'Red' list, as one of the UK's most rapidly declining breeding birds. There is little suggestion of any population recovery, though since some birds in recent years have refrained from breeding, the decrease in total population size may be slightly less than the decrease in breeding numbers, and some nonbreeding birds might return if conditions improve in future. However, the prospects for this species do not appear to be good. While the declines in Shetland and Orkney are clearly related to reductions in sandeel abundance, declines in Argyll may possibly be due to climate change, as there is no evidence of decreases in small pelagic fish stocks in Argyll waters and breeding numbers of seabirds from which Arctic skuas steal food have not declined in Argyll (ap Rheinallt et al. 2007). However, Arctic skua has been identified as one of the seabird species most likely to be adversely affected in the British Isles by climate change (Oswald et al. 2011). Foster and Marrs (2012) estimated that Arctic skua breeding numbers in Scotland declined by 74% from 1986 to 2011. There is no reason to expect the breeding population of Arctic skuas in Britain to recover in the foreseeable future.

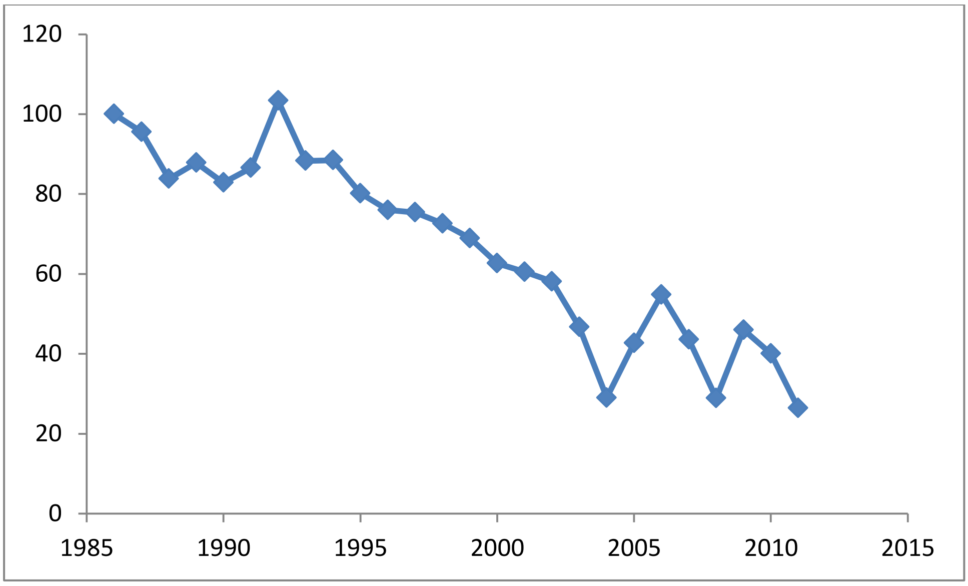

Figure 10.1.1. Arctic skua breeding population index for Scotland. Data from JNCC online database.

10.1.3 Factors affecting survival rates

On Fair Isle, annual survival rates of breeding adult Arctic skuas averaged 0.801 in 1948-62, 0.886 in 1973-75, and 0.747 from 1976-78 (O'Donald 1983). The high rate in 1973-75 was attributed to the colony being studied intensively by scientists so that probably no illegal shooting of adults took place during that period. In contrast, in 1976-78 scientists were only occasionally present and many adults were found shot dead on the island. During 1948-62 it is thought that there was some illegal shooting, but less than in 1976-78 (O'Donald 1983). Shooting clearly affected survival rates of this species on Fair Isle, but shooting of Arctic skuas appears to be very much less common at other colonies; Fair Isle is somewhat exceptional in this regard (Furness 1987). However, survival studies at other colonies have been much less complete than on Fair Isle so survival rates at less disturbed colonies are uncertain. On Foula, Shetland, survival rates of colour ringed adults were reported by Phillips et al. (1998) as 0.883 between 1992 and 1995. Survival rates of pale phase adults were 0.917 in 1993-94 and 0.902 in 1994-95, and of dark phase birds were 0.890 in 1993-94 and 0.906 in 1994-95 (Phillips and Furness 1998a). Davis et al. (2005) carried out an experiment to test the hypothesis that low breeding success of Arctic skuas at Foula, Shetland was due to food shortage (specifically to low abundance of the Shetland sandeel stock). They provided supplementary food to a sample of breeding Arctic skuas within the colony and used other pairs as controls. Supplementary fed pairs not only achieved higher productivity, but adults also spent significantly less time away from the territory searching for food, retained some of the supplementary food themselves rather than feeding it all to their chicks (demonstrated by measurement of body condition and of stable isotopes which differed between the natural and supplementary foods), and showed a significantly higher return rate to the colony the next season than did controls. The study concluded that poor food supply not only reduced productivity, but also reduced adult survival rate. This effect of food supply on adult survival is also consistent with analysis of kittiwake adult survival rates in relation to sandeel abundance (Oro and Furness 2002) or presence of a sandeel commercial fishery (Frederiksen et al. 2004).

10.1.4 Breeding success in the British Isles

In 1988-90 in Shetland, many Arctic skua colonies fledged no young, because they were unable to find adequate amounts of food (sandeels) (Walsh et al. 1991). All monitored colonies in Shetland and most in Orkney showed improved breeding success in 1991 compared to 1986-90, apparently related to high recruitment of sandeels in 1991 (Walsh et al. 1992). On Westray in 1991, many pairs failed at the egg stage apparently due to sheep or large gulls (Walsh et al. 1992). Most monitored colonies also showed higher productivity in 1992 than in 1988-90, apparently reflecting increased sandeel stock biomass since 1991 (Walsh et al. 1993). Slight reductions in productivity were caused by predation and disturbance: on Mousa predation was by otters, on Noss disturbance from tourists and neighbouring Great skuas reduced hatching success, Great skuas caused post-fledging mortality of Arctic skuas on Foula, Fair Isle and six colonies on mainland Shetland, Arctic skua chicks were killed pre-fledging on Hoy, and predation of eggs by sheep and common gulls occurred on Westray (Walsh et al. 1993). The main factor affecting Arctic skua productivity in 1993 (which averaged 1 chick per pair at Shetland colonies) was reported to be predation from great skuas and large gulls (Walsh et al. 1994). In 1995, productivity at Shetland colonies varied between 0.9 and 1.21 chicks per pair, the only factor noted to adversely affect productivity at any of these colonies being wet weather during incubation on Foula (Thompson et al. 1996). Low food availability in Shetland was considered to be the main cause of reduced productivity there in 1997, which also resulted in low attendance by adults (Thompson et al. 1998). Low food availability was considered to affect productivity in 1998 in Shetland, but many surviving chicks were killed either before or after fledging by Great skuas (Thompson et al. 1999). In 1999 in Shetland, productivity (0.46 chicks per pair) was apparently reduced by poor food availability, and chicks and fledglings were killed by Great skuas, whereas in Orkney a good food supply led to higher productivity (0.83 chicks per pair) (Upton et al. 2000). In 2000 in Shetland, productivity averaged 0.57 chicks per pair, this low value being attributed primarily to low abundance of sandeels but also some predation of chicks and adults by Great skuas, and some losses of clutches to the severe storm in mid-June (Mavor et al. 2001). In 2001, Arctic skua productivity was the lowest yet recorded during the monitoring programme, due to scarcity of sandeels in Shetland waters (Mavor et al. 2002). In 2002, lack of sandeels around Shetland greatly reduced productivity (0.18 chicks per pair), whereas food availability at Orkney was considered to be good, and productivity was moderately high (0.74 chicks per pair) (Mavor et al. 2003). In 2003, scarcity of sandeels around Shetland resulted in long-term decline in Arctic skua breeding numbers, extensive nonbreeding, late laying, and breeding success much below levels seen before the collapse of the Shetland sandeel stock (Mavor et al. 2004). The sandeel shortage that affected Arctic skua breeding success on Shetland in 2001-2003 recurred in 2004 and resulted in the lowest productivity since the monitoring programme began in 1986 (Mavor et al. 2005). In addition to adults abandoning eggs and chicks starving, predation by great skuas was also intense. On Handa in 2003, breeding success was high and food availability was apparently high throughout the season, but predation of fledglings by great skuas resulted in 50 to 60% of fledglings being killed (Mavor et al. 2004). Breeding success in Orkney in 2004 was also affected by food shortage, but this was not the case in NW Scotland (Mavor et al. 2005). Mavor et al. (2008) described the 2005 and 2006 breeding seasons for Arctic skuas as follows. The 2005 breeding season was poor, with few young fledged. Food appeared to be scarce. Only two chicks hatched on Foula. On Fair Isle, depredation by great skuas and low food availability resulted in only five young fledging from 71 territories. In Orkney, food availability appeared to be low, some pairs did not attempt to breed, and productivity averaged only 0.3 chicks per pair. On Handa in 2005, productivity was reduced below normal levels by unidentified predators taking eggs, and by great skuas taking Arctic skua chicks before fledging. Post-fledging mortality of Arctic skuas on Handa (killed by great skuas) was reported to be lower than normal in 2005, but still around 40%. In 2006, food was scarce around Foula. Productivity on Fair Isle was higher than in 2005, but post-fledging mortality was high as great skuas were seen to kill fledglings on several territories. In Orkney, breeding success was higher than in 2005, but food shortage in 2006 reduced chick survival. On Coll, heavy depredation by great skuas and great black-backed gulls reduced fledging success.

Davis et al. (2005) carried out an experiment to test the hypothesis that low breeding success of Arctic skuas at Foula, Shetland was due to food shortage (specifically to low abundance of the Shetland sandeel stock). They provided supplementary food to a sample of breeding Arctic skuas within the colony and used other pairs as controls. Supplementary fed pairs achieved significantly higher productivity, partly because the supplementary food allowed chicks to grow better, but also because the supplementary food allowed adults to spend more time guarding their chicks so reduced the numbers lost to predators. This work therefore demonstrates not only that productivity was limited by food supply but also that predation rates interact with food supply such that losses to predators increase when birds are struggling to find food.

Table 10.1.1 Meta-analysis of main factors contributing to reduced productivity of Arctic skuas at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Food shortage	37
Great skua predation	19
Gull predation	4
Sheep	2
Wet weather	2
Human disturbance	1
Otter predation	1

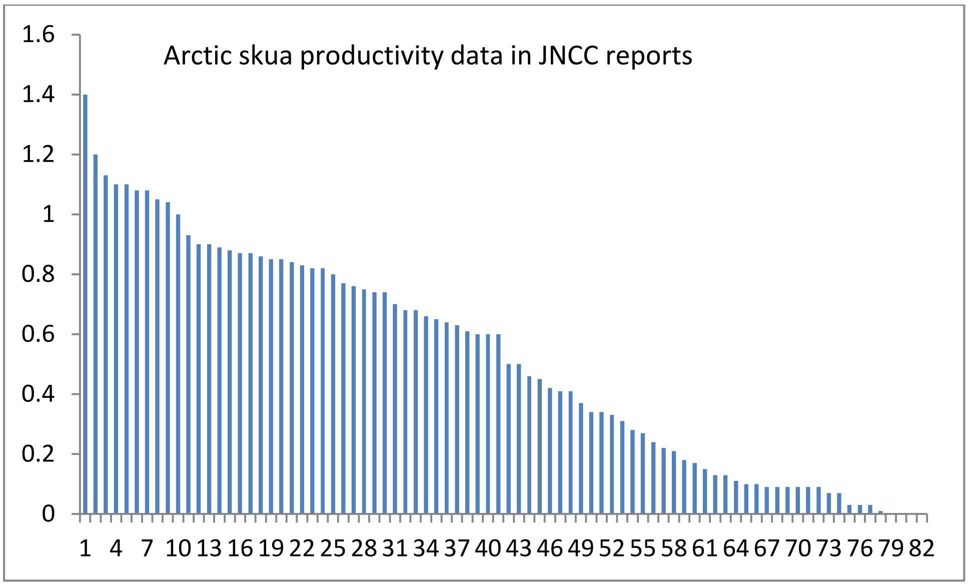

Figure 10.1.2. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of Arctic skuas at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

10.2 Management options

	10.2.1 Closure of sandeel and sprat fisheries close to breeding area SPAs	10.2.2 Provision of supplementary food to breeding pairs	10.2.3 Exclusion of great skuas from buffer zone around colonies
Evidence of success for this species	Low C=High*	High C=High	Moderate C=Mod
Evidence of success for similar species	High C=Mod	High C=High	Moderate C=Mod
Cost-effectiveness	Uncertain C=Low	High C=High	High C=Mod
Feasibility	Moderate C=Low	High C=High	Low C=Low
Practicality	Moderate C=Low	Moderate C=High	Moderate C=Low
Applies at SPA populations	Yes C=High	Yes C=High	Yes C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

10.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs

There is very strong evidence that Arctic skua productivity and survival are greatly affected by prey fish abundance around colonies. Arctic skua foraging mostly occurs within 28 km of colonies but maximum foraging range may be as much as 100 km. Closure of fishing for sandeels and sprats within 50 km of SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

The SPAs for Arctic skua are in Shetland (Fair Isle, Fetlar, Foula) and Orkney (Hoy, Papa Westray, Rousay, and West Westray). These are also the regions around the British Isles where sandeel stock declines have been most pronounced in recent decades. Sandeel stock biomass in Shetland waters has never recovered fully since the local sandeel fishery at Shetland was closed in 1990 due to the depleted state of that sandeel stock. However, the extent to which declines in sandeel stocks at Orkney can be attributed to fishing pressures is unclear. There has been no fishery targeting sandeels in Orkney waters, but connectivity between sandeel populations in Orkney and elsewhere in the North Sea is uncertain. It is possible that changes in sandeel abundance have been influenced more by climate change effects or predation impacts on sandeel larvae from increased stocks of adult herring in the northern North Sea (Frederiksen et al. 2007).

10.2.2 Provision of supplementary food to breeding pairs

There is strong evidence that Arctic skua productivity and survival are greatly affected by food supply. Supplementary feeding (for example on cat food, hen's eggs or day-old chicks) increases productivity and survival. Given that Arctic skua colonies are relatively small it would be possible to provide supplementary food to boost productivity and survival of Arctic skuas, in a similar way to the established practice of supplementary feeding of hen harriers to reduce their predatory impact on red grouse.

Breeding adult Arctic skuas in the UK weigh around 430g on average (Phillips and Furness 1997). Based on the allometric equation relating field metabolic rate to body mass of Charadriiformes (Ellis and Gabrielsen 2002), an adult Arctic skua has a field metabolic rate of 894 kJ/day. Assuming a food utilisation efficiency of 80% (Hilton et al. 2000), this means a food intake of 1,118 kJ/day. Each pair of Arctic skuas therefore requires at least 2,236 kJ per day from its food, and slightly more than this if feeding chicks. Davis et al. (2005) provided 625 kJ per day to experimental pairs, and compared to control pairs given no extra food, this increased productivity from 0.52 ±0.1 to 0.89±0.11, and increased apparent adult survival rate (measured as return rate the next season) from 0.725 to 0.900. Supplementary food provided chicks with 20% (range 5 to 40%) of their dietary protein (Davis et al. 2005). This study indicates that providing pairs of Arctic skuas with supplementary food increases productivity and survival. Provisioning Arctic skuas from around the time they start to lay eggs (May) until chicks fledge (July) would be a simple mamagement measure, since most Arctic skua colonies are small (tens of breeding pairs, even in the SPA populations – see Table 10.2.1) and most are on fairly accessible moorland terrain. There would be a need, however, to establish the best food to provide. Davis et al. (2005) provided cat food and hens' eggs, but day-old chicks or fish might be suitable options. Most Arctic skua pairs do not readily adapt to taking supplementary food, but require a "training period" during which they need to be provided with the supplementary food (usually cat food) alongside a broken hen's egg (broken because they have difficulty in breaking the shell themselves). Once trained, individuals will then accept the supplementary food alone. Davis et al. (2005) mention a few other considerations, including the need to avoid attracting other scavengers to the food (for example by feeding skuas in the evening when their attendance on the territory is high), and placing food close to the centre of each territory to avoid disputes between neighbouring pairs. Such care is critically important since uneaten food could attract large scavengers such as great skuas, great black-backed gulls or ravens and these could have strong negative effects on Arctic skua productivity. All of the SPA populations (Table 10.2.1) are on inhabited islands. Fair Isle has a Bird Observatory employing seasonal staff, Fair Isle and Foula have employed rangers on the islands, Fetlar, Papa Westray and Hoy have RSPB reserves with staff. So it would probably be fairly easy to establish a supplementary feeding regime at any of these SPAs. Supplementary feeding would require about half a day of the time of one person each day from early May to mid-July (a minimum of about 80 days). For benefits of increased chick survival to be fully realised, it might also be necessary at some colonies to consider whether control of great skuas may be necessary to avoid fledgling Arctic skuas being killed by neighbouring great skuas (see section 10.2.3). Supplementary feeding of Arctic skuas would not raise such major issues of public disquiet as supplementary feeding of great skuas. Arctic skuas are generally liked and their numbers are relatively small so the cost of supplementary food for these birds would be very considerably less than for populations of great skuas. Nevertheless, supplementary feeding may not be popular with the general public so the issue would need to be treated sensitively. For that reason, and because care would need to be taken to avoid food attracting larger scavengers, practicality is classified as Moderate.

Table 10.2.1. Most recent published counts of Arctic skua numbers in SPA populations

Colony	Pairs	Year	Great skua pairs adjacent to Arctic skuas out of whole colony	Reference
Fetlar*	5	2011	Tens of pairs out of ca. 400 pairs	Shetland Bird Club (2012)
Foula*	41	2011	Tens of pairs out of ca. 1,800 pairs	Shetland Bird Club (2012)
Fair Isle*	70	2010	Tens of pairs out of 280 pairs	Shaw (2012)
Papa Westray	44	2010	About half of the 29 pairs	Meek et al. (2011)
Westray	27	2010	About half of the 19 pairs	Meek et al. (2011)
Rousay	37	2010	Tens of pairs out of 85 pairs	Meek et al. (2011)
Hoy*	16	2010	Tens of pairs out of 1,346 pairs	Meek et al. (2011)

*These islands are also designated as SPAs for the great skua

10.2.3 Exclude great skuas from buffer zone around Arctic skua colonies

The main predation factor reducing Arctic skua productivity (and reducing adult survival) is predation by neighbouring great skuas. Not only is this recognised in the JNCC data meta-analysis (Table 10.1.1), but it has been reported in several papers. Furness (1977) observed the spread of great skua colony edge into Arctic skua territory on Foula in the 1970s, when great skuas killed many adult Arctic skuas during conflicts over territory ownership. More recently, with reduced food availability, great skuas have killed not only a high proportion of Arctic skua chicks, but also a very high proportion of recently fledged Arctic skuas. Predation by great skuas tends to occur especially where great skuas hold territories on the edge of Arctic skua territory (Phillips et al. 1998), so that they can very easily move in to kill any unattended chicks, and can attack fledglings as they make their first and relatively inept flights (Phillips et al. 1998, Mavor et al. 2008). Estimates vary, but in many cases great skuas kill all surviving Arctic skua chicks either before or after fledging, and even when food supplies for both species are good, great skuas kill around 10 to 20% of Arctic skua fledglings (Furness 1987).

Establishing a buffer zone around Arctic skua colonies from which great skuas are prevented from establishing breeding territories, would reduce the predation impact of great skuas on Arctic skuas. All Arctic skua SPAs are on islands where there are also great skuas nesting, but only relatively small numbers of great skuas nest adjacent to Arctic skua territories (Table 10.2.1) because great skuas prefer areas with less human activity than Arctic skuas will tolerate, and prefer areas with taller vegetation than Arctic skuas prefer (Furness 1987). Once their territory is established, removal of great skuas could only be achieved by killing the adults or by trapping and removing the birds into permanent captivity, since released birds will return to their territory. Since Fair Isle, Fetlar, Foula, and Hoy are also SPAs for great skuas, it might be particularly difficult to manage their breeding distributions on those islands. Westray and Papa Westray are SPAs for Arctic skua but have only recently been colonised by great skuas and removal of great skuas from those islands might be considered as a more acceptable management strategy to protect Arctic skuas. Although Rousay is an SPA for Arctic skua but not for great skua, it has a colony of great skuas that has increased to 85 pairs (there were only 13 in 1982 when there were 96 pairs of Arctic skuas there, Meek et al. (2011)).

11. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR GREAT SKUA

11.1 Great skua ecology

Great skua is classified as a SPEC 4 species; Favourable conservation status (secure) but concentrated in Europe. The world population of the species totals around 16,000 breeding pairs, with all of these within Europe (Mitchell et al. 2004). Numbers have increased during the 20th century, and the species has colonised new regions, including Norway, north Russia, Svalbard, Bear Island and Jan Mayen (Mitchell et al. 2004).

11.1.1 The species in the British Isles

Seabird 2000 estimated that 9,600 pairs bred in Great Britain (all in N and W Scotland) (Mitchell et al. 2004). Great skuas nest on moorland and rough grassland mostly in upland areas away from human habitation. Nests are depressions in the vegetation with minimal lining material. Most pairs lay a clutch of two eggs, the mean clutch size being 1.8 (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). Great skuas catch sandeels when these come to the sea surface, steal some from other seabirds, scavenge discards from fishing boats, and attack and kill a wide range of seabirds, terrestrial birds and mammals up to the size of swans and lambs (Furness 1987). The UK SPA suite includes about 6,300 pairs of great skuas (74% of the British population) on 9 sites: Fair Isle, Fetlar, Foula, Handa, Hermaness Saxa Vord and Valla Field, Hoy, Noss, Ronas Hill – North Roe and Tingon, and St Kilda (http://jncc.defra.gov.uk/pdf/ukspa/ukspa-a6-80.pdf). The adult survival rate in Shetland has averaged 0.9 (Ratcliffe et al. 2002) but appears to have declined as sandeel abundance fell and birds worked harder to try to rear chicks. The mean age of first breeding is around 6 years old (Furness 1987). British great skuas migrate to spend the winter over the continental shelf seas of southern Europe or West Africa (Magnusdottir et al. 2012). The main threats to British great skuas appear to be lack of sandeels and reductions in fishery discarding in the seas around their main colonies (Forrester et al. 2007). Great skuas have high pollutant burdens compared to most European seabirds but toxic impacts are not detectable (Bourgeon et al. 2012). Great skuas from Iceland use two wintering areas, one off Canada and one in southern Europe. Birds migrating to the former do not come into British waters, while birds wintering in the latter area migrate through British waters and mix to some extent in winter with British great skuas although on average Icelandic birds appear to winter slightly further north than those from British colonies (Magnusdottir et al. 2012). Given the slightly smaller population size in Iceland, Faroes and Norway than in Britain, and the fact that about half of the Icelandic birds migrate to Canada, it is likely that ‘foreign’ birds represent less than 25% of those seen migrating through British waters in autumn or spring.

11.1.2 Present, and likely future trends

Breeding numbers of great skuas increased from 1900 up to about 2000 (Mitchell et al. 2004), but while numbers may have continued to increase at small colonies, in recent years the numbers at the largest colonies have declined. At Foula (Shetland), there were 2,495 pairs in 1985-86 but only 2,293 in 2000 (Mitchell et al. 2004), and numbers there have fallen further since then. On Unst (Shetland) there were 1,569 pairs in 1992 but 1,385 in 2000 (Mitchell et al. 2004). At Hoy (Orkney), there were 1,973 pairs in 2000 but 1,346 pairs in 2010 (Meek et al. 2011). These changes have been described as a density-dependent response to a combination of reduced food supply and increased predation (Meek et al. 2011). Continued scarcity of sandeels in the northern North Sea and likely further reductions in fishery discarding culminating in a probable discard ban in the future (Votier et al. 2013), is likely to result in some further decrease in great skua breeding numbers, especially at the larger colonies where competition for food is highest among great skuas (Votier et al. 2007, 2008), since great skuas feed extensively on discards not only as nonbreeders but also throughout the breeding season, and their breeding success is also correlated with sandeel stock biomass (Votier et al. 2004). These larger colonies also tend to be SPAs for the species, so breeding numbers are likely to decrease particularly strongly at colonies designated as SPAs for great skuas.

11.1.3 Factors affecting survival rates

Based on ring recovery data from British great skua colonies, Furness (1978) estimated adult great skua survival of 0.93 between 1938 and 1974, a period of continuous growth of great skua numbers and typically high breeding success (Furness 1987). Based on observations of individually colour ringed birds, annual survival rates of breeding adult great skuas at Foula, Shetland, varied between 0.93 and 0.82, averaging 0.89 over a period of years (1989-1999) when numbers in the colony were declining and breeding success was low (Ratcliffe et al. 2002). Annual survival rate correlated with Shetland sandeel stock biomass, indicating that this was a major environmental driver of survival in this population. However, since most mortality appeared to occur in winter, Ratcliffe et al. (2002) suggested that nutritional stress and reproductive effort breeding in years of poor food supply affect survivorship on migration or in wintering quarters. The results obtained by Ratcliffe et al. (2002) suggest that adult survival rate may typically be 0.93 when food is abundant, but that the survival rate may fall by 0.11 under conditions of food shortage during the breeding season. Great skuas can be killed as bycatch on long line fisheries and through entanglement in fishing nets, but the magnitude of fishery bycatch is unknown. There are suggestions that bycatch in fisheries in wintering areas used by great skuas (especially off West Africa) might be higher than has been recognised. However, there is a need to quantify this bycatch before it would be possible to assess whether reducing bycatch would represent a cost-effective compensation measure.

11.1.4 Breeding success in the British Isles

In 1991, Great skua productivity in Shetland (0.69 chicks per pair) on Fair Isle (0.7) and on Orkney (0.8) was higher than in 1988-90 apparently a consequence of high recruitment of sandeels in 1991, with this improved food supply leading to higher attendance of chicks by adults so higher chick survival (Walsh et al. 1992). In 1993, productivity was generally good in Shetland (around 0.8 chicks per pair), but reduced by ‘cannibalism’ in Orkney (0.4 chicks per pair) with some chicks being killed by neighbouring adults (Walsh et al. 1994). In 1997, adult attendance was unusually low (compared with previous years) suggesting a shortage of food, and there was considerable killing of chicks by neighbouring adults, resulting in low productivity at most Shetland colonies (Thompson et al. 1998). In 1998 in Shetland, productivity averaged 0.6 chicks per pair, with predation, poor weather and food shortage all contributing to reduction in productivity (Thompson et al. 1999). In 1999, productivity averaged 0.71 chicks per pair in Shetland and 0.51 in Orkney. Reasons for losses included poor weather, reduced food availability, and predation by conspecifics (Upton et al. 2000). In 2000, productivity in Shetland averaged 0.8 chicks per pair, with most losses attributed to low abundance of sandeels and some consequent ‘cannibalism’ of chicks. In 2001, productivity at monitored sites was the lowest since the programme began in 1986 with the exception of 1998; however, low availability of sandeels at Shetland affected great skuas less than Arctic skuas (Mavor et al. 2002). It is believed that low availability of fishery discards contributed to poor productivity in 2001, and also led to some chicks being killed and eaten by neighbouring adults (Mavor et al. 2002). Poor food supply depressed breeding success at the main Shetland colonies in 2003, whereas productivity was above average in Orkney (Mavor et al. 2004). Many great skua chicks at Shetland colonies were killed and eaten by neighbouring adult great skuas, although this predation was evidently a consequence of food scarcity (Mavor et al. 2004). The lack of sandeels in 2004 resulted in very low productivity in Shetland and Orkney, but there was no evidence of food shortage in NW Scotland (Mavor et al. 2005). In 2005, great skua productivity at Handa was lower than normal, and analysis of regurgitated pellets indicated less fish than normal (with over 40% of pellets consisting of bird remains) (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for great skuas as follows. Productivity was higher than in recent years, though still reduced as a consequence of food shortage. In addition, killing of great skua chicks by adult great skuas from neighbouring territories reduced breeding success further.

Table 11.1.1 Meta-analysis of main factors contributing to reduced productivity of Great skuas at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	**Cases reported**
Food shortage	25
Great skua predation of chicks	9
Wet weather	2

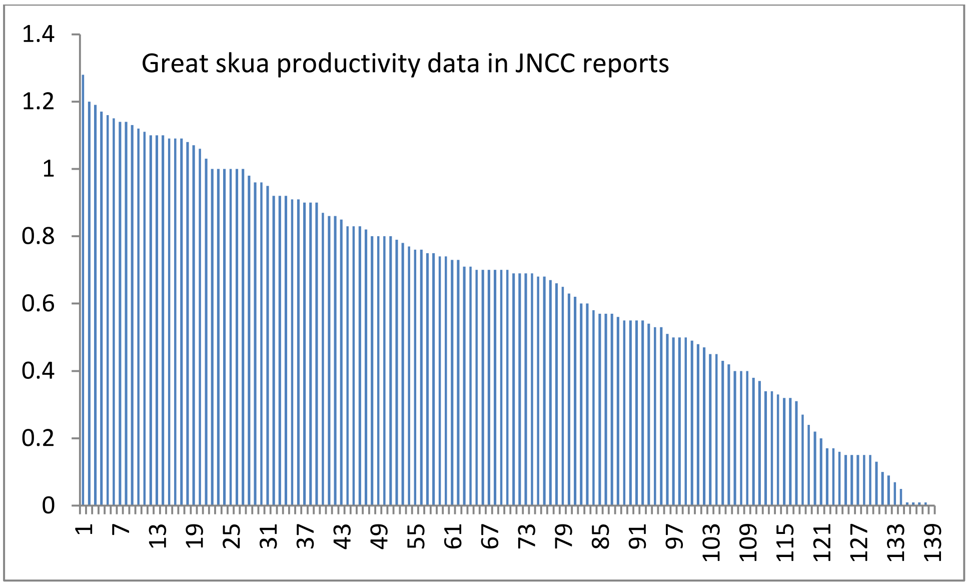

Figure 11.1.1. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of great skuas at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

11.2 Management options

	11.2.1 Closure of sandeel and sprat fisheries close to colonies	11.2.2 Supplementary feeding at colonies	11.2.3 Reduce fishery bycatch
Evidence of success for this species	Low C=Low*	High C=High	Low C=Low
Evidence of success for similar species	High C=Low	High C=High	Low C=Low
Cost-effectiveness	Uncertain C=Low	Low C=High	Low C=Low
Feasibility	Moderate C=Low	High C=High	Low C=Low
Practicality	Moderate C=Low	Low C=High	Low C=Low
Applies at SPA populations	Yes C=High	Yes C=High	No C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

11.2.1 *Closure of sandeel and sprat fishing close to breeding area SPAs*

There is strong evidence that great skua productivity and survival are greatly affected by prey fish abundance around colonies. Great skua foraging mostly occurs within 36 km of colonies but maximum foraging range may be as much as 100 km or more. Closure of fishing for sandeels and sprats within 50 km of SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. Furthermore, great skua SPAs are almost all in areas where there are currently no sandeel or sprat fisheries (six in Shetland - Hermaness, Foula, Fetlar, Ronas Hill, Noss, Fair Isle; Hoy in Orkney, St Kilda in the Western Isles, and Handa in NW Scotland). Discards from trawl fisheries are also important food for great skuas, but it would be nonsense to encourage continued discarding to support populations of scavenging seabirds that are being sustained at artificially high levels by the subsidy of discards they have been receiving for many decades (Votier et al. 2004, 2007, 2008, 2013).

11.2.2 *Supplementary feeding of birds in SPAs*

Great skuas are scavengers, and will take a very wide variety of foods on an opportunistic basis. Supplementary feeding of breeding pairs of great skuas with cat food has been carried out on an experimental basis to test ideas about their ecology and life history (e.g. Ratcliffe unpubl. PhD. Thesis, University of Glasgow; Kalmbach unpubl. PhD. Thesis, University of Glasgow; Hammer pers. comm.). Great skuas are quick to learn of new feeding opportunities (Furness 1987). It would be relatively easy to feed great skuas in SPAs with supplementary food. However, the relatively large colony sizes (1,000 to 2,000 breeding pairs at Foula, Hoy, Hermaness for example) would require considerable quantities of food to be provided to increase breeding success and survival rates. Typically, a great skua has a field metabolic rate of around 2000 kJ/day (Ellis and Gabrielsen 2002). With a food utilisation efficiency of around 0.8 this represents about a pair of breeding great skuas will consume around 1,000 g of food per day. So a colony of 1,000 pairs needs around 1 tonne of food per day, from early May to late July (about 80 days). Providing even half of this requirement as supplementary food would be logistically challenging and expensive. There would be strong public opposition to feeding great skuas in many parts of the species' breeding range, as this species is highly unpopular in areas such as Orkney and Shetland. Furthermore, the general public throughout the UK would be likely to question whether spending money on food for scavenging seabirds would be acceptable. So the Practicality of this measure is scored Low despite the evidence for it being a measure likely to succeed in the objective of increasing survival and productivity.

11.2.3 *Reduce bycatch in fisheries*

There is too little data on bycatch rates to be able to assess whether reducing bycatch would significantly increase great skua survival rates. If large numbers are killed as bycatch in several fisheries in the wintering areas used by great skuas, then reducing bycatch rates could represent a cost-effective compensation measure. There is also a lack of evidence regarding bycatch rates of great skuas in EU fisheries, but the current belief is that numbers killed as fishery bycatch in EU waters are probably relatively small.

12. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR LESSER BLACK-BACKED GULL

12.1 Lesser black-backed gull ecology

The lesser black-backed gull is classified as SPEC 4 Favourable conservation status (secure) but concentrated in Europe. The total breeding population of the species is around 300,000 pairs, with 179,000 pairs in the subspecies '*graellsii*' which occurs from Greenland to Portugal (Mitchell et al. 2004).

12.1.1 The species in the British Isles

Around 117,000 breeding pairs of lesser black-backed gulls were recorded in the Seabird 2000 survey in Great Britain, the Isle of Man and Channel Islands, and 4,800 pairs in Ireland (Mitchell et al. 2004). Lesser black-backed gulls nest on the ground in colonies that tend to be of moderate or large numbers. Colonies are often on islands or in sand dunes, but can be on moorland some distance from the coast. Clutches of 2 or 3 eggs predominate, the mean clutch size being 2.6 eggs (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). Lesser black-backed gulls tend to have a more marine diet than herring gulls, feeding to a greater extent on small fish, and less on terrestrial or intertidal foods (Kim and Monaghan 2006). There is an urban-nesting population of this species which has been growing faster than the population using natural breeding sites, but numbers of urban-nesting lesser black-backed gulls are much smaller than numbers of urban-nesting herring gulls and represent a very small fraction of the total population of the species (Raven and Coulson 1997).

The SPA suite in the UK supports bout 88,600 pairs, which is 'virtually the whole UK population' (http://jncc.defra.gov.uk/pdf/UKSPA/UKSPA-A6-84.pdf). These breed on ten SPAs: Ailsa Craig, Alde-Ore Estuary, Bowland Fells, Firth of Forth Islands, Isles of Scilly, Lough Neagh and Lough Beg, Morecambe Bay, Rathlin Island, Ribble and Alt Estuaries, and Skomer and Skokholm.

The adult survival rate has been estimated at 0.91 (Wanless et al. 1996) and 0.90 (Poole et al. 1998), and the mean age of first breeding is 4 years old (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). British lesser black-backed gulls mostly migrate to spend the winter in north Africa, but increasing numbers (though still a small minority) remain close to breeding areas overwinter. Few 'foreign' lesser black-backed gulls seem to migrate through British waters. Some Icelandic birds may do so, but there is very little evidence of that from the ringing that has been done in Iceland. Continental birds of the subspecies 'intermedius' (from southern Scandinavia and Netherlands) are very infrequently seen in the British Isles, while birds of the subspecies 'fuscus' which breed in northern Scandinavia migrate southeastwards to the Middle East and East Africa, and do not normally pass through British waters (Forrester et al. 2007).

12.1.2 Present, and likely future trends

Lesser black-backed gull breeding numbers in Britain and Ireland increased considerably from 1900 to 2000, reaching 116,684 pairs in the Seabird 2000 census with 57% in England the Isle of Man and Channel Islands, 21% in Scotland, 18% in Wales and 4% in Ireland (Mitchell et al. 2004). This increase was initially triggered by protective legislation and reduced exploitation, but subsequently also encouraged by increased feeding opportunities from fishery discards and edible waste at landfill sites (Mitchell et al. 2004). Since reaching a peak around the mid-1990s, breeding numbers have fallen slightly (Figure 12.1.1). Reasons for the recent decline are thought to include culling (between 1999 and 2002 some 29,000 gulls mostly lesser black-backed gulls were culled at Tarnbrook Fell alone (Mitchell et al. 2004)), reduced food availability from changes in refuse disposal and reduced discarding by fisheries, predation, competition from other large seabirds for food and nest sites, and habitat changes (Mitchell et al. 2004, Forrester et al. 2007). Figure 12.1.1 suggests that breeding numbers are currently (i.e. since about 2005) approximately stable in the UK as a whole. Given that this species appears generally to be more dependent on marine foods than are herring gulls, reductions in amounts of fishery discards may affect this species more, and some decrease in breeding numbers seems likely in response to reduced fishery discarding in coming years.

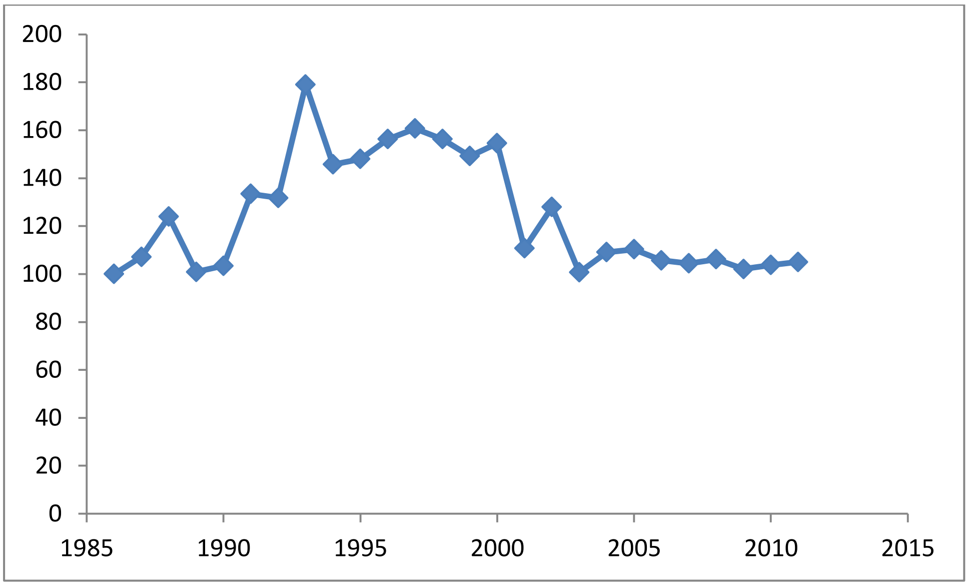

Figure 12.1.1. Lesser black-backed gull breeding population index for the UK. Data from JNCC online database.

12.1.3 *Factors affecting survival rates*

Poole et al. (1998) estimated adult survival rate at 0.9 on Skomer, but declining from 0.976 in 1978 to 0.806 in 1994 but for uncertain reasons, though possibly related to reduced food supply there in the 1990s (Thompson et al. 1996). Wanless et al. (1996) estimated adult survival rate at 0.91 (standard error 0.012) on the Isle of May for a sample of colour ringed breeding adults between 1989 and 1994. They found no significant annual variation in survival rates of this sample (although annual values varied between 0.89 and 0.94), so were unable to identify any environmental factors affecting survival rates. Estimation of the extent to which management might alter survival rates is therefore extremely difficult.

12.1.4 *Breeding success in the British Isles*

In 1995, productivity was low at Skokholm and Skomer, possibly related to low activity of fishing boats trawling in the general area (Thompson et al. 1996). In 1998, causes of low productivity were mainly unidentified, but there were several instances of possible botulism, at Eigg (significant mortality of juveniles), Strangford Lough (up to 400 dead adult and sub-adult gulls), and South Walney (two waves of mortality of adults and chicks) (Thompson et al. 1999). At Orford Ness in 1999, 16% of nests failed due to fox predation (Upton et al. 2000). A total of 50 adults (from a population of 250 pairs) died of botulism at Old Lighthouse Island (Co. Down) in 1999 (Upton et al. 2000). In 2000 an outbreak of botulism caused substantial adult mortality at South Walney for the third successive year (Mavor et al. 2001). In 2000, 11,946 birds were culled at Tarnbrook Fell as part of a long-term water quality management strategy. At Orford Ness, in 2000, 75% of nests (in a colony of 23,000 pairs), failed due to fox predation (Mavor et al. 2001). Breeding numbers at Orford Ness fell from 24,000 pairs in 2001 to 6,500 pairs in 2002 due to fox activity at the colony because fox control was not carried out there in 2002 (Mavor et al. 2003). Mink greatly reduced breeding success at Argyll colonies where no mink control was carried out (Mavor et al. 2004). At Rockliffe Marsh in 2004, more than half of the nests were abandoned in early June, apparently due to food shortage (Mavor et al. 2005). Several colonies in Argyll have been abandoned due to the presence of mink (Mavor et al. 2006). Colonies where mink were present produced 0.13 chicks per pair whereas a colony where mink were trapped out produced 0.5 to 1 chicks per pair (Mavor et al. 2006). At Tarnbrook Fell, the deployment of falcons and controlled disturbance reduced breeding success and reduced breeding numbers in the centre of the colony, but resulted in the formation of new satellite colonies developing some distance away from the original colony (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for lesser black-backed gulls as follows. Data indicated low breeding success at most colonies in 2006. Experimental removal of mink from around certain colonies in Argyll indicated that mink removal boosted breeding output in this species in 2006 in Argyll by 50% relative to control sites where mink were not trapped. Breeding success of urban-nesting lesser black-backed gulls tends to be high except where efforts are made to reduce numbers breeding in urban environments (Raven and Coulson 1997).

Table 12.1.1 Meta-analysis of main factors contributing to reduced productivity of Lesser black-backed gulls at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	**Cases reported**
Botulism	7
Mink predation	5
Food shortage	3
Fox predation	3
Culling operations	1

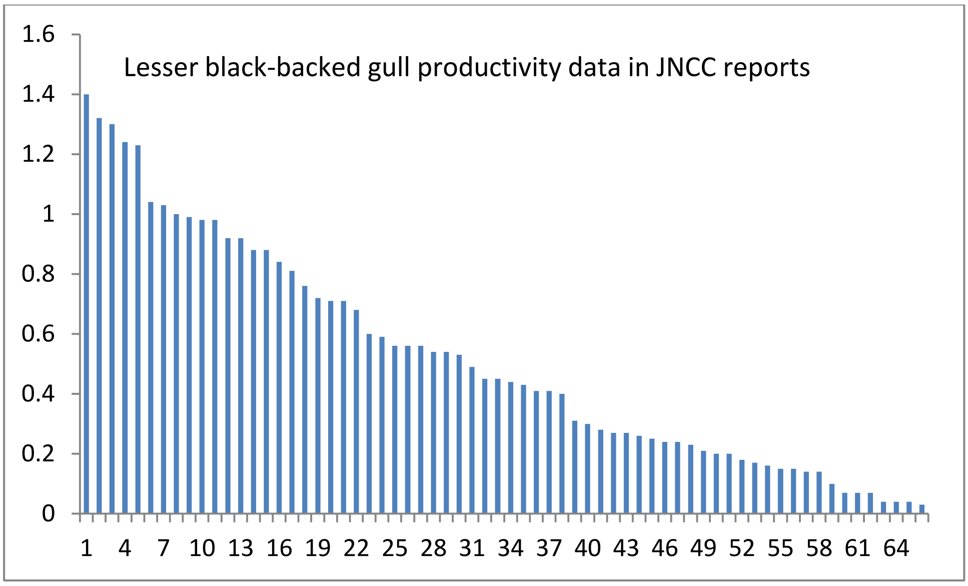

Figure 12.1.2. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of lesser black-backed gulls at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

12.2 Management options

	12.2.1 Mink eradication	12.2.2 Fencing out foxes	12.2.3 End culling	12.2.4 Closure of sandeel and sprat fisheries	12.2.5 Rat eradication
Evidence of success for this species	High C=High*	High C=High	High C=High	Low C=Low	High C=High
Evidence of success for similar species	High C=High	High C=High	High C=High	High C=Low	High C=High
Cost-effectiveness	High C=High	Moderate C=High	High C=High	Uncertain C=Low	High C=High
Feasibility	High C=High	Moderate C=High	Moderate C=High	Moderate C=Low	High C=High
Practicality	Moderate C=Low	Moderate C=High	High C=High	Moderate C=Low	High C=High
Applies at SPA populations	No C=High	Some C=High	Few C=High	Yes C=High	Few C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

12.2.1 Eradication of American mink

Mink are a major determinant of productivity at several colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of invasive alien mink would allow lesser black-backed gull productivity to increase at colonies where this predator is present.

Eradication of mink from small islands can be achieved relatively easily by trapping, although eradicating mink from large areas is much more challenging. Efforts by Scottish Natural Heritage to eradicate mink from the Western Isles have been costly and protracted. However, mink swim well, and are likely to re-colonise islands that are within 2 km of habitat with mink present. Eradication of mink from mainland areas is more difficult, although there is evidence from mink eradication projects in the highlands of Scotland and in Iceland that mink can be eradicated from river catchments and in such cases re-colonisation by mink from adjacent river catchments may be inhibited by upland areas in between.

Where eradication of mink is difficult to achieve due to re-colonisation from adjacent areas, Craik (2007) showed that trapping mink in the immediate vicinity of seabird colonies in late winter and early spring normally leads to a 'mink-free' status for the seabird colony through the summer, resulting in considerably improved productivity (from around 0.2 chicks per pair up to around 0.8 chicks per pair). However, trapping mink every year represents a long-term commitment that will be more expensive in the long term than eradication if that can be achieved. An alternative to annual control of mink may be mink-proof fencing to protect colonies, such that the need for annual trapping is removed once the area has been fenced and mink within the fenced area have been removed. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012) (see 12.2.2 below). By 2006, in total, around 109 km of predator-proof fencing had been erected in various areas of mainland New Zealand to exclude predators from sites with important populations of native animals and birds (Scofield et al. 2011, Innes et al. 2012, Scofield and Cullen 2012). This predator-proof fencing is expensive, costing around £100 per m to construct, and around £1 per m per year to maintain, with a life-span in New Zealand of around 25 years, so a considerable rate of depreciation (Scofield et al. 2011). However, maintenance costs and life span will depend very much on the environment where the fencing is set up. In New Zealand, where much of the fencing is in forested habitat, trees falling onto the fence can cause expensive damage, as can cyclones (Scofield et al.

2011). In the predominantly open habitat of UK seabird colonies such fencing would be under less risk of damage, although corrosion from salt spray would be a consideration. There are several companies in New Zealand providing predator-proof fencing, for example see:
http://www.xcluder.co.nz/
http://www.meshindustries.com/Home
http://www.pestprooffences.co.nz/

There are several examples of the use of predator-proof fences to protect seabirds from mammals. A predator-proof fence completed in 2007 stretches 10.6 km across the neck of the peninsula from coast to coast at Cape Kidnappers Peninsula, North Island, New Zealand. This fence protects a privately owned and financed seabird restoration project where grey-faced petrels and Cook's petrels are being re-introduced (Cooper 2013 http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels).

A good example of successful deployment of a predator-proof fence to protect a seabird colony is one erected in 2001 to protect 36-ha on Pitt Island (Chatham Islands, New Zealand) from feral cats and pigs. Between 2002 and 2005, 200 endangered Chatham petrel chicks from the only known breeding site on South East Island (Chatham Islands) were moved into the fenced reserve. In 2012, 17 pairs from these translocated birds returned to breed, producing 15 chicks http://www.chathams.co.nz/index.php/naturalheritage/116-chatham-petrel-recovery

Cooper(2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels lists a further ten examples of deployment of predator-proof fencing around seabird colonies in New Zealand, Hawaii (USA) and Azores (Portugal).

12.2.2 Exclusion of foxes from colonies

Foxes are a major determinant of productivity at several colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Foxes can be controlled on small islands by shooting, although recolonisation from the mainland may be an issue for islands situated within 1 or 2 km of the mainland. Fencing of colonies to exclude foxes would allow lesser black-backed gull productivity to increase at colonies where this predator is present. In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require some maintenance. A more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka' ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Such completely predator-proof fencing may be unnecessary to protect gull colonies from foxes, but might be appropriate for colonies subject to predation by rats or mink as well as by foxes (see section 12.2.1).

12.2.3 End culling

Culling of breeding adult lesser black-backed gulls has previously taken place at many colonies for a variety of reasons, including reduction of impacts on tern colonies and reduction in bacterial contamination of drinking water supplies. Refusal to permit large-scale culling could increase survival rates of adult lesser black-backed gulls. There are additional measures taken to control impacts of gulls, including removal of nests from urban locations, and disturbance of birds by trained falcons to reduce their use of urban refuse. However, such measures are unlikely to have a significant impact on gull demography at the level of the national population.

12.2.4 Closure of sandeel and sprat fishing close to breeding area SPAs

Food shortage is implicated as a cause of reduced productivity at some colonies in some years. Lesser black-backed gulls depend more on small pelagic fish than do herring gulls. Mean foraging range of lesser black-backed gull is 72 km and maximum foraging range 181 km (Thaxter et al. 2012), so closure of sandeel and sprat fishing within 100 km of SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

12.2.5 Eradication of rats

Although relatively few lesser black-backed gull colonies appear to be subject to rat predation impacts according to the JNCC annual reports, there is evidence that eradication of rats can increase lesser black-backed gull breeding success at islands where rats are numerous. On Ailsa Craig, brown rats colonized in 1889, and there was still a colony of lesser black-backed gulls present in 1990, but their breeding success was about one-third that expected, and the reduction was attributed to rats killing gull chicks (Zonfrillo 2001). Rats on Ailsa were eradicated in 1991 and lesser black-backed gull breeding success improved immediately, to about three times the productivity experienced when rats were present (Zonfrillo 2001, and pers. comm.).

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

At colonies that are on the mainland, or are on islands very close to the mainland so that rats would easily be able to recolonize the island, predator-proof fencing might be an alternative option (see 12.2.2.).

13. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR HERRING GULL

13.1 Herring gull ecology

The herring gull subspecies breeding in Britain is *Larus argentatus argenteus*, which has a total population of about 180,000 to 200,000 pairs distributed across the British Isles, the Netherlands, Belgium, northern France, western Germany, the Faroe Islands and Iceland. The subspecies *L. a. argentatus* breeds in Scandinavia, the Baltic States, Poland, Russia, parts of Germany, southern and eastern France, and numbers around 500,000 to 600,000 pairs (Mitchell et al. 2004), and birds from especially the northern part of its range visit the British Isles in winter.

13.1.1 The species in the British Isles

Seabird 2000 found around 143,000 pairs of herring gulls breeding in Great Britain the Isle of Man and Channel Islands, and about 6,500 pairs in Ireland (Mitchell et al. 2004). Herring gulls breed in a range of colonies from small to large, and in a range of habitats from flat ground to cliffs, and will readily nest on buildings. The urban-nesting population of this species has been growing faster than the population using natural breeding sites, but numbers of urban-nesting herring gulls represent a very small fraction of the total population of the species (Raven and Coulson 1997). The clutch size is usually 2 or 3 eggs, with a mean clutch size of 2.6 (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). Herring gulls are generalists taking a wide diversity of foods, including intertidal prey and terrestrial foods as well as marine foods. The UK SPA suite for this species holds about 54,600 pairs of herring gulls (about 32% of the UK population of the species) across 12 sites: Ailsa Craig, Alde-Ore Estuary, Buchan Ness to Collieston Coast, Canna and Sanday, East Caithness Cliffs, Firth of Forth Islands, Flamborough Head and Bempton Cliffs, Fowlsheugh, Morecambe Bay, Rathlin Island, St Abb's Head to Fast Castle, and Troup Pennan and Lion's Heads (http://jncc.defra.gov.uk/pdf/ukspa/ukspa-a6-85.pdf). Herring gull adult survival rate has been estimated at 0.93 (Glutz von Blotzheim & Bauer 1982), 0.92 (Coulson and Butterfield 1986), 0.94 (Chabrzyk and Coulson 1976), 0.88 (Wanless et al. 1996), 0.88 (Pons and Migot 1995), and 0.81 (Poole et al. 1998), and age of first breeding is 3 to 7 years, averaging 4.5 (Cramp and Simmons 1977-1994, Glutz von Blotzheim & Bauer 1982). British herring gulls are not migratory, but rather disperse over short distances, mostly remaining within Britain all year round (Wernham et al. 2002). Large numbers of herring gulls of the subspecies *argentatus* visit Britain to spend the winter in the North Sea and eastern Britain, though very few of those birds reach the west coast of Britain or sea areas to the west of the country (Wernham et al. 2002).

13.1.2 Present, and likely future trends

Herring gull breeding numbers increased considerably from 1900 to 1969-70 when an estimated 344,000 pairs nested in Britain and Ireland. From 1969 to 2000 numbers declined by about 50-60%, with the population in 2000 estimated at 150,000 pairs, with 49% in Scotland, 38% in England the Isle of Man and Channel Islands, 9% in Wales, and 4% in Ireland (Mitchell et al. 2004). From 2000 to 2011 numbers have declined further (Figure 13.1.1). The recent decline seems to have been greater in Scotland and in Northern Ireland than in the UK as a whole (compare Figures 13.1.1, 13.1.2 and 13.1.3). Foster and Marrs (2012) estimated a 58% decline in the index of breeding numbers in Scotland between 1986 and 2011. Mitchell et al. (2004) stated '*Whilst the increase in the herring gull population through much of the 20th century is attributable, at least in part, to the availability of plentiful and easily accessible food supplies from artificial sources, especially refuse tips, fishery operations and sewage outlets, the decline in recent decades in the amount of food from these same sources is suspected to have contributed to the population declines witnessed since the SCR Census* [1985]'. The anticipated phased introduction of a total ban on fishery discarding in EU waters is likely to further reduce food supply for scavenging seabirds over coming years or decades, and that is likely to affect herring gulls as well as other scavenging seabirds.

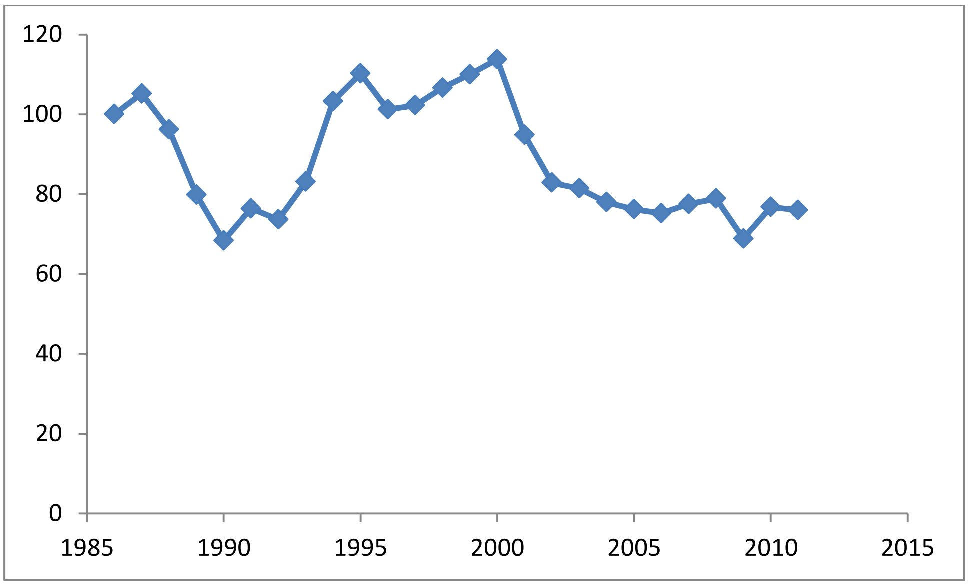

Figure 13.1.1. Herring gull breeding population index for the UK. Data from JNCC online database.

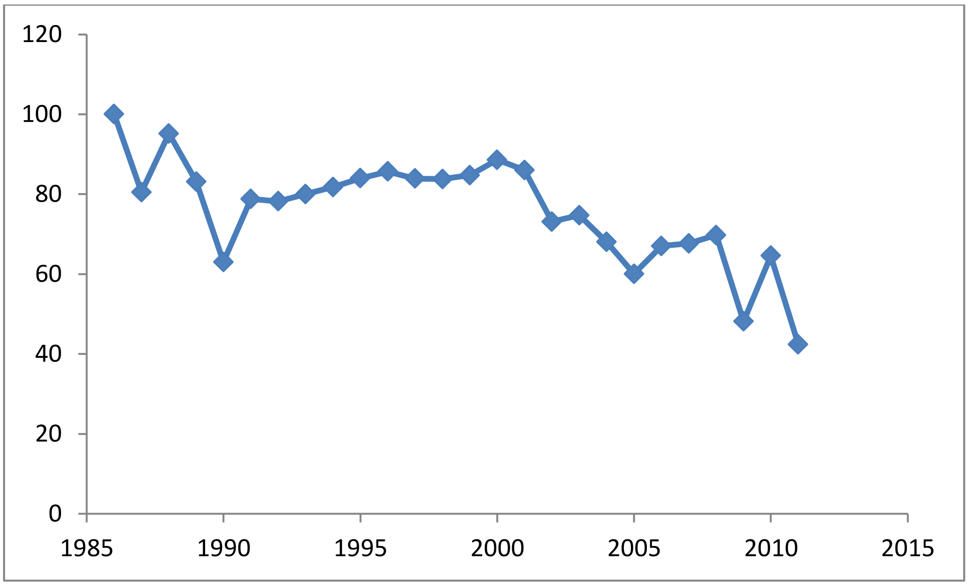

Figure 13.1.2. Herring gull breeding population index for Scotland. Data from JNCC online database.

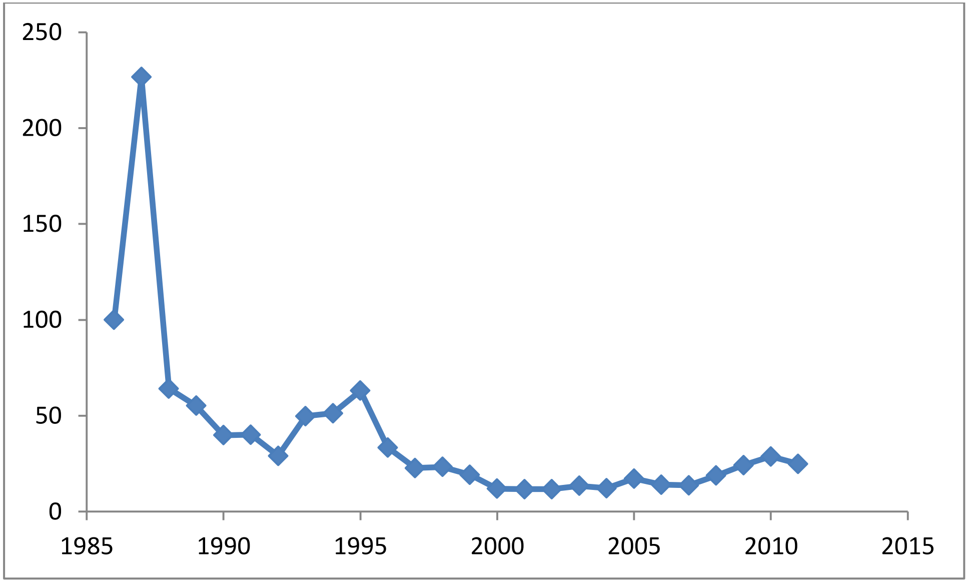

Figure 13.1.3. Herring gull breeding population index for Northern Ireland. Data from JNCC online database.

13.1.3 Factors affecting survival rates

Wanless et al. (1996) estimated adult survival rate at 0.88 (standard error 0.013) on the Isle of May for a sample of colour ringed breeding adults between 1989 and 1994, a period following a major cull that greatly reduced numbers in that colony. They found no significant annual variation in survival rates of this sample (although annual values varied between 0.84 and 0.92), so were unable to identify any environmental factors affecting survival rates. Survival of herring gulls on the Isle of May prior to culling was estimated at 0.935 (Chabrzyk and Coulson 1976), but with a large standard error (0.1) so that this value is not significantly different from that estimated by Wanless et al. (1996). Estimation of the extent to which management might alter survival rates is therefore extremely difficult.

13.1.4 Breeding success in the British Isles

In 1992, flooding by high tides reduced productivity at one colony in East Anglia, and botulism affected productivity and adult survival at one colony in Northern Ireland (Walsh et al. 1993). In 1994 in Argyll, colonies affected by mink predation produced 0.16 chicks per pair whereas colonies where mink were absent produced 0.72 chicks per pair (Walsh et al. 1995). In 1995 in Argyll, colonies affected by mink predation produced 0.18 chicks per pair whereas colonies where mink were absent produced 0.77 chicks per pair (Thompson et al. 1996). In 1996 in Argyll, colonies affected by mink predation produced 0.16-0.23 chicks per pair whereas colonies where mink were absent produced 0.58-0.79 chicks per pair (Thompson et al. 1997). In 1997 in Argyll, at nine colonies where mink were absent or controlled productivity averaged 0.96 chicks per pair, whereas at ten colonies where there was evidence of mink activity in the area, productivity averaged 0.23 chicks per pair (Thompson et al. 1998). On Skokholm in 1997, only 0.44 chicks fledged per pair where some nests were washed away by heavy seas in mid-season (Thompson et al. 1998). In 1998 in Argyll, seven colonies where mink were controlled produced 1.07 fledglings per pair, while 25 colonies with no mink control produced 0.34 fledglings per pair (Thompson et al. 1999). In 1999 at Argyll colonies where mink were known to be active, only 0.11 chicks fledged per pair, whereas at colonies where mink were controlled, productivity averaged 0.81 chicks fledged per pair (Upton et al. 2000). In 2000 at Argyll colonies 0.33 chicks per pair were produced where mink was present, whereas 1.21 chicks per pair were reared at colonies where mink were controlled (Mavor et al. 2001). At Orford Ness in 2000, 75% of 6,750 pairs of Herring gulls suffered breeding failure due to foxes (Mavor et al. 2001). Outbreaks of botulism killed chicks and adults at Ynysoedd Gwylan and at South Walney (Mavor et al. 2001). In 2001, half of the herring gull colonies monitored in Argyll that were exposed to mink produced no young at all and averaged 0.13 chicks per pair, whereas colonies where mink were controlled produced 0.83 chicks per pair (Mavor et al. 2002). Low productivity at Canna in 2001 was attributed to a reduction in fishery discards in the area (Mavor et al. 2002). In 2002, mink were considered to be reducing productivity at unprotected colonies in Argyll by about 30% compared to colonies where mink were trapped (Mavor et al. 2004). Lack of predator control at Orford Ness in 2002 resulted in Herring gull breeding numbers falling there from 6,800 pairs in 2001 to 2,575 pairs in 2002 (Mavor et al. 2003). In 2003, mink were considered to be reducing productivity at unprotected colonies in Argyll by about 25% compared to colonies where mink were trapped (Mavor et al. 2004). Mink removal at some colonies in Argyll in 2004 increased productivity to 0.9 chicks per nest compared to 0.52 at colonies where mink were not controlled, suggesting that mink reduced productivity by 42% (Mavor et al. 2005). In 2005, mink reduced productivity by about 38% (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for herring gulls as follows. Breeding success varied considerably among colonies. Experimental removal of mink from around certain colonies in Argyll indicated that mink removal boosted breeding output in this species in 2006 in Argyll by 36% relative to control sites where mink were not trapped. Breeding success of urban-nesting herring gulls tends to be high except where efforts are made to reduce numbers breeding in urban environments (Raven and Coulson 1997).

Table 13.1.1 Meta-analysis of main factors contributing to reduced productivity of Herring gulls at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Mink predation	12
Flooding	2
Botulism	2
Fox predation	2
Food shortage	1

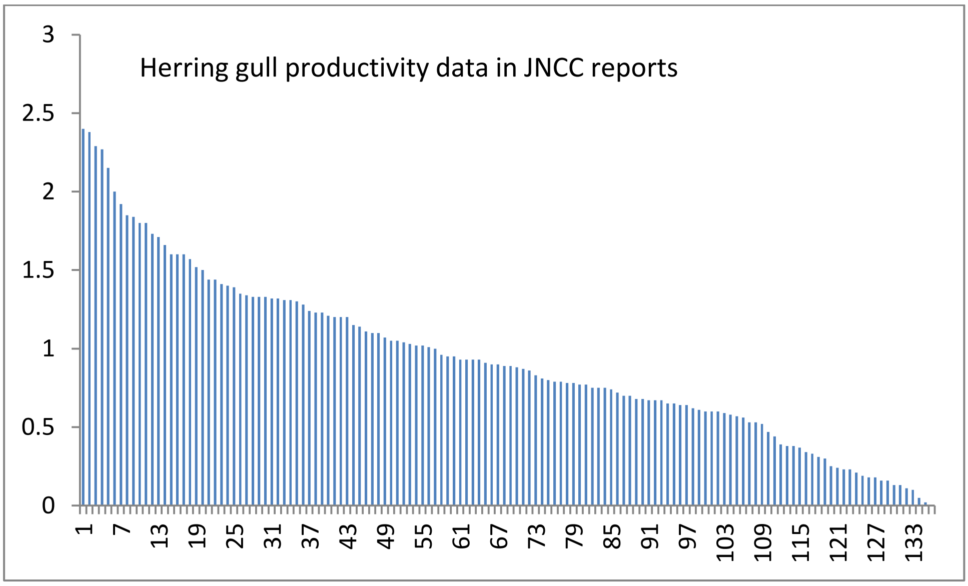

Figure 13.1.4. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of herring gulls at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

13.2 Management options

	13.2.1 Mink eradication	13.2.2 Exclusion of foxes	13.2.3 End culling	13.2.4 Rat eradication
Evidence of success for this species	High C=High*	High C=High	High C=High	High C=High
Evidence of success for similar species	High C=High	High C=High	High C=High	High C=High
Cost-effectiveness	High C=High	Moderate C=High	High C=High	High C=High
Feasibility	High C=High	Moderate C=High	Moderate C=High	High C=High
Practicality	Moderate C=High	Moderate C=High	High C=High	High C=High
Applies at SPA populations	No C=High	Some C=High	Few C=High	Few C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

13.2.1 Eradication of American mink

Mink are a major determinant of productivity at several colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of invasive alien mink would allow herring gull productivity to increase at colonies where this predator is present.

Eradication of mink from small islands can be achieved relatively easily by trapping, although eradicating mink from large areas is much more challenging. Efforts by Scottish Natural Heritage to eradicate mink from the Western Isles have been costly and protracted. However, mink swim well, and are likely to re-colonise islands that are within 2 km of habitat with mink present. Eradication of mink from mainland areas is more difficult, although there is evidence from mink eradication projects in the highlands of Scotland and in Iceland that mink can be eradicated from river catchments and in such cases re-colonisation by mink from adjacent river catchments may be inhibited by upland areas in between.

Where eradication of mink is difficult to achieve due to re-colonisation from adjacent areas, Craik (2007) showed that trapping mink in the immediate vicinity of seabird colonies in late winter and early spring normally leads to a ‘mink-free’ status for the seabird colony through the summer, resulting in considerably improved productivity (varying from year to year but on average from around 0.2 chicks per pair up to around 0.8 chicks per pair). However, trapping mink every year represents a long-term commitment that will be more expensive in the long term than eradication if that can be achieved. An alternative to annual control of mink may be mink-proof fencing to protect colonies, such that the need for annual trapping is removed once the area has been fenced and mink within the fenced area have been removed. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012) (see 13.2.2 below).

13.2.2 Exclusion of foxes from colonies

Foxes are a major determinant of productivity at several colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Foxes can be controlled on small islands by shooting, although recolonisation from the mainland may be an issue for islands situated within 1 or 2 km of the mainland. Fencing of colonies to exclude foxes would allow herring gull productivity to increase at colonies where this predator is present. In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require some maintenance. A more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka’ ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Such completely predator-proof fencing may be unnecessary to protect gull colonies from foxes, but might be appropriate for colonies subject to predation by rats or mink as well as by foxes.

By 2006, in total, around 109 km of predator-proof fencing had been erected in various areas of mainland New Zealand to exclude predators from sites with important populations of native animals and birds (Scofield et al. 2011, Innes et al. 2012, Scofield and Cullen 2012). This predator-proof fencing is expensive, costing around £100 per m to construct, and around £1 per m per year to maintain, with a life-span in New Zealand of around 25 years, so a considerable rate of depreciation (Scofield et al. 2011). However, maintenance costs and life span will depend very much on the environment where the fencing is set up. In New Zealand, where much of the fencing is in forested habitat, trees falling onto the fence can cause expensive damage, as can cyclones (Scofield et al. 2011). In the predominantly open habitat of UK seabird colonies such fencing would be under less

risk of damage, although corrosion from salt spray would be a consideration. There are several companies in New Zealand providing predator-proof fencing, for example see:
http://www.xcluder.co.nz/
http://www.meshindustries.com/Home
http://www.pestprooffences.co.nz/

There are several examples of the use of predator-proof fences to protect seabirds from mammals. A predator-proof fence completed in 2007 stretches 10.6 km across the neck of the peninsula from coast to coast at Cape Kidnappers Peninsula, North Island, New Zealand. This fence protects a privately owned and financed seabird restoration project where grey-faced petrels and Cook's petrels are being re-introduced (Cooper 2013 http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels).

A good example of successful deployment of a predator-proof fence to protect a seabird colony is one erected in 2001 to protect 36-ha on Pitt Island (Chatham Islands, New Zealand) from feral cats and pigs. Between 2002 and 2005, 200 endangered Chatham petrel chicks from the only known breeding site on South East Island (Chatham Islands) were moved into the fenced reserve. In 2012, 17 pairs from these translocated birds returned to breed, producing 15 chicks http://www.chathams.co.nz/index.php/naturalheritage/116-chatham-petrel-recovery

Cooper (2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels lists a further ten examples of deployment of predator-proof fencing around seabird colonies in New Zealand, Hawaii (USA) and Azores (Portugal).

13.2.3 End culling

Culling of breeding adult herring gulls has previously taken place at many colonies for a variety of reasons, including reduction of impacts on tern colonies and reduction in bacterial contamination of drinking water supplies. Refusal to permit large-scale culling could increase survival rates of adult herring gulls. There are additional measures taken to control impacts of gulls, including removal of nests from urban locations, and disturbance of birds by trained falcons to reduce their use of urban refuse. However, such measures are unlikely to have a significant impact on gull demography at the level of the national population.

13.2.4 Eradication of rats

Although relatively few herring gull colonies appear to be subject to rat predation impacts according to the JNCC annual reports, there is evidence that eradication of rats can increase herring gull breeding success at islands where rats are numerous. On Ailsa Craig, brown rats colonized in 1889, and there was still a colony of herring gulls present in 1990, but their breeding success was about one-third that expected, and the reduction was attributed to rats killing gull chicks (Zonfrillo 2001). Rats on Ailsa were eradicated in 1991 and herring gull breeding success improved immediately, to about three to four times the productivity experienced when rats were present (Zonfrillo 2001, and pers. comm.).

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so

£150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

At colonies that are on the mainland, or are on islands very close to the mainland so that rats would easily be able to recolonize the island, predator-proof fencing might be an alternative option (see 13.2.2.).

14. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR GREAT BLACK-BACKED GULL

14.1 Great black-backed gull ecology

The great black-backed gull is a SPEC 4 species (Favourable conservation status (secure) but concentrated in Europe). The world population is around 170,000 to 180,000 pairs, with 100,000 to 110,000 of these in Europe (excluding Russia) and around 60,000 pairs in North America (Mitchell et al. 2004). Some 40,000 pairs nest in Norway, predominantly on the Arctic north coast; many of these birds visit Britain and the North Sea for the winter (Wernham et al. 2002).

14.1.1 The species in the British Isles

Seabird 2000 found about 17,000 pairs breeding in Britain and 2,300 in Ireland (Mitchell et al. 2004). Great black-backed gulls mostly breed in small colonies or scattered pairs along suitable coastlines, although there are some large colonies. Nests are on the ground, often close to colonies of seabirds on which these gulls may feed. The clutch is usually of 2 or 3 eggs, with a mean clutch size of 2.6 eggs (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). Great black-backed gulls have a varied diet, more predatory than that of herring gulls. Birds nesting in large colonies mainly feed on fish, including fishery discards (which form a major part of their diet, especially in winter), whereas birds nesting as isolated pairs tend to feed on seabirds such as auks and kittiwakes. The UK SPA suite for this species holds about 4,400 pairs (23% of the UK total) across 6 sites: Calf of Eday, Copinsay, East Caithness Cliffs, Hoy, Isles of Scilly, and North Rona and Sula Sgeir (http://jncc.defra.gov.uk/pdf/UKSPA/UKSPA-A6-86.pdf). Adult survival rate is thought to be around 0.93 by analogy with herring and lesser black-backed gulls, and age of first breeding is around 4 or 5 years old (Cramp and Simmons 1977-1994, Glutz von Blotzheim & Bauer 1982). British great black-backed gulls rarely move more than a few tens of kilometres from their nesting sites, but large numbers of great black-backed gulls from north Norway visit the North Sea and eastern Britain during autumn and early winter. Those birds tend to return to north Norway by about February, and very rarely cross to western areas of the British Isles.

14.1.2 Present, and likely future trends

The breeding population increased during the 20th century, reaching 22,412 pairs of coastal-breeding great black-backed gulls in 1969-70 (a minimal number nesting inland were not surveyed). In 1985-88 there were 20,892 pairs, indicating a decline from what appears to have been an all-time peak in numbers around 1969. This declining trend was confirmed in Seabird 2000, when there were 19,691 coastal-breeding pairs plus 22 inland pairs (Mitchell et al. 2004). Of this total in 2000, 75% were in Scotland, 12% in Ireland, 11% in England the Isle of Man and Channel Islands, and 2% in Wales.

The increase in numbers through most of the 20th century has been attributed mainly to protection after long periods of persecution and exploitation, although this species also makes considerable use of trawl fishery discards, being able to swallow larger fish than any other UK seabird apart from gannet, and being able to rob discards from smaller scavenging seabirds (even great skuas) (Hudson and Furness 1988, 1989). Mitchell et al. (2004) suggest that fishery discards are almost certain to have influenced the continued growth in numbers of great black-backed gulls, although they point out that very little research into the demography or ecology of this species has been carried out. Hudson (1982) showed that birds at large great black-backed gull colonies in Ireland fed mainly on fishery discards while breeding, whereas pairs nesting in isolation or in small colonies fed mainly on smaller seabirds such as puffins. The same is true in Scotland, England and Wales (Poole 1995), with isolated breeding pairs also feeding on rabbits and hares as well as a wide range of seabirds (Mitchell et al. 2004).

Although in a few cases declines in breeding numbers since 1969 can be attributed to culling great black-backed gulls for the conservation of smaller seabirds, declines in breeding numbers since

1969 have occurred particularly at larger colonies, consistent with the idea that these represent density-dependent responses to reductions in fishery discard rates (which have already been occurring since the late 1960s (Votier et al. 2004)). In addition, in some northern colonies great black-backed gulls breed adjacent to great skua colonies. Increases in great skua numbers seem to have gone hand in hand with decreases in great black-backed gull numbers at such sites. A clear example of this is on Hoy, Orkney, where great black-backed gull breeding numbers fell from 3000 pairs in 1969 to 1,163 pairs in 1985 to 389 pairs in 2000, while great skua numbers increased from 72 pairs in 1969, to 1,563 in 1985 to 1,973 in 2000 (Mitchell et al. 2004).

The index of breeding numbers at UK monitored colonies (Figure 14.1.1) shows a decline from a peak reached around 2000 to 2011. However, the decline appears to be more severe in Scotland (Figure 14.1.2) than in the UK as a whole. Foster and Marrs (2012) reported a 53% decrease in an index of breeding numbers of great black-backed gulls in Scotland between 1986 and 2011. Phased elimination of discarding by EU fisheries is likely to reduce great black-backed gull numbers. This species, like the great skua, feeds extensively on discards while breeding as well as in winter. As Mitchell et al. (2004) stated '*it is probable that productivity during the breeding season and increased winter survival both increased as a consequence of feeding on discards*'. Removal of this food subsidy is likely to reverse that, leading to declines in great black-backed gull numbers, especially at the largest colonies, which are almost all SPAs for this species.

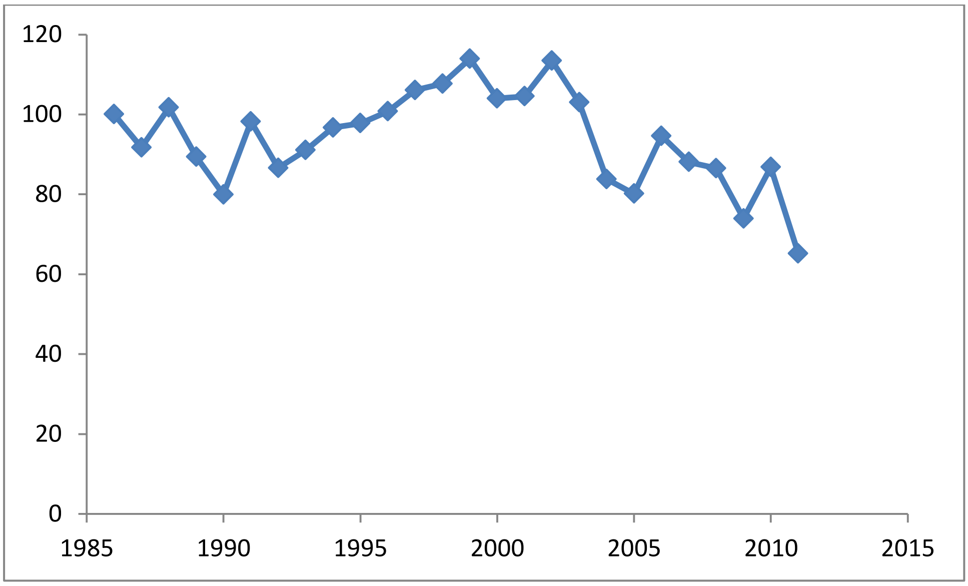

Figure 14.1.1. Great black-backed gull breeding population index for the UK. Data from JNCC online database.

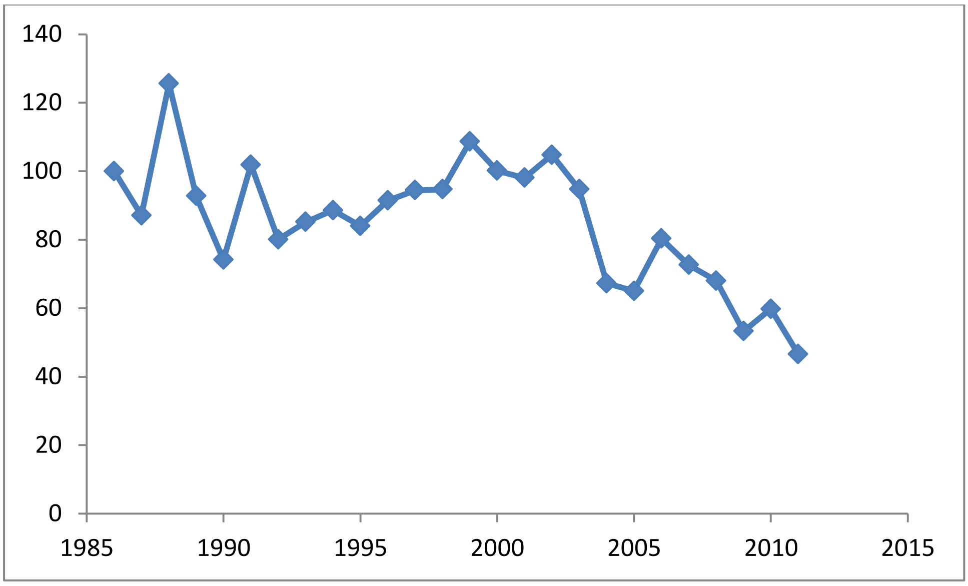

Figure 14.1.2. Great black-backed gull breeding population index for Scotland. Data from JNCC online database.

14.1.3 Factors affecting survival rates

There has been very little research into survival rates of great black-backed gulls. Adult survival rate is reported to be around 0.93 by analogy with related but slightly smaller gull species (Cramp and Simmons 1977-1994, Glutz von Blotzheim & Bauer 1982). However, factors affecting survival rates are uncertain, and have not been quantified. Estimation of the extent to which management might alter survival rates is therefore extremely difficult.

14.1.4 Breeding success in the British Isles

In relation to the 1995 breeding season, Thompson et al. (1996) commented that the two colonies with highest productivity (Isle of May and Nigg oil terminal) were those where there is protection of this species from disturbance and persecution. In 1998 in Argyll, colonies where mink were controlled produced 1.25 chicks per pair while those where there was no mink control produced 1.11 chicks per pair (Thompson et al. 1999). On Noss, a high proportion of fledglings were killed by Great skuas (Thompson et al. 1999). In 1999 in Argyll, there was complete breeding failure at 30 of 61 monitored sites, with mink apparently responsible for failure at 17 of these and possibly at another 6 (Upton et al. 2000). In 2000 in Argyll, Great black-backed gulls were monitored at 73 sites, mostly with small numbers of pairs. There was complete breeding failure at 18 sites, with mink considered responsible at 10 of these. However, breeding success overall averaged 1.17 chicks per pair (Mavor et al. 2001). In 2001, mink reduced productivity at 14 out of 55 Great black-backed gull breeding sites in Argyll with only 7 young fledging from 53 nests, whereas at all 58 sites productivity averaged 0.92 chicks per pair (Mavor et al. 2002). In 2001 at Nigg, there was evidence of mammal predation affecting productivity (Mavor et al. 2002). Mink in parts of Argyll were estimated to reduce productivity by 9% in 2004 (Mavor et al. 2005). In 2005, mink were estimated to reduce productivity by 41%, although other predators including brown rats may also have contributed to this reduction (Mavor et al. 2006). On Hoy in 2004 breeding success was very low, and chicks were underweight suggesting food shortage (Mavor et al. 2005). In 2005, complete breeding failure at the Nigg colony (for the third year in succession) was attributed to low food availability (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for great black-backed gulls as follows. Breeding success was high at some colonies but low in NW Scotland, NW England and NE Ireland. Mink control around some colonies in Argyll increased breeding success in those colonies to 0.81 chicks per nest compared to control colonies which produced 0.67 chicks per nest.

Table 14.1.1 Meta-analysis of main factors contributing to reduced productivity of Great black-backed gulls at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	**Cases reported**
Mink predation	7
Food shortage	3
Great skua predation	2
Human disturbance	1
Rat predation	1

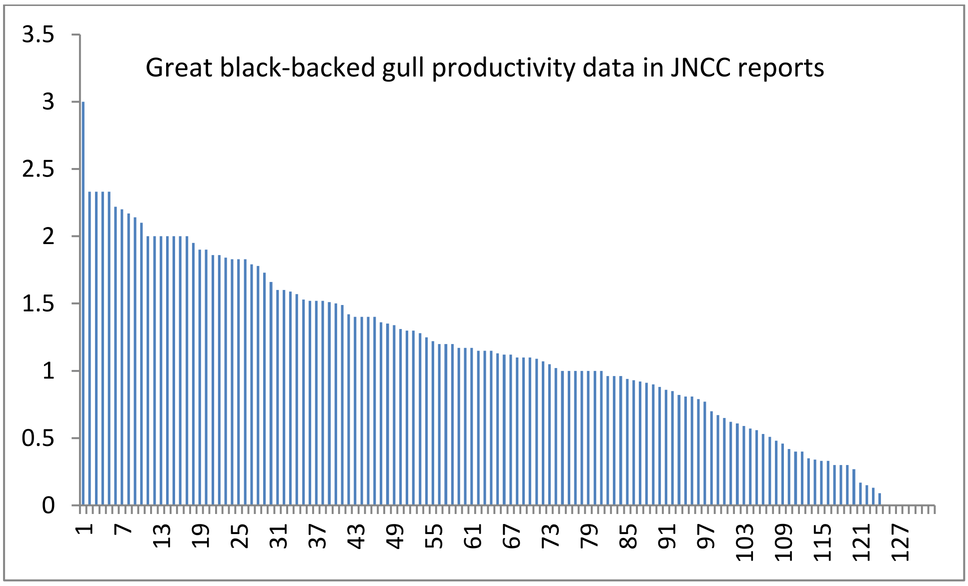

Figure 14.1.3. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of great black-backed gulls at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

14.2 Management options

	14.2.1 End culling	14.2.2 Mink eradication	14.2.3 Exclusion of foxes	14.2.4 Closure of sandeel and sprat fisheries close to colonies	14.2.5 Rat eradication
Evidence of success for this species	High C=High*	High C=High	High C=High	Low C=Low	High C=High
Evidence of success for similar species	High C=High	High C=High	High C=High	High C=Mod	High C=High
Cost-effectiveness	High C=High	High C=High	Moderate C=High	Uncertain C=Low	High C=High
Feasibility	Moderate C=High	High C=High	Moderate C=High	Moderate C=Low	High C=High
Practicality	High C=High	Moderate C=High	Moderate C=High	Moderate C=Low	High C=High
Applies at SPA populations	Few C=High	No C=High	Few C=High	Yes C=High	Few C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

14.2.1 End culling

Culling of breeding adult great black-backed gulls has previously taken place at many colonies for a variety of reasons, including reduction of impacts on tern and other seabird colonies. For example, breeding numbers on Skomer were reduced by culling from over 300 pairs in 1971 to below 40 pairs in 1990 to reduce their impact on Manx shearwaters (Poole 1995). Refusal to permit large-scale culling could increase survival rates of adult great black-backed gulls.

14.2.2 Eradication of American mink

Mink are a major determinant of productivity at several colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of invasive alien mink would allow great black-backed gull productivity to increase at colonies where this predator is present.

Eradication of mink from small islands can be achieved relatively easily by trapping, although eradicating mink from large areas is much more challenging. Efforts by Scottish Natural Heritage to eradicate mink from the Western Isles have been costly and protracted. However, mink swim well, and are likely to re-colonise islands that are within 2 km of habitat with mink present. Eradication of mink from mainland areas is more difficult, although there is evidence from mink eradication projects in the highlands of Scotland and in Iceland that mink can be eradicated from river catchments and in such cases re-colonisation by mink from adjacent river catchments may be inhibited by upland areas in between.

Where eradication of mink is difficult to achieve due to re-colonisation from adjacent areas, Craik (2007) showed that trapping mink in the immediate vicinity of seabird colonies in late winter and early spring normally leads to a ‘mink-free’ status for the seabird colony through the summer, resulting in considerably improved productivity (varying from year to year but on average from around 0.2 chicks per pair up to around 0.8 chicks per pair). However, trapping mink every year represents a long-term commitment that will be more expensive in the long term than eradication if that can be achieved. An alternative to annual control of mink may be mink-proof fencing to protect colonies, such that the need for annual trapping is removed once the area has been fenced and mink within the fenced area have been removed. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012) (see 14.2.3 below).

14.2.3 Exclusion of foxes from colonies

Foxes are a major determinant of productivity at several colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Fencing of colonies to exclude foxes would allow great black-backed gull productivity to increase at colonies where this predator is present. In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require some maintenance. A more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka’ ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Such completely predator-proof fencing may be unnecessary to protect gull colonies from foxes, but might be appropriate for colonies subject to predation by rats or mink as well as by foxes.

By 2006, in total, around 109 km of predator-proof fencing had been erected in various areas of mainland New Zealand to exclude predators from sites with important populations of native animals

and birds (Scofield et al. 2011, Innes et al. 2012, Scofield and Cullen 2012). This predator-proof fencing is expensive, costing around £100 per m to construct, and around £1 per m per year to maintain, with a life-span in New Zealand of around 25 years, so a considerable rate of depreciation (Scofield et al. 2011). However, maintenance costs and life span will depend very much on the environment where the fencing is set up. In New Zealand, where much of the fencing is in forested habitat, trees falling onto the fence can cause expensive damage, as can cyclones (Scofield et al. 2011). In the predominantly open habitat of UK seabird colonies such fencing would be under less risk of damage, although corrosion from salt spray would be a consideration. There are several companies in New Zealand providing predator-proof fencing, for example see:
http://www.xcluder.co.nz/
http://www.meshindustries.com/Home
http://www.pestprooffences.co.nz/

There are several examples of the use of predator-proof fences to protect seabirds from mammals. A predator-proof fence completed in 2007 stretches 10.6 km across the neck of the peninsula from coast to coast at Cape Kidnappers Peninsula, North Island, New Zealand. This fence protects a privately owned and financed seabird restoration project where grey-faced petrels and Cook's petrels are being re-introduced (Cooper 2013 http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels).

A good example of successful deployment of a predator-proof fence to protect a seabird colony is one erected in 2001 to protect 36-ha on Pitt Island (Chatham Islands, New Zealand) from feral cats and pigs. Between 2002 and 2005, 200 endangered Chatham petrel chicks from the only known breeding site on South East Island (Chatham Islands) were moved into the fenced reserve. In 2012, 17 pairs from these translocated birds returned to breed, producing 15 chicks http://www.chathams.co.nz/index.php/naturalheritage/116-chatham-petrel-recovery

Cooper (2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels lists a further ten examples of deployment of predator-proof fencing around seabird colonies in New Zealand, Hawaii (USA) and Azores (Portugal).

14.2.4 Closure of sandeel and sprat fishing close to breeding area SPAs

Food shortage is implicated as cause of reduced productivity at some colonies in some years. Great black-backed gulls depend more on small pelagic fish than do herring gulls. Mean foraging range of great black-backed gull is around 40 km and maximum foraging range is likely to be at least twice this distance (Ratcliffe 2000 cited in Langston 2010), so closure of sandeel and sprat fishing within 60 km of SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

14.2.5 Eradication of rats

Although relatively few great black-backed gull colonies appear to be subject to rat predation impacts according to the JNCC annual reports, there is evidence that eradication of rats can increase great black-backed gull breeding success at islands where rats are numerous. On Ailsa Craig, brown rats colonized in 1889, and there was still a colony of great black-backed gulls present in 1990, but their breeding success was about one-third that expected, and the reduction was attributed to rats killing gull chicks (Zonfrillo 2001). Rats on Ailsa were eradicated in 1991 and great black-backed gull breeding success improved immediately, to about three times the productivity experienced when rats were present (Zonfrillo 2001, and pers. comm.). Great black-backed gull breeding numbers increased, probably because rats had been eliminated from the competition for scavenging of bird corpses, so leaving more food for great black-backed gulls.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

At colonies that are on the mainland, or are on islands very close to the mainland so that rats would easily be able to recolonize the island, predator-proof fencing might be an alternative option (see 14.2.3.).

15. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR BLACK-LEGGED KITTIWAKE

15.1 Black-legged kittiwake ecology

The world population of the species is around 4.3 to 5.2 million breeding pairs (Mitchell et al. 2004), with around 2.5 to 3 million pairs breeding within the North Atlantic region (Stroud et al. 2001). The largest European populations are in Iceland (600,000 to 800,000 pairs), and Norway (770,000 pairs including Svalbard and Bear Island).

15.1.1 The species in the British Isles

Seabird 2000 found 370,000 pairs in Great Britain, the Isle of Man and Channel Islands and 49,000 pairs in Ireland (Mitchell et al. 2004). Kittiwakes mainly nest relatively low down on steep cliffs, though colonies can occur on waterside buildings and walls in places where natural cliff sites are unavailable or absent. Kittiwakes lay one to three eggs, most often two, with a mean clutch size of 2.01 in the British Isles (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). At North Sea colonies in Britain, kittiwakes feed their chicks primarily on sandeels (Furness and Tasker 2000), even at colonies where neighbouring common guillemots are simultaneously feeding chicks mainly on sprats or young herring (Lewis et al. 2001), but at colonies in the Irish Sea they may feed chicks mainly on sprats (Chivers et al. 2012). Euphausiids and other zooplankton can be important in their diet, and they will feed on small scraps of offal and discards at fishing vessels and on spilled stomach contents of larger fish, especially in winter (Garthe et al. 1996).

The UK SPA suite for this species holds around 384,000 pairs (78% of the UK total) across 33 sites: Ailsa Craig, Buchan Ness to Collieston Coast, Calf of Eday, Canna and Sanday, Cape Wrath, Copinsay, East Caithness Cliffs, Fair Isle, Farne Islands, Firth of Forth Islands, Flamborough Head and Bempton Cliffs, Flannan Isles, Foula, Fowlsheugh, Handa, Hermaness Saxa Vord and Valla Field, Hoy, Marwick Head, Mingulay and Berneray, North Caithness Cliffs, North Colonsay and Western Cliffs, North Rona and Sula Sgeir, Noss, Rathlin Island, Rousay, Rum, Shiant Isles, Skomer and Skokholm, St Abb's Head to Fast Castle, St Kilda, Sumburgh Head, Troup Pennan and Lion's Heads, West Westray. Adult survival rate averages 0.81 (del Hoyo et al. 1996) and age of first breeding is around 4 years old (Cramp and Simmons 1977-1994).

British kittiwakes mostly winter in the western North Atlantic, although a small proportion of the population may remain in British waters all year round (Frederiksen et al. 2012). Kittiwakes from other European and high latitude colonies in the North Atlantic and Barents Sea may pass through British waters on migration, but most of these birds also winter in the western North Atlantic in the same general area used by British birds (Frederiksen et al. 2012).

15.1.2 Present, and likely future trends

Kittiwake numbers increased throughout the British Isles from 1900 to around 1985. Operation Seafarer in 1969-70 found 448,000 pairs in Britain and Ireland. The SCR Census in 1985-88 found 540,000 pairs, while Seabird 2000 found 416,000 pairs (Mitchell et al. 2004). The long-term increase for most of the 20th century has been attributed in part to reduced persecution and exploitation, but changes in food supplies are also likely to have had an influence. Increases in sandeel abundance in the 1960s and 1970s followed depletion of large predatory fish such as cod and whiting, and predators on sandeel larvae and competitors for zooplankton (herring and mackerel). Kittiwakes feed primarily on sandeels while breeding, and their breeding success and survival rate are strongly influenced by sandeel stock size and by commercial fisheries on sandeels (Furness and Tasker 2000, Lewis et al. 2001a,b, Oro and Furness 2002, Mitchell et al. 2004, Frederiksen et al. 2004). Since the peak in numbers around 1985, declines in kittiwake numbers have been most severe in north Scotland (especially Shetland), associated with the collapse of sandeel stocks in the northern North Sea and consequent increases in predation impacts on kittiwakes, especially from great skuas in northern areas. JNCC monitoring data show a large decline in the index of kittiwake breeding numbers in the UK (Figure 15.1.1). However, this index includes varying trends in different national populations. In Scotland, (Figure 15.1.2) the decline is more extreme than in England (Figure 15.1.3), while in Welsh colonies the decline is smallest (Figure 15.1.4). Foster and Marrs (2012) reported a decline in the index of breeding numbers in Scotland of 66% between 1986 and 2011. The strong decline in Scotland emphasises the importance of the collapse of sandeel stocks in the northern North Sea and associated increases in predation by large gulls and great skuas.

Future prospects for kittiwakes look bleak in the north of Britain as the root of much of the problem seems to be climate change impacts on lower trophic levels affecting kittiwake food supply. In addition, a phased ban on fishery discards in EU waters will reduce food supply in the form of offal and fragments of discarded fish on which kittiwakes feed in winter though rather less in summer (Garthe et al. 1996). The discard ban will also reduce food supply to great skuas and great black-backed gulls, which is likely in the short term to cause those birds to increase depredation of kittiwakes, though in the long term may reduce numbers of these larger scavengers to more sustainable levels. There may be relatively few kittiwakes left in northern parts of the British Isles by then. In southern Britain, the situation may be less extreme, as sandeel stocks in the southern North Sea appear to be more sustainable than in the north, and possibly sprats may provide an alternative food in some areas such as the Irish Sea (possibly contributing to the lower decrease in kittiwake numbers at Welsh colonies).

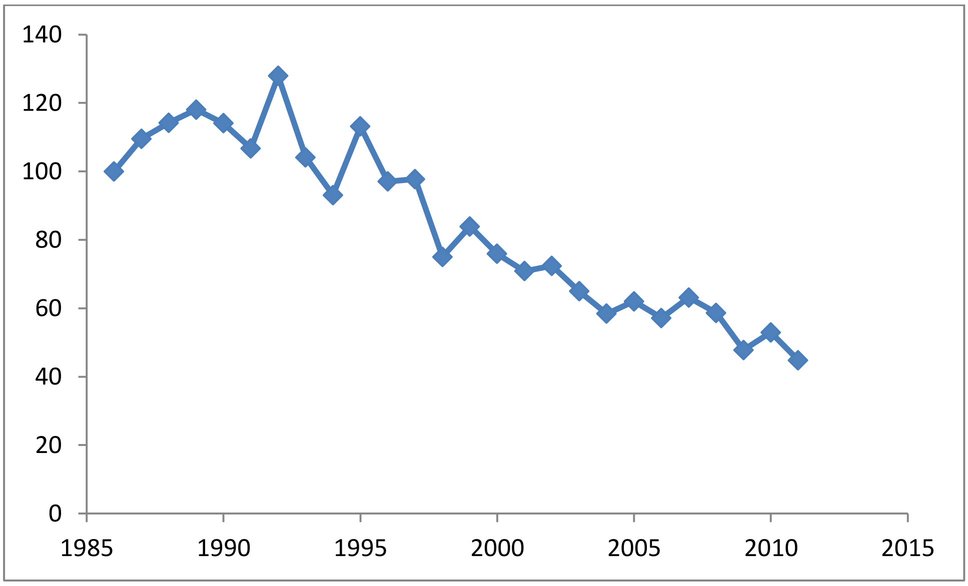

Figure 15.1.1. Kittiwake breeding population index for the UK. Data from JNCC online database.

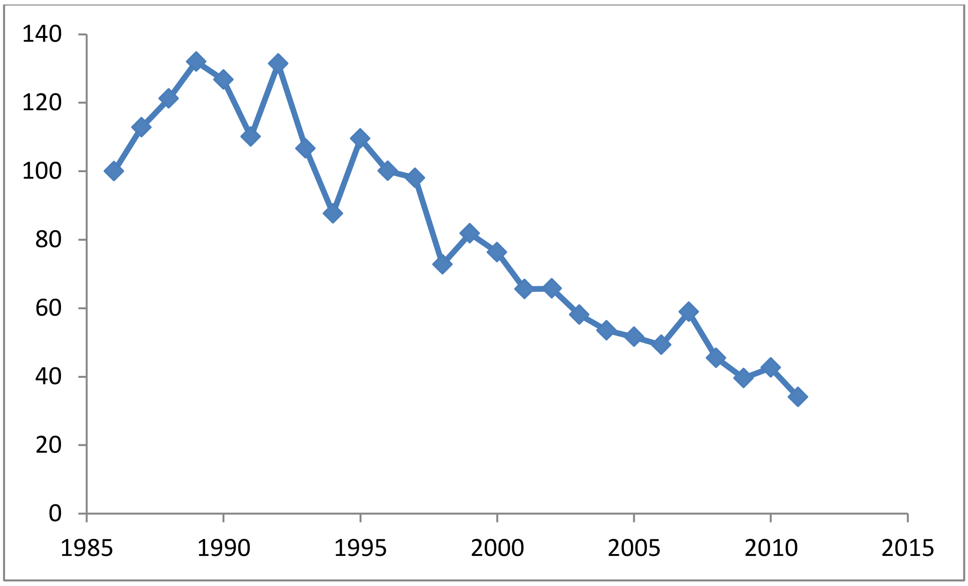

Figure 15.1.2. Kittiwake breeding population index for Scotland. Data from JNCC online database.

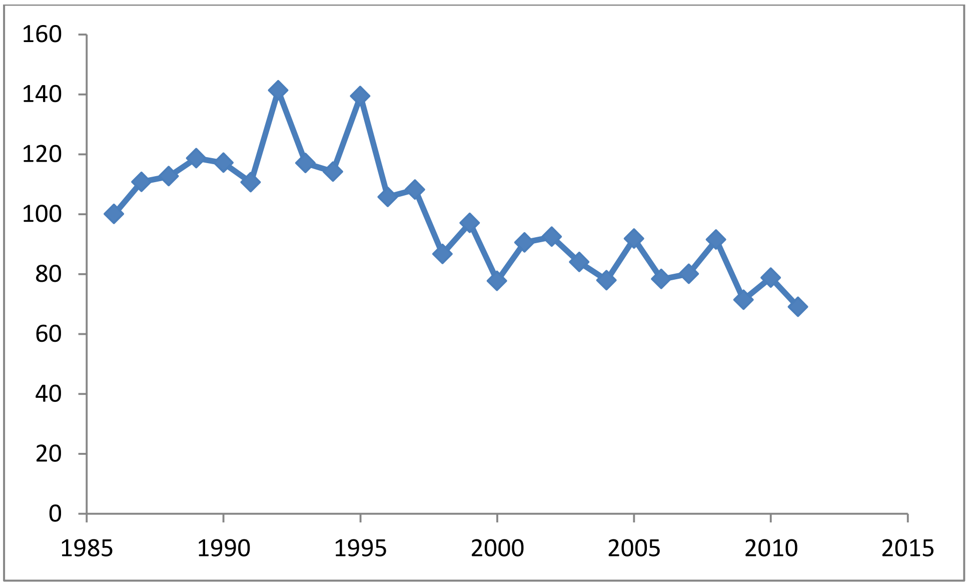

Figure 15.1.3. Kittiwake breeding population index for England. Data from JNCC online database.

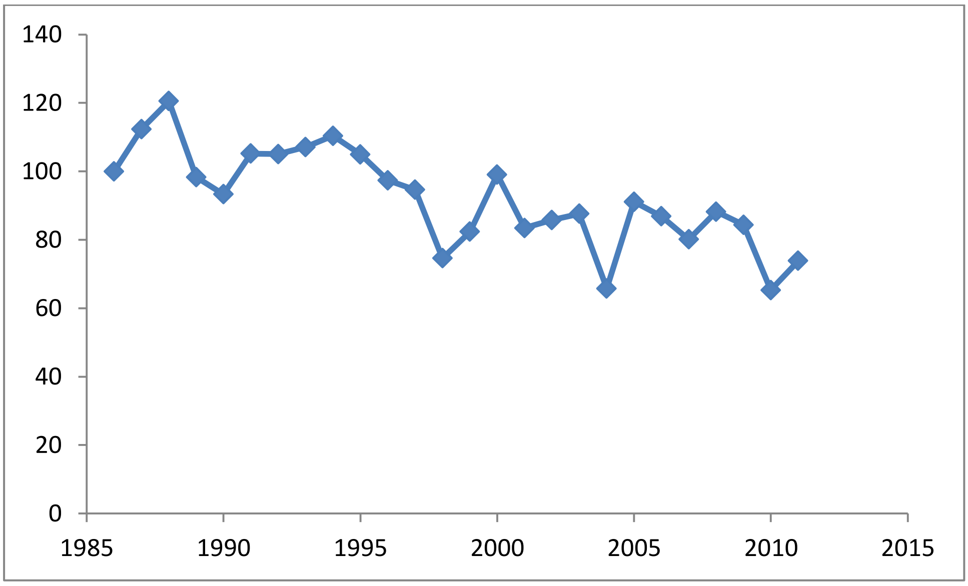

Figure 15.1.4. Kittiwake breeding population index for Wales. Data from JNCC online database.

15.1.3 Factors affecting survival rates

Several studies have reported kittiwake adult survival rates but without investigating influences on these rates of specific environmental factors (e.g. Danchin and Monnat 1992, Hatch et al. 1993 (mean 0.912 at a Pacific colony), Cam et al. 1998, Golet et al. 1998 (mean 0.922 at a Pacific colony), Coulson and Strowger 1999 (mean 0.79 at a colony in NE England), Harris et al. 2000 (0.882 at the Isle of May), Rothery et al. 2002 (mean 0.832 on Fair Isle), Lerche-Jørgensen et al. 2012) (mean 0.82 at a Danish colony)). Studying colour ringed adults in Brittany in 1980-1993, Cam et al. (1998) estimated annual survival at 0.79 for breeders and 0.65 for non-breeders; they suggested that lower survival of nonbreeders reflected their poorer individual quality or body condition. Aebischer and Coulson (1990) reported a mean survival rate of 0.8 but with variation from 0.85 in 1954-1965 to only 0.65 in 1982-85. They suggested that the decrease in survival in the 1980s might most likely be due to changes in abundance of small pelagic fish on which the kittiwakes depend. Sandvik et al. (2005) reported a mean survival rate of 0.88 for breeding adult kittiwakes colour ringed at Hornøya, north Norway, 1990-2002. In that analysis they showed that annual variation in kittiwake survival was strongly affected by pelagic fish stock biomass (herring and capelin both being important), and by sea surface temperature variation. Oro and Furness (2002) showed that kittiwake breeding adult annual survival rates at a colony in Shetland varied between 0.98 and 0.53 (with a mean of 0.8), with a strong effect of sandeel abundance and a weak influence of great skua breeding success. Survival of kittiwakes was higher when 0 group sandeel abundance was higher and was slightly reduced when great skua breeding success was higher (suggesting more predation by skuas when they had chicks to feed). Kittiwake survival increased by about 0.2 from lowest to highest observed sandeel abundance. Frederiksen et al. (2004) analysed environmental factors affecting survival rates of breeding adult kittiwakes at the Isle of May colony. They found that survival rate varied between 0.98 in 1986-87 and 0.82 in 1998-99, with 35 to 52% of the annual variation in survival rate being explained by the presence or absence of a commercial fishery for sandeels in the area and sea surface temperature (SST). Survival was lower when there was a sandeel fishery and when SST was higher. This is consistent with the fishery depleting the local sandeel stock, and with sandeel recruitment decreasing with higher SST (Arnott and Ruxton 2002). On average, kittiwake adult survival rate was reduced by about 0.05 during the period when a commercial fishery for sandeels was active in the area. The results presented by Frederiksen et al. (2004) are closely consistent with those of Oro and Furness (2002), but for kittiwakes breeding in different regions and associated with different stocks of sandeels and different commercial fisheries exploiting those stocks. These results are also consistent with changes in adult survival rates in relation to food supply reported for Arctic skua (Davis et al. 2005) and great skua (Ratcliffe et al. 2002).

15.1.4 Breeding success in the British Isles

Poor availability of sandeels in Shetland waters led to greatly reduced productivity in 1988-90 (zero at most Shetland colonies in 1988, 1989 and 1990) (Walsh et al. 1991). Kittiwake productivity in 1991 was much higher in Shetland (0.56 chicks per nest) than it had been there in 1988-90, apparently reflecting high recruitment of sandeels in 1991 (Walsh et al. 1992). In 1992, productivity averaged 0.73 chicks per nest, this improvement over previous years being attributed to increased abundance of sandeels in the northern North Sea and little impact of predation or weather (Walsh et al. 1993). In 1993, overall productivity was moderate, averaging 0.63 chicks fledged per nest. Reduction in productivity was broadly attributed to food shortage, but with some instances of predation; for example Kettla Ness colony failed completely due to predation by great skuas (Walsh et al. 1994). In 1994 productivity overall was moderately high averaging 0.72 chicks per nest, but was reduced at some Shetland colonies by Great skua predation, and at the Isles of Scilly by brown rat predation (Walsh et al. 1995). In 1994, poor weather was considered to have had a minor impact at some colonies, but food shortage was thought likely to be the most important factor determining Kittiwake productivity (Walsh et al. 1995). In 1995, poor productivity of colonies on southern Irish Sea coasts was thought to be related to low availability of food (Thompson et al. 1996). In 1996 in

Shetland, productivity was thought not to be affected by food shortage, but varied among colonies in relation to predator impacts, specifically the amount of predation by Great skuas (Thompson et al. 1997). The lowest breeding success in the west in 1996 was on St Kilda (0.62 chicks per nest) where a severe gale in late May washed nests off some study plots (Thompson et al. 1997). In 1997 in Shetland, productivity was reduced by lack of sandeels and by predation (mostly by Great skuas) (Thompson et al. 1998). Along the east coast of Britain, severe north-easterly gales with associated rain and heavy seas destroyed large numbers of Kittiwake nests at exposed colonies (Thompson et al. 1998). In 1998 in Shetland productivity (0.06 chicks per pair) was drastically reduced by scarcity of sandeels in Shetland waters leading to low colony attendance by adults, and starvation of chicks in nests in late June and early July (Thompson et al. 1999). Productivity at colonies in east Scotland and east England was very patchy, with some colonies affected by food shortage (e.g. Isle of May, where young were left unattended by adults and died) and some colonies affected by egg losses due to heavy rain (Thompson et al. 1999). On the Isles of Scilly, breeding success was only 0.14 chicks per pair, most losses being due to predation of chicks by cats (Thompson et al. 1999). In 1999, productivity averaged 0.81 chicks per nest across 42 colonies. In Shetland (average 0.74 chicks per nest) great skuas killed some chicks and fledglings (Upton et al. 2000). At Bullers of Buchan and Lowestoft, wet weather washed away some nests. At Canna, peregrines took several fledglings. At St Abbs Head, predation by mink was thought to be responsible for the loss of half of the chicks in one monitoring plot. On Tyneside, construction of an artificial tower next to a demolished mill colony attracted birds to nest on the new structure, with 65 pairs fledging an average of 1.17 chicks per nest (Upton et al. 2000). Breeding success in 2000 averaged 0.78 chicks per nest across 40 colonies, slightly above the long-term average 1986-1999 of 0.72 (Mavor et al. 2001). Productivity in 2001 was generally poor, with complete breeding failure at 8 colonies in Shetland '*undoubtedly caused by food (sandeel) shortage*' (Mavor et al. 2002). Mean body weights of adult birds caught at nests on Foula, Shetland, in 2001 were significantly lower than in previous years (Mavor et al. 2002). At St Abbs Head in 2001, mink were suspected of causing significant chick mortality (Mavor et al. 2002). Low productivity in Shetland in 2002 (0.2 chicks per nest) was attributed to scarcity of sandeels (Mavor et al. 2003). Low productivity in Shetland in 2003 was attributed to scarcity of sandeels; evidence for that included regurgitated food samples mainly comprising fishery discard fragments, and mean body weights of nesting adults being about 30g below normal (Mavor et al. 2004). In Orkney in 2003, many chicks died from mid-July onwards, but were intact in nests (i.e. not victims of predation), suggesting food shortage in Orkney waters but only late during the breeding season in contrast to the situation in Shetland (Mavor et al. 2004). In SE Scotland, kittiwakes achieved high breeding success and evidently found plenty of sandeels as these formed 91% of the diet by weight (Mavor et al.2004). In 2005, productivity was generally higher than in 2004, but in some areas food availability appeared to be low during chick rearing (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for black-legged kittiwakes as follows. Productivity at all monitored colonies in Britain and Ireland averaged 0.54 chicks per pair, below the long-term mean of 0.68 for 1986-2005. Low success in 2006 was primarily attributed to food shortage. For example, an apparent shortage of food, with resultant starvation of chicks, was noted on Noss. However, predation of chicks was considered to have reduced success at Hermaness and Foula. At North Sutor, low productivity was attributed in part to increased predation by great black-backed gulls due to low food availability. On the Isle of May, the body condition of chicks at fledging appeared to be very poor due to a rapid deterioration in feeding conditions late in the breeding season. Brood neglect was high at this time, leaving chicks exposed to weather and predators. On the Farne Islands, predation by large gulls was a problem at some kittiwake sub-colonies. At Lowestoft, many nests were destroyed early in the breeding season by foxes, although this was noted to be unusual. On Ailsa Craig, food became scarce during chick-rearing causing high mortality of chicks, with only 0.14 chicks fledged per nest, the lowest productivity at that colony since 1990. Productivity was also unusually low at Skomer in 2006, with predation by great black-backed gulls thought to be a major factor depressing productivity.

Frederiksen et al. (2004) showed that breeding success of kittiwakes on the Isle of May correlated strongly (and negatively) with sea surface temperature and was lower in years when a commercial sandeel fishery operated in the area. These two factors explained 81% of the variation in kittiwake breeding success.

Table 15.1.1 Meta-analysis of main factors contributing to reduced productivity of Kittiwakes at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Food shortage (often related to climate change)	43
Great skua predation	6
Extreme weather conditions	5
Gull predation	3
Mink predation	2
Fox predation	1
Feral cat predation	1
Rat predation	1
Peregrine predation and disturbance	1

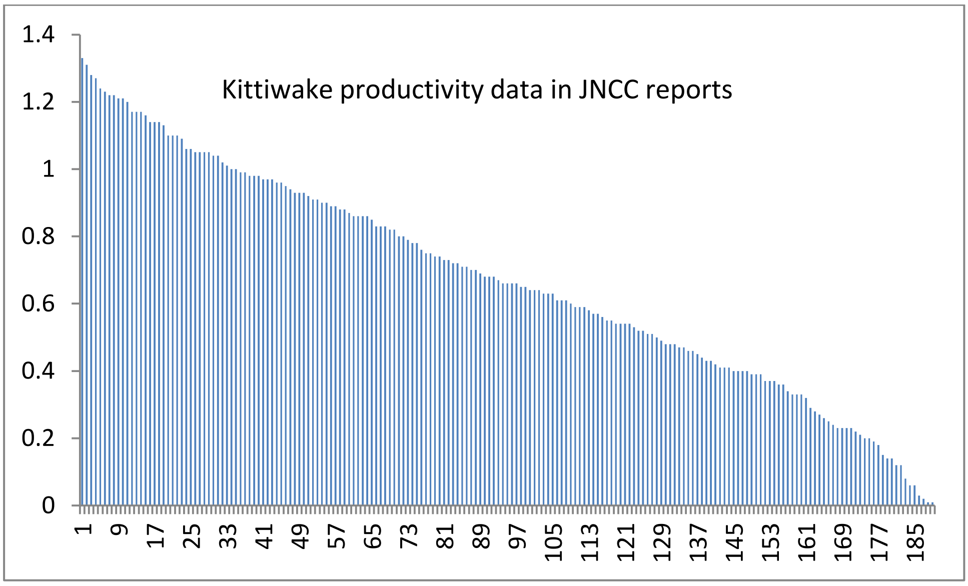

Figure 15.1.5. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of black-legged kittiwakes at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

15.2 Management options

	15.2.1 Closure of sandeel and sprat fisheries in UK waters	15.2.2 Eradicate Mink	15.2.3 Feral cat eradication	15.2.4 Rat eradication	15.2.5 Exclusion of foxes	15.2.6 Exclusion of great skuas	15.2.7 Artificial structures for colonies
Evidence of success for this species	High C=High*	Low C=Mod	Low C=Mod	Unknown C=Mod	Low C=Mod	Moderate C=Mod	High C=High
Evidence for similar species	High C=Mod	High C=High	High C=High	High C=High	Low C=High	Low C=High	High C=High
Cost-effectiveness	Uncertain C=Low	High C=High	High C=High	Low C=High	Low C=High	Moderate C=Low	Low C=High
Feasibility	Moderate C=Low	High C=High	Moderate C=High	Low C=High	Low C=High	Moderate C=High	Moderate C=High
Practicality	Moderate C=Low	Low C=High	Low C=High	Low C=High	Low C=High	Low C=High	Low C=High
Applies at SPA populations	Yes C=High	Few C=High	No C=High	Few C=High	Few C=High	Few C=High	No C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

15.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs or throughout UK waters

The single most important factor that could be influenced by management, and which reduces productivity and survival of kittiwakes, appears to be food supply (identified 43 times out of a total of 63 in the meta-analysis of JNCC monitoring data), and especially abundance of sandeels which are the main diet of breeding kittiwakes at almost all UK colonies. Mean foraging range of breeding kittiwakes is around 26 km, but maximum range is at least 100 km, possibly 200 km. Closure of sandeel fishery within 200 km of SPAs would effectively be equivalent to closure of all sandeel fishing in UK waters given the widespread distribution of kittiwake SPAs around the British Isles. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

An area off the east coast of Scotland was closed to sandeel fishing from 2000 because kittiwake breeding success in the area had fallen to very low levels after intensive fishing for sandeels for several years by the Danish industrial fleet on the sand banks close to the east coast of Scotland. The aim of this fishery closure was to allow sandeel stocks in the area to recover (which sampling of sandeels in the area confirmed to be the consequence; Greenstreet et al. 2006), and kittiwake productivity was used as the proxy for sandeel abundance in the absence of data on sandeel stock biomass (Camphuysen 2005). Frederiksen and Wanless (2006) reviewed the evidence that this closure increased productivity of kittiwakes and other seabirds. They found clear evidence that the closure resulted in increased productivity of kittiwakes within the study area compared with a control area outside the closure zone, results which agreed with the earlier findings of Frederiksen et al. 2004) that the sandeel fishery in this area reduced productivity of kittiwakes on the Isle of May during the years of the fishery. Productivity of kittiwakes did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (the difference was 0.28 chicks per nest and statistically highly significant ($p<0.0001$). Analysis of productivity data for monitored colonies of other seabirds was based on much smaller sample sizes, and showed less clear results due to chance variations. However, for Sandwich tern the pattern was similar, within the study zone there was higher productivity in no fishery years (0.62 chicks per pair) than in fishery years (0.38 chicks per pair) whereas productivity in the control area was consistently high. However, high variation and small sample size (there were only two colonies within the closure zone) resulted in this difference not being statistically significant. Data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie was more or less completely deserted from 1992-1998 and that this species did not breed on the Isle of May until 1999. Combined with the observed (non-significant) reduction in productivity in the closure zone during the fishery years, Frederiksen and Wanless (2006) concluded that '*this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes*'. For Arctic tern, European shag, common guillemot, razorbill and Atlantic puffin, the closure could not be shown to have any strong effect on productivity by analysis of data for each individual species in turn, but a one-way MANOVA with data for all species showed a significant effect of the fishery closure on productivity of the seabird community, but this was predominantly driven by the strong and clear effect on kittiwakes.

In Shetland, sandeel stock biomass fell rapidly in the late 1980s during the period of a local industrial fishery for sandeels, and that fishery was closed in 1990. Breeding success of kittiwakes in Shetland in 1990 was 0.12 chicks per pair. In 1991, the first year of the sandeel fishery closure, sandeel recruitment was higher than it had been in any of the previous seven years during heavy fishing, and kittiwake productivity in 1991 at 0.57 chicks per pair was the highest at monitored Shetland colonies since 1986 (JNCC data and ICES sandeel stock data). However, it is unclear whether the high recruitment in 1991 was caused by the fishery closure or whether the timing was coincidental.

Closure of all sandeel and sprat fisheries in UK waters would bring the UK into the same management position as exists in the USA, where fishing on the small pelagic fish such as sandeels that are keystone species for marine food webs (including large predatory fish of high commercial importance) is prohibited. The considerable body of evidence shows that kittiwake productivity and adult survival would benefit from higher average abundances of these small prey fish, and the JNCC data together with the detailed research at the Isle of May (Frederiksen et al. 2004, Frederiksen and Wanless 2006) show that the depletion of sandeel stocks has been the single most influential factor reducing productivity of kittiwakes at colonies in the British Isles. Closure of sandeel fisheres in UK waters would have little direct cost for British fishermen, since the fishery is almost entirely carried out by the Danish fishing fleet. Closure of sprat fisheries in UK waters would affect small numbers of fishermen in NW Scotland (there is a small fishery for sprats carried out by a few local fishing vessels near to Mull in Argyll and there is a small fishery for sprats carried out by a few local fishing vessels in the English Channel (ICES 2013). Closure of these fisheries would be likely to have some beneficial effects for commercially important fish stocks such as cod and whiting (Greenstreet et al. 2006).

15.2.2 Eradication of American mink

Mink are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of invasive alien mink would allow kittiwake productivity to increase at colonies where this predator is present, but evidence suggests that relatively few kittiwake colonies could benefit from mink eradication.

Eradication of mink from islands can be achieved relatively easily by trapping. However, mink swim well, and are likely to re-colonise islands that are within 2 km of habitat with mink present. Eradication of mink from mainland areas is more difficult, although there is evidence from mink eradication projects in the highlands of Scotland and in Iceland that mink can be eradicated from river catchments and in such cases re-colonisation by mink from adjacent river catchments may be inhibited by upland areas in between.

Where eradication of mink is difficult to achieve due to re-colonisation from adjacent areas, Craik (2007) showed that trapping mink in the immediate vicinity of seabird colonies in late winter and early spring normally leads to a 'mink-free' status for the seabird colony through the summer, resulting in considerably improved productivity. However, trapping mink every year represents a long-term commitment that will be more expensive in the long term than eradication if that can be achieved. An alternative to annual control of mink may be mink-proof fencing to protect colonies, such that the need for annual trapping is removed once the area has been fenced and mink within the fenced area have been removed. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012) (see 14.2.3).

15.2.3 Eradication of feral cats

Feral cats are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of feral cats would allow kittiwake productivity to increase at colonies where this predator is present, but evidence suggests that relatively few kittiwake colonies could benefit from feral cat eradication.

Eradication of feral cats can be more complex than eradication of rats, especially where there is a resident human population with pet cats, as experienced on Ascension Island (Ratcliffe et al. 2009). In that example, large numbers of pet cats were killed by the poison deployment (38% of 168 domestic cats were accidentally killed) despite efforts to keep domestic pet cats safe (which had all been neutered as part of the project). Nevertheless, that eradication successfully removed all of the feral cats and has resulted in re-establishment of seabirds that had been extirpated from the main island (Ratcliffe et al. 2009). This project cost £650,000 over three years, with 24,200 person-hrs of cat control work and 14,480 person-hrs of monitoring work on an island of 97 km^2 (9,700 ha so a cost of £67 per ha which is not far from the costs per unit area of rat eradication programmes on UK islands) but where fieldwork is made relatively easy by the climate, terrain and existence of roads (Ratcliffe et al. 2009). Cats have been successfully eradicated from many islands, the largest being Marion Island (290 km^2) in the sub-Antarctic Indian Ocean, where cats were eradicated in 2000, but this required a 15-year programme including shooting and poisoning and hunting with dogs and introduction of disease (feline panleucopaenia virus) to eradicate a population of around 3,400 cats at the start of the programme that were estimated to be killing over 450,000 seabirds per year (van Aarde 1980, Bloomer and Bester 1992, Bester et al. 2000). The second-largest island from which cats have been eradicated is Macquarie Island (120 km^2) in the sub-Antarctic which had a population of around 2,400 cats, and where the programme of trapping, hunting and deployment of the poison 1080, took 25 years before success was achieved around 2000 (Copson and Whinan 2001). Obviously these very long programmes are much more expensive, and the long timescale required can be attributed to relatively limited expertise in this type of work when those two projects started, and major logistical difficulties of working on uninhabited sub-Antarctic islands far from the mainland. Nogales et al. (2004) provide a useful tabulation of data about 48 islands where eradication of cats has been carried out and summary information on these eradication programmes, indicating that most programmes are now completed within 1-3 years, although none of the projects reviewed in that paper were in the British Isles. Although trapping, shooting, hunting with dogs, poisoning with baits and introducing disease have been the main methods used to eradicate cats, virus-vectored immune-contraception may hold promise in the near future (Courchamp and Cornell 2000).

15.2.4 Eradication of rats

Brown rats are a factor reducing productivity at a few colonies (JNCC Annual Reports on Seabird Numbers and Breeding Success). Eradication of rats would allow kittiwake productivity to increase at colonies where this predator is present, but evidence suggests that relatively few kittiwake colonies could benefit from rat eradication.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

15.2.5 Exclusion of foxes from colonies

Foxes are a factor reducing productivity at a very few colonies (JNCC Annual Reports on Seabird Numbers and Breeding Success). Fencing exclosure of foxes would allow kittiwake productivity to increase at colonies where this predator is present, but evidence suggests that very few kittiwake colonies could benefit from fencing out foxes.

Foxes can be controlled on small islands by shooting, although recolonisation from the mainland may be an issue for islands situated within 1 or 2 km of the mainland. Fencing of colonies to exclude foxes would allow kittiwake productivity to increase at colonies where this predator is present. In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require some maintenance. A more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka’ ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Such completely predator-proof fencing may be unnecessary to protect kittiwake colonies from foxes, but might be appropriate for colonies subject to predation by rats or mink as well as by foxes.

By 2006, in total, around 109 km of predator-proof fencing had been erected in various areas of mainland New Zealand to exclude predators from sites with important populations of native animals and birds (Scofield et al. 2011, Innes et al. 2012, Scofield and Cullen 2012). This predator-proof fencing is expensive, costing around £100 per m to construct, and around £1 per m per year to maintain, with a life-span in New Zealand of around 25 years, so a considerable rate of depreciation (Scofield et al. 2011). However, maintenance costs and life span will depend very much on the environment where the fencing is set up. In New Zealand, where much of the fencing is in forested habitat, trees falling onto the fence can cause expensive damage, as can cyclones (Scofield et al. 2011). In the predominantly open habitat of UK seabird colonies such fencing would be under less risk of damage, although corrosion from salt spray would be a consideration. There are several companies in New Zealand providing predator-proof fencing, for example see:
http://www.xcluder.co.nz/
http://www.meshindustries.com/Home
http://www.pestprooffences.co.nz/

There are several examples of the use of predator-proof fences to protect seabirds from mammals. A predator-proof fence completed in 2007 stretches 10.6 km across the neck of the peninsula from coast to coast at Cape Kidnappers Peninsula, North Island, New Zealand. This fence protects a privately owned and financed seabird restoration project where grey-faced petrels and Cook's petrels are being re-introduced (Cooper 2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels).

A good example of successful deployment of a predator-proof fence to protect a seabird colony is one erected in 2001 to protect 36-ha on Pitt Island (Chatham Islands, New Zealand) from feral cats and pigs. Between 2002 and 2005, 200 endangered Chatham petrel chicks from the only known breeding site on South East Island (Chatham Islands) were moved into the fenced reserve. In 2012, 17 pairs from these translocated birds returned to breed, producing 15 chicks http://www.chathams.co.nz/index.php/naturalheritage/116-chatham-petrel-recovery

Cooper (2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels lists a further ten examples of deployment of predator-proof fencing around seabird colonies in New Zealand, Hawaii (USA) and Azores (Portugal).

15.2.6 Exclude great skuas from buffer zone around kittiwake colonies

Several kittiwake colonies are affected by great skua depredations (Votier et al. 2004, 2007, 2008). Evidence indicates that the great skuas that kill kittiwakes tend to be birds nesting close to kittiwake colonies (Furness 1987, Votier et al. 2007). Removal of great skuas and prevention of great skuas establishing territories adjacent to kittiwake colonies could increase kittiwake productivity, but much of this predation stems from food shortage causing great skuas to increase depredations on other seabirds. Although this approach would probably be impossible at colonies where great skuas are an SPA feature, there are many kittiwake colonies where great skuas are not features of SPAs and are only present in relatively small numbers. Removal of great skuas from these sites could increase kittiwake survival and productivity.

15.2.7 Construction of artificial structures to support kittiwake colonies

Kittiwakes will breed on structures such as warehouses overhanging the sea, harbour walls, and even bridges over tidal rivers. So construction of artificial nesting sites for kittiwakes is possible, but in most areas of their breeding range there is no shortage of natural nesting habitat (cliffs), and not all of the potential nesting habitat is occupied, so provision of artificial cliffs would be unlikely to provide useful breeding habitat for this species and would be an expensive measure.

16. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR SANDWICH TERN

16.1 Sandwich tern ecology

Sandwich tern is an Annex 1, Schedule 1 species with SPEC 3 status (Unfavourable conservation status (declining) and concentrated in Europe). The world population is estimated at 160,000 to 170,000 breeding pairs, with three subspecies, one predominantly in Europe, one in North America, and one in South America (Mitchell et al. 2004). The nominate subspecies found in Europe and west Asia (*sandvicensis*) numbers around 90,000 to 100,000 breeding pairs (Mitchell et al. 2004).

16.1.1 The species in the British Isles

Seabird 2000 found 11,000 pairs in Great Britain the Isle of Man and Channel Islands, and 3,700 pairs in Ireland (Mitchell et al. 2004). Sandwich terns nest on the ground on low-lying offshore islets or in remote sand dunes. Most colonies are on North Sea or Irish Sea coasts, in areas with extensive sheltered shallow waters. Sandwich terns nest in a relatively small number of large and dense, highly synchronous, colonies on bare ground. Local colonies may be abandoned in response to predation pressures, competition with gulls, vegetation succession, and coastal erosion processes (Mitchell et al. 2004). The maximum clutch size is normally two eggs, and the mean clutch size is 1.6 (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). While breeding they feed predominantly on small pelagic fish, in the British Isles on sandeels, sprats and young herring. Their fishing success is severely hampered by strong winds and rough seas (Dunn 1973, Taylor 1983, Stienen et al. 2000), so they tend to feed mainly in sheltered bays and estuaries (Mitchell et al. 2004).

The GB SPA suite holds around 10,000 pairs (72% of GB population) and there are 16 sites in the UK: Alde-Ore Estuary, Carlingford Lough, Chichester and Langstone Harbours, Coquet Island, Duddon Estuary, Farne Islands, Firth of Forth Islands, Foulness, Larne Lough, Loch of Strathbeg, Morecambe Bay, North Norfolk Coast, Solent and Southampton Water, Strangford Lough, Ynys Feurig Cemlyn Bay and The Skerries, Ythan Estuary Sands of Forvie and Meikle Loch.

Adult survival rate averages 0.87 to 0.94 (Robinson 2010), and the age of first breeding is usually 3 or 4 years (Cramp and Simmons 1977-1994). British Sandwich terns migrate along coasts to winter off west Africa. Other European populations also migrate coastally to broadly the same wintering area, and while some continental birds pass through British waters on migration, most probably remain on the continental side of the North Sea.

16.1.2 Present, and likely future trends

Seabird 2000 estimated the total British and Irish population of Sandwich terns at 14,252 pairs, with 63% in England, 26% in Ireland, 7% in Scotland, and 3% in Wales (Mitchell et al. 2004). This represents a slight decline from the total of 16,047 pairs in 1985-88, but a slightly higher number than the 12,073 pairs present in 1969-70 (Mitchell et al. 2004). Although the overall population in the British Isles has remained fairly stable over recent decades, numbers at individual colonies have often changed dramatically. Decreases at particular colonies have mainly been caused by predators causing breeding failure then abandonment of particular colonies (Mitchell et al. 2004). Mitchell et al. (2004) concluded '*only colonies on [predator-free] offshore islands are immune from attack, and these are scarce within the Sandwich tern's British and Irish range. Restoration of existing offshore islands or creation of new islands from dredge-spoil may be necessary to maintain Sandwich tern populations into the 21st century*'. Changes in numbers since Seabird 2000 are indicated by the JNCC index of breeding numbers from monitoring colonies; a high proportion of the main Sandwich tern colonies is included in this index so it is likely to be reliable. In the UK as a whole, the index of Sandwich tern breeding numbers has remained fairly stable from 1986 to 2011 (Figure 16.1.1). However, the index for England suggests a slight decline in numbers there (Figure 16.1.2), so there may have been some redistribution of birds into colonies in Ireland (where there are too few data for the index to be plotted with confidence). Foster and Marrs (2012) estimated a 48% decrease in the index of breeding numbers of Sandwich terns in Scotland from 1986 to 2011, but numbers breeding in Scotland represent only 5-7% of the total in the British Isles.

Future trends are difficult to predict for this species, although the suggestion of Mitchell et al. (2004) that the species may run out of safe breeding sites in the British Isles would indicate that a decline in numbers would be more likely than an increase, in the absence of sympathetic management.

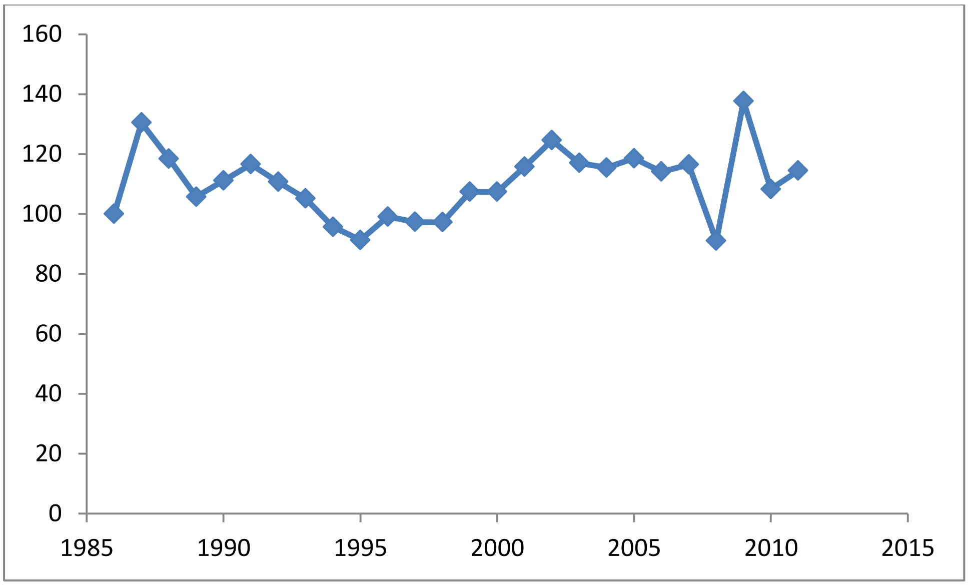

Figure 16.1.1. Sandwich tern breeding population index for the UK. Data from JNCC online database.

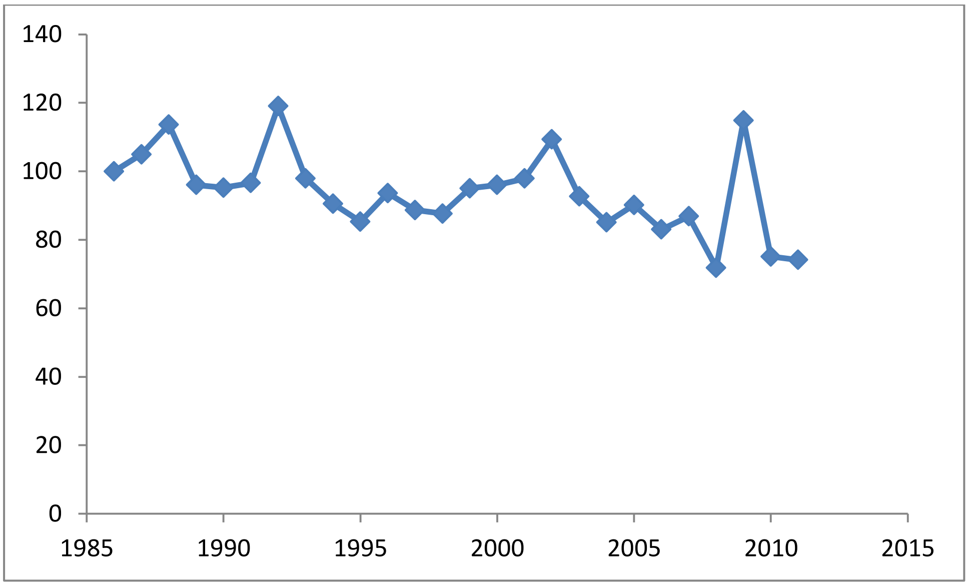

Figure 16.1.2. Sandwich tern breeding population index for England. Data from JNCC online database.

16.1.3 Factors affecting survival rates

Discounting earlier estimates of Sandwich tern survival rate which may have been biased by ring losses (Robinson 2010), the only published estimate of Sandwich tern survival rates that appears to be unbiased, estimated adult survival at 0.899 (standard error 0.029), 2nd/3rd year survival at 0.741 (s.e. 0.206) and 1st year survival at 0.313 (s.e. 0.191) (Robinson 2010). No environmental covariates were included in this analysis, so the influences of environmental factors on survival rates cannot be quantified. Estimation of the extent to which management might alter survival rates is therefore extremely difficult.

16.1.4 Breeding success in the British Isles

In 1986-1988, fox predation was identified as the main cause of reduced productivity of Sandwich terns at monitored colonies (Walsh et al. 1990). In 1989, fox predation at Scolt Head was very severe and few chicks survived (Walsh et al. 1991). In 1990, predation by foxes and flooding by high tides were identified as the two most important factors reducing Sandwich tern productivity at monitored colonies (Walsh et al. (1991). No Sandwich terns nested at Scolt Head in 1990, apparently in response to the breeding failure there in 1989 caused by fox predation (Walsh et al. 1991). In 1991, productivity averaged 0.45 chicks per pair across 16 colonies, the lowest value since monitoring started in 1986. The main impact on productivity identified was predation by foxes, which reduced productivity at Hodbarrow (520 pairs) to 0, at Foulney (332 pairs) to 0.15, at Sands of Forvie (1,115 pairs) to 0.3 (Walsh et al. 1992). In 1992, 360 pairs at Hodbarrow failed to rear any chicks as a result of depredation by foxes. Walsh et al. (1993) reported of Sandwich terns throughout the British Isles '*A major cause of breeding failure is predation by foxes, and where they are successfully excluded productivity can be greatly increased. The impact of foxes is illustrated by the situation at Hodbarrow where they have caused total breeding failure at the chick stage in the last two years. The use of electric fencing to deter foxes from entering colonies is encouraged where practicable*'. In 1993, predation reduced productivity at several colonies: foxes accounted for many of the failures at Strathbeg, and at Hodbarrow all young being raised by 100 pairs were taken by stoats. On Lindisfarne foxes depredated all 40 nests. In contrast, productivity was particularly high at Scolt Head, which was attributed to a good food supply and reduced predation by foxes. High productivity at Blakeney was attributed to a plentiful food supply. Severe weather was thought to have reduced productivity at Strathbeg and in NE Ireland (Walsh et al. 1994). In 1994, productivity was high at 0.84 across 13 colonies. Low productivity at Rye Harbour was attributed to chilling of chicks among tall vegetation during wet weather. Mink were recorded near colonies in NW Ireland, but had little impact on productivity. Foxes were noted as a likely predator in several regions (Walsh et al. 1995). In 1995, productivity averaged 0.66. Predation reduced productivity at several colonies: Havergate suffered complete failure, attributed to foxes, and foxes were also thought to be responsible for complete failure at Foulney. In contrast, at Cemlyn, stoat and crow control was carried out, and productivity was 0.78 chicks per pair (at a colony of 650 pairs). At Hodbarrow, heat stress may have contributed to reducing productivity (Thompson et al. 1996). In 1996, productivity averaged 0.68, and no specific impacts at individual colonies were identified (Thompson et al. 1997). In 1997, productivity averaged 0.55. Flooding by high tides washed out all nests at Scolt Head (productivity 0.0), and some at Blakeney (productivity 0.5). Productivity was reduced by mink and badgers at Dungeness (productivity 0.0), and by badgers at Rye (productivity 0.0). In 1998, productivity averaged 0.49, well below average. A few colonies were affected by bad weather: Scolt Head failed completely due to flooding by high tides, while wet and windy weather at Loch Swilly reduced foraging ability of adults but only reduced productivity slightly (0.88). Several colonies were affected by predators: herring gulls took chicks at Scolt Head, peregrines and large gulls affected productivity slightly at Anglesey (0.98 compared to 1.14 and 1.33 in the two previous years), and rat predation affected the colony at Lady's Island Lake (Thompson et al. 1999). In 1999, mean productivity was 0.53 chicks per pair, lower than in most years. At the Farnes, food shortage resulted in the colony of 1,946 pairs rearing only 292 chicks (0.15). Gull predation reduced productivity on the Isle of May, cold weather was blamed for low productivity at Loch Ryan, and grazing deer were thought to be responsible for extensive damage to eggs on Brownsea Island where productivity was only 0.01 chicks per pair (Mavor et al. 2000). In 2000, productivity was generally good (1 chick per pair). At Loch Ryan, despite some losses to flooding, productivity (0.92) was close to average. There was no sign of food shortage at the Farnes where productivity was 0.56, higher than in the previous year when food had been scarce. Herring gull predation at Blakeney resulted in productivity of 0.0 there (Mavor et al. 2001). In 2001, productivity was close to average in most colonies, but an estimated 50% of chicks at the Farnes died during poor weather in mid-June, and many of the survivors were depredated by gulls (Mavor et al. 2002). In 2002, overall productivity (0.69) was marginally below the long-term average for 1986-2001 (0.71). The colony at Loch Ryan failed (0.0) due to flooding. Lindisfarne failed (0.0) due to oystercatcher predation of eggs. Brownsea failed due to predation (but predator species was not reported) (Mavor et al. 2003). In 2003, mean productivity of 0.8 chicks per pair across 15 colonies was above average. However, productivity at Sands of Forvie (0.71) was reduced by stoat and gull predation, while it was reduced at Blakeney (0.79) by rat predation, and at Langstone Harbour (0.39) by food shortage, while flooding depressed productivity at Lough Swilly and mink reduced breeding success to only 0.12 chicks per pair at Mulroy Bay (Mavor et al. 2004). In contrast, productivity was at least 1 chick per pair at Loch Ryan, Holkham, Rye Harbour, Anglesey, and Hodbarrow, sites where no problems with weather, food supply or predation were reported (Mavor et al. 2004). In 2004, mean productivity across monitored colonies was 0.52 chicks per pair, the lowest for 12 years (Mavor et al. 2005). Breeding success in NE England was depressed by storms in June, while chicks that survived were severely affected by shortage of food (Mavor et al. 2005). In E England colonies were also affected by storms and by food shortage. Predation by foxes resulted in breeding failure at North Solent. In SW England food was plentiful and productivity was high although bad weather and predation accounted for some losses (Mavor et al. 2005). In 2005, productivity was reduced at Sands of Forvie by a presumed shortage of sandeels resulting in only 22 fledged young from 570 pairs, and presumed food shortages were blamed for low productivity at the two main colonies in NE England (Mavor et al. 2006). Predation depressed productivity at Blakeney, bad weather and food shortage reduced productivity at Langstone Harbour, while mink predation reduced productivity at Mulroy Bay to half the normal level (Mavor et al. 2006). In 2006, Sandwich tern productivity averaged 0.77 chicks per pair across twelve colonies, above the long term average, but predators reduced breeding success at Langstone Harbour to the lowest level for four years, and lesser black-backed gulls reduced productivity at Hodbarrow to well below average by eating tern chicks (Mavor et al. 2008). Human recreation activities (and dogs) can cause inadvertent disturbance to tern colonies accessible to walkers, although such disturbance was not identified at any of the colonies monitored in the JNCC annual monitoring of seabird numbers and breeding success. Human disturbance can act in combination with other factors, for example pushing colonies down the beach towards high tide line and therefore increasing risk of flooding or exposure to predation.

Table 16.1.1 Meta-analysis of main factors contributing to reduced productivity of Sandwich terns at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Fox predation	23
Tidal flooding	11
Food shortage	10
Gull predation	9
Extreme weather conditions	9
Mink predation	5
Rat predation	3
Stoat predation	3
Badger predation	2
Crow predation	2
Oystercatcher predation	1
Peregrine predation and disturbance	1
Heat stress	1
Deer	1

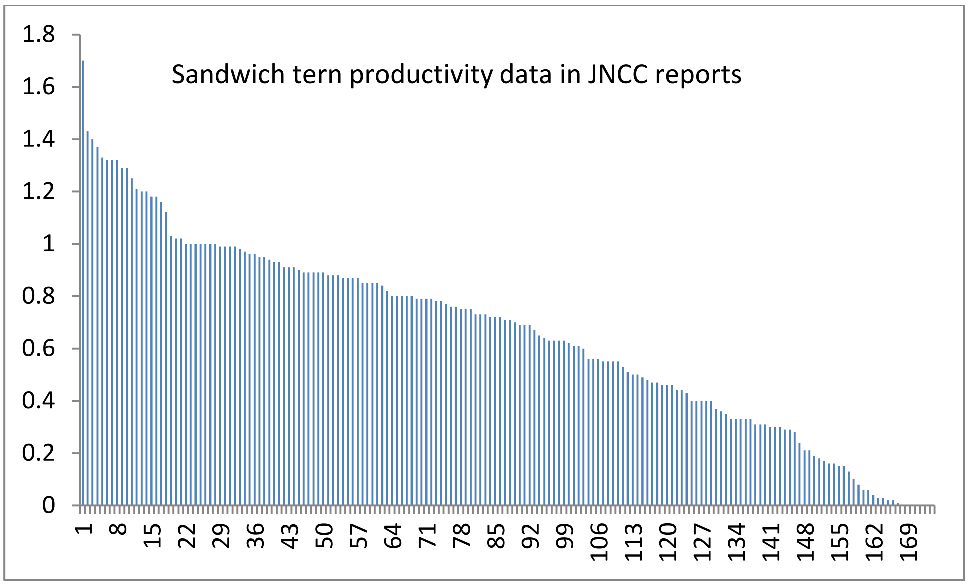

Figure 16.1.3. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of Sandwich terns at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

16.2 Management options

	16.2.1 Closure of sandeel and sprat fisheries close to colonies	16.2.2 Eradicate Mink	16.2.3 Eradicate Feral cats	16.2.4 Eradicate Rats	16.2.5 Exclude foxes	16.2.6 Control stoats	16.2.7 Flood control	16.2.8 Exclude large gulls
Evidence of success for this species	Moderate C=Low	High C=High	Low C=Low	Low C=Mod	High C=High	High C=High	High C=High	Mod C=Low
Evidence of success for similar species	High C=Mod	High C=High	High C=High	High C=High	High C=High	High C=High	High C=High	Mod C=Low
Cost-effectiveness	Uncertain C=Low	High C=High	High C=High	High C=High	Moderate C=High	Moderate C=High	High C=Mod	Mod C=Low
Feasibility	Moderat C=Low	High C=High	Moderate C=High	High C=High	Moderate C=High	High C=High	High C=High	Mod C=Low
Practicality	Moderat C=Low	Moderat C=High	Moderate C=High	Low C=High	Moderate C=High	High C=High	Moderate C=High	High C=Mod
Applies at SPA populations	Yes C=High	No C=High	Few C=High	Few C=High	Some C=High	Few C=High	Some C=High	Yes C=High

*Confidence in each score is listed as ‘High’ ‘Moderate’ or ‘Low’ based on amount and consistency of evidence

16.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs

Food shortage is implicated as a cause of reduced productivity at some colonies in some years. Mean foraging range of Sandwich terns is around 15 km and maximum foraging range is up to 70 km (Langston 2010), so closure of sandeel and sprat fishing within 60 km of SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider.

However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

An area off the east coast of Scotland was closed to sandeel fishing from 2000 because kittiwake breeding success in the area had fallen to very low levels after intensive fishing for sandeels for several years by the Danish industrial fleet on the sand banks close to the east coast of Scotland. The aim of this fishery closure was to allow sandeel stocks in the area to recover (which sampling of sandeels in the area confirmed to be the consequence; Greenstreet et al. 2006), and kittiwake productivity was used as the proxy for sandeel abundance in the absence of data on sandeel stock biomass (Camphuysen 2005). Frederiksen and Wanless (2006) reviewed the evidence that this closure increased productivity of kittiwakes and other seabirds. They found clear evidence that the closure resulted in increased productivity of kittiwakes within the study area compared with a control area outside the closure zone, results which agreed with the earlier findings of Frederiksen et al. 2004) that the sandeel fishery in this area reduced productivity of kittiwakes on the Isle of May during the years of the fishery. Productivity of kittiwakes did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (the difference was 0.28 chicks per nest and statistically highly significant ($p<0.0001$). Analysis of productivity data for monitored colonies of other seabirds was based on much smaller sample sizes, and showed less clear results due to chance variations. However, for Sandwich tern the pattern was similar, within the study zone there was higher productivity in no fishery years (0.62 chicks per pair) than in fishery years (0.38 chicks per pair) whereas productivity in the control area was consistently high. However, high variation and small sample size (there were only two colonies within the closure zone) resulted in this difference not being statistically significant. Data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie was more or less completely deserted from 1992-1998 and that this species did not breed on the Isle of May until 1999. Combined with the observed (non-significant) reduction in productivity in the closure zone during the fishery years, Frederiksen and Wanless (2006) concluded that '*this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes*'. For Arctic tern, European shag, common guillemot, razorbill and Atlantic puffin, the closure could not be shown to have any strong effect on productivity by analysis of data for each individual species in turn, but a one-way MANOVA with data for all species showed a significant effect of the fishery closure on productivity of the seabird community, but this was predominantly driven by the strong and clear effect on kittiwakes.

In Shetland, sandeel stock biomass fell rapidly in the late 1980s during the period of a local industrial fishery for sandeels, and that fishery was closed in 1990. Breeding success of kittiwakes in Shetland in 1990 was 0.12 chicks per pair. In 1991, the first year of the sandeel fishery closure, sandeel recruitment was higher than it had been in any of the previous seven years during heavy fishing, and kittiwake productivity in 1991 at 0.57 chicks per pair was the highest at monitored Shetland colonies since 1986 (JNCC data and ICES sandeel stock data). However, it is unclear whether the high recruitment in 1991 was caused by the fishery closure or whether the timing was coincidental. No Sandwich terns breed in Shetland, so this case study does not provide evidence specifically for Sandwich terns, but the principles are thought to apply.

Closure of all sandeel and sprat fisheries in UK waters would bring the UK into the same management position as exists in the USA, where fishing on the small pelagic fish such as sandeels that are keystone species for marine food webs (including large predatory fish of high commercial importance) is prohibited. The body of strongly suggests that Sandwich tern productivity and adult survival would benefit from higher average abundances of these small prey fish. Closure of sandeel fisheres in UK waters would have little direct cost for British fishermen, since the fishery is almost entirely carried out by the Danish fishing fleet. Closure of sprat fisheries in UK waters would affect small numbers of fishermen in NW Scotland (there is a small fishery for sprats carried out by a few local fishing vessels near to the isle of Mull in Argyll and there is a small fishery for sprats carried out by a few local fishing vessels in the English Channel (ICES 2013). Closure of these fisheries would be likely to have some beneficial effects for commercially important fish stocks such as cod and whiting (Greenstreet et al. 2006).

16.2.2 Eradication of American mink

Mink are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of invasive alien mink would allow Sandwich tern productivity to increase at colonies where this predator is present.

Eradication of mink from islands can be achieved relatively easily by trapping. However, mink swim well, and are likely to re-colonise islands that are within 2 km of habitat with mink present (and most Sandwich tern colonies are either on islands very close to the mainland or are on the mainland). Eradication of mink from mainland areas is more difficult, although there is evidence from mink eradication projects in the highlands of Scotland and in Iceland that mink can be eradicated from river catchments and in such cases re-colonisation by mink from adjacent river catchments may be inhibited by upland areas in between.

Where eradication of mink is difficult to achieve due to re-colonisation from adjacent areas, Craik (2007) showed that trapping mink in the immediate vicinity of seabird colonies in late winter and early spring normally leads to a ‘mink-free’ status for the seabird colony through the summer, resulting in considerably improved productivity of gulls (from around 0.2 chicks per pair up to around 0.8 chicks per pair). However, trapping mink every year represents a long-term commitment that will be more expensive in the long term than eradication if that can be achieved. An alternative to annual control of mink may be mink-proof fencing to protect colonies, such that the need for annual trapping is removed once the area has been fenced and mink within the fenced area have been removed. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012) (see 16.2.5 below).

16.2.3 Eradication of feral cats

Feral cats are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of feral cats would allow Sandwich tern productivity to increase at colonies where this predator is present.

Eradication of feral cats can be more complex than eradication of rats, especially where there is a resident human population with pet cats, as experienced on Ascension Island (Ratcliffe et al. 2009). In that example, large numbers of pet cats were killed by the poison deployment (38% of 168 domestic cats were accidentally killed) despite efforts to keep domestic pet cats safe (which had all been neutered as part of the project). Nevertheless, that eradication successfully removed all of the feral cats and has resulted in re-establishment of seabirds that had been extirpated from the main island (Ratcliffe et al. 2009). This project cost £650,000 over three years, with 24,200 person-hrs of cat control work and 14,480 person-hrs of monitoring work on an island of 97 km^2 (9,700 ha so a cost of £67 per ha which is not far from the costs per unit area of rat eradication programmes on UK islands) but where fieldwork is made relatively easy by the climate, terrain and existence of roads (Ratcliffe et al. 2009). Cats have been successfully eradicated from many islands, the largest being Marion Island (290 km^2) in the sub-Antarctic Indian Ocean, where cats were eradicated in 2000, but this required a 15-year programme including shooting and poisoning and hunting with dogs and introduction of disease (feline panleucopaenia virus) to eradicate a population of around 3,400 cats at the start of the programme that were estimated to be killing over 450,000 seabirds per year (van Aarde 1980, Bloomer and Bester 1992, Bester et al. 2000). The second-largest island from which cats have been eradicated is Macquarie Island (120 km^2) in the sub-Antarctic which had a population of around 2,400 cats, and where the programme of trapping, hunting and deployment of the poison 1080, took 25 years before success was achieved around 2000 (Copson and Whinan 2001). Obviously these very long programmes are much more expensive, and the long timescale required can be attributed to relatively limited expertise in this type of work when those two projects started, and major logistical difficulties of working on uninhabited sub-Antarctic islands far from the mainland. Nogales et al. (2004) provide a useful tabulation of data about 48 islands where eradication of cats has been carried out and summary information on these eradication programmes, indicating that most programmes are now completed within 1-3 years, although none of the projects reviewed in that paper were in the British Isles. Although trapping, shooting, hunting with dogs, poisoning with baits and introducing disease have been the main methods used to eradicate cats, virus-vectored immune-contraception may hold promise in the near future (Courchamp and Cornell 2000). An alternative to eradicating feral cats may be cat-proof fencing to protect colonies. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012) (see 16.2.5 below).

16.2.4 Eradication of rats

Brown rats are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of invasive alien rats would allow Sandwich tern productivity to increase at colonies where this predator is present.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

An alternative to eradicating rats that would probably be more appropriate where colonies are on mainland sites or on islands close to the mainland coast, may be rat-proof fencing to protect colonies. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012) (see 16.2.5 below).

16.2.5 Exclusion of foxes from colonies

Foxes are a factor reducing productivity at several colonies including important SPA populations, and their depredations appear to reduce adult survival although that impact has not been quantified. Exclosure fencing to keep foxes out of colonies would allow Sandwich tern productivity to increase at colonies where this predator is present.

Foxes can be controlled on small islands by shooting, although recolonisation from the mainland may be an issue for islands situated within 1 or 2 km of the mainland. Fencing of colonies to exclude foxes would allow Sandwich tern productivity to increase at colonies where this predator is present.

In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require some maintenance. A more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka' ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Such completely predator-proof fencing may be unnecessary to protect colonies just from foxes, but might be especially appropriate for colonies subject to predation by rats or mink as well as by foxes.

By 2006, in total, around 109 km of predator-proof fencing had been erected in various areas of mainland New Zealand to exclude predators from sites with important populations of native animals and birds (Scofield et al. 2011, Innes et al. 2012, Scofield and Cullen 2012). This predator-proof fencing is remarkably expensive, costing around £100 per m to construct, and around £1 per m per year to maintain, with a life-span in New Zealand of around 25 years, so a considerable rate of depreciation (Scofield et al. 2011). However, maintenance costs and life span will depend very much on the environment where the fencing is set up. In New Zealand, where much of the fencing is in forested habitat, trees falling onto the fence can cause expensive damage, as can cyclones (Scofield et al. 2011). In the predominantly open habitat of UK seabird colonies such fencing would be under less risk of damage, although corrosion from salt spray would be a consideration. There are several companies in New Zealand providing predator-proof fencing, for example see:
http://www.xcluder.co.nz/
http://www.meshindustries.com/Home
http://www.pestprooffences.co.nz/

There are several examples of the use of predator-proof fences to protect seabirds from mammals. A predator-proof fence completed in 2007 stretches 10.6 km across the neck of the peninsula from coast to coast at Cape Kidnappers Peninsula, North Island, New Zealand. This fence protects a privately owned and financed seabird restoration project where grey-faced petrels and Cook's petrels are being re-introduced (Cooper 2013 http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels).

A good example of successful deployment of a predator-proof fence to protect a seabird colony is one erected in 2001 to protect 36-ha on Pitt Island (Chatham Islands, New Zealand) from feral cats and pigs. Between 2002 and 2005, 200 endangered Chatham petrel chicks from the only known breeding site on South East Island (Chatham Islands) were moved into the fenced reserve. In 2012, 17 pairs from these translocated birds returned to breed, producing 15 chicks http://www.chathams.co.nz/index.php/naturalheritage/116-chatham-petrel-recovery

Cooper (2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels lists a further ten examples of deployment of predator-proof fencing around seabird colonies in New Zealand, Hawaii (USA) and Azores (Portugal).

16.2.6 Control of stoats close to colonies

Stoats are a factor reducing productivity at several colonies including important SPA populations, and their depredations appear to reduce adult survival although that impact has not been quantified. It may be necessary to deploy trapping to reduce stoat numbers at colonies or to eradicate them from islands. However, a more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka’ ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Deploying such predator-proof fencing around colonies affected by stoats would allow Sandwich tern productivity to increase at colonies where this predator is currently present and reducing productivity.

16.2.7 Protection of colonies from flooding or engineering of new nesting habitat in safer locations

Flooding (by unusually high tides and storm-driven waves) has been identified as a factor reducing Sandwich tern productivity in some years at some colonies (the JNCC meta-analysis indicated this to be the main impact in at least 11 cases out of about 90 and to be the second most frequently identified impact). Engineering work to improve sea defences at colonies and to increase the amount of suitable nesting habitat in areas less at risk from flooding (because Sandwich terns need bare ground for nesting and are displaced from some nesting areas by excessive growth of vegetation) could greatly increase productivity by providing safer nesting sites. Costs would be extremely site-specific, but this seems a likely approach to identify since Sandwich terns select areas of bare ground for nesting and such sites tend to degrade due to natural processes of erosion or colonisation by vegetation. Creation of bare ground in well protected areas away from flooding risk would therefore seem to be a cost effective approach to increasing Sandwich tern productivity at some colonies. Such engineering work may need to be carried out every few years (outside the tern breeding season), depending on the mobility of coastal habitat and the rate of vegetation development on specific sites. Sites where such engineering work might be most cost-effective should be selected on the recent history of flooding and the ease with which engineering might be carried out. It would be important to monitor the effectiveness of such management, in relation to the limited plasticity of colony site characteristics and population distribution.

16.2.8 Exclude large gulls from nesting close to colonies

Predation by large gulls affects productivity at some colonies in some years (this was identified by the meta-analysis of JNCC data to be the third most frequently identified cause of reduced productivity, reported 9 times). Since there are relatively few important Sandwich tern colonies, removal of gull territories adjacent to these key colonies could improve productivity of Sandwich terns. It is likely that most of the Sandwich tern colonies affected by gull predation are affected by small numbers of gulls that have developed specialised habits of feeding on tern eggs and chicks, and so the numbers of gulls that would need to be removed to increase tern productivity are likely to be small.

17. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR COMMON TERN

17.1 Common tern ecology

Common tern is listed on Annex 1 and Schedule 1. The world population is around 460,000 to 620,000 pairs, in three subspecies, *hirundo*, *longipennis*, and *tibetana*. The nominate subspecies, *hirundo*, is widely distributed in Europe, the Middle East, west Asia, and North America. Within Europe the population is around 220,000 to 340,000 pairs (Stroud et al. 2001).

17.1.1 The species in the British Isles

Seabird 2000 found 10,000 pairs of common terns in Great Britain the Isle of Man and the Channel Islands, and 4,200 pairs in Ireland (Mitchell et al. 2004). Common terns nest on the ground in colonies that vary in size from a few pairs up to many hundreds of pairs, but very rarely reaching 1,000 pairs as birds feed on fish within a few km of the colony so large colonies become limited by local food supply. Colonies are usually on islets close to the shore, but can be on man-made structures including flat rooftops, floating platforms, derelict jetties, abandoned lock gates. Common terns nest on man-made structures more than any other seabird in the British Isles. Breeding sites are often abandoned if mammal predators can gain access, or if breeding numbers of gulls increase. Most pairs lay two or three eggs, the mean clutch size being 2.6 (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). Common terns feed on small fish, but may take a wider range of fish species than caught by Arctic terns or Sandwich terns. Nevertheless, most colonies depend on local stocks of either sandeels, or sprats or young herring. Common terns tend to favour more sheltered feeding areas than Arctic terns, and their fishing success is reduced by strong wind or rough sea (Cramp and Simmons 1977-1994).

The GB SPA suite holds around 5,700 pairs of common terns (46% of the population). The UK SPA suite holds 6,993 pairs, across 22 sites: Breydon Water, Carlingford Lough, Coquet Island, Cromarty Firth, Dungeness to Pett Level, Farne Islands, Firth of Forth Islands, Foulness, Glas Eileanan, Inner Moray Firth, Larne Lough, Lough Neagh and Lough Beg, Monach Isles, North Norfolk Coast, Poole Harbour, Ribble and Alt Estuaries, Solent and Southampton Water, Strangford Lough, The Dee Estuary, The Wash, Ynys Fenrig Cemlyn Bay and The Skerries, Ythan Estuary Sands of Forvie and Meikle Loch.

Adult survival rate averages 0.88 (del Hoyo et al. 1992-2006), and age of first breeding is usually 3 or 4 years old (Cramp and Simmons 1977-1994). British common terns migrate along the coast to winter off west Africa. Continental populations of common terns winter in a broadly similar area, and while some continental birds may migrate through British waters, most probably pass southwards along the eastern side of the North Sea.

17.1.2 Present, and likely future trends

Seabird 2000 estimated the population of common terns in Britain and Ireland at about 14,500 pairs, with 33% in England and the Channel Islands, 33% in Scotland, 29% in Ireland, and 5% in Wales (Mitchell et al. 2004). This total is only 2% lower that the numbers found in the SCR Census in 1985-88 and in Operation Seafarer in 1969-70. However, the apparent stability hides major changes in distribution.

Although relatively stable in the UK as a whole (Figure 17.1.1), numbers have decreased in Scotland (Figure 17.1.2), but increased since 1986 in England (Figure 17.1.3) and probably also in Ireland (where survey data are less complete). Foster and Marrs (2012) estimated a 43% decrease in the index of breeding numbers of common terns in Scotland from 1986 to 2011. The future trends in breeding numbers are difficult to predict, but the common tern is vulnerable to impacts of predators at colonies and the decline in Scotland in recent years probably reflects at least in part the lack of sandeels in Shetland and other northern areas in recent years.

Common tern colonies can be displaced by large gulls moving in to breed. There is clear evidence that such displacement reduces tern numbers locally, but terns may be able to re-locate to areas without nesting gulls, and at least in some cases the regional population may be unaffected even when large tern colonies have been displaced by gulls (Jennings et al. 2012). But in some places, alternative nesting habitat for terns may not be available; management options may therefore include either provision of artificial nesting sites for terns, or removal of gulls from existing tern colony areas.

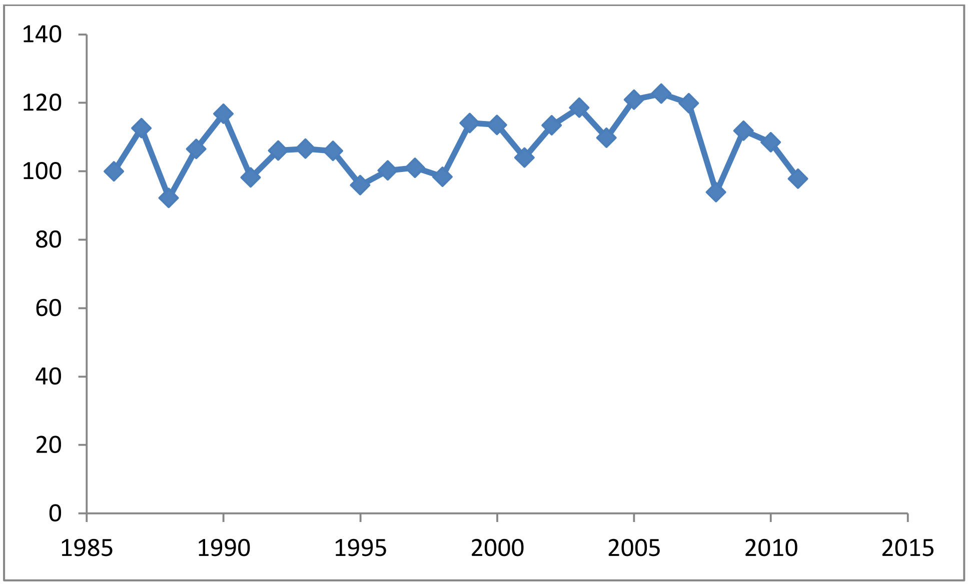

Figure 17.1.1 Common tern breeding population index for the UK. Data from JNCC online database.

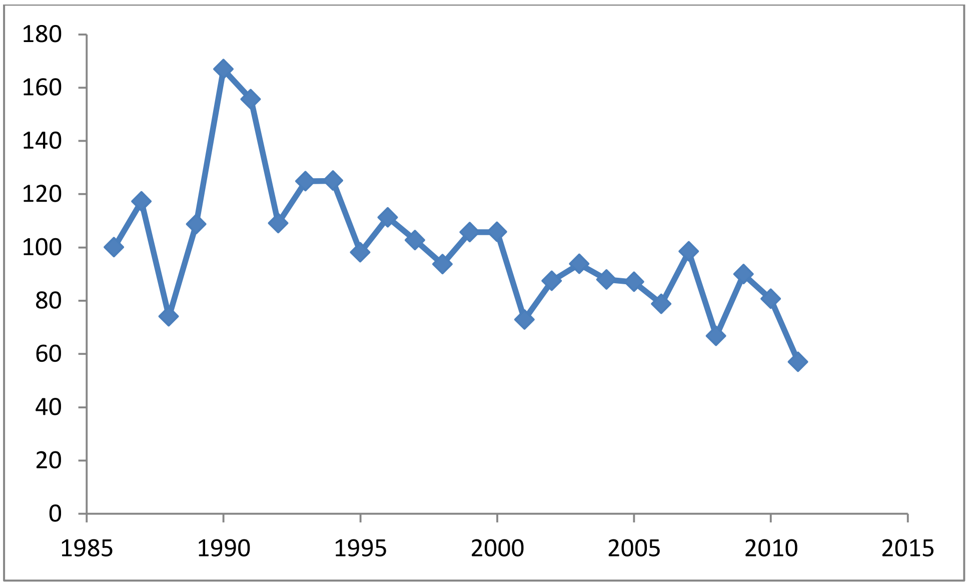

Figure 17.1.2. Common tern breeding population index for Scotland. Data from JNCC online database.

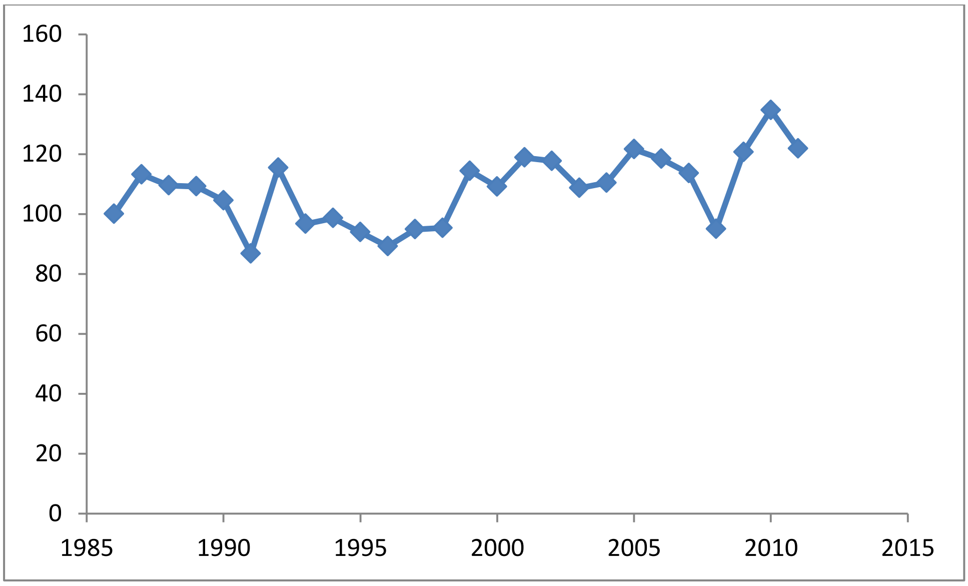

Figure 17.1.3. Common tern breeding population index for England. Data from JNCC online database.

17.1.3 *Factors affecting survival rates*

Del Hoyo et al. (1992-2006) indicated a survival rate of 0.88 for adult common terns. Nisbet and Cam (2002) estimated a survival rate of 0.88 (s.e. 0.04) for adult common terns at a North American colony. Becker and Ludwigs (2004), working at a colony in Germany, estimated adult survival at 0.9 and survival from fledging to age 2 at 0.47. Ezard et al. (2006) presented an estimate of adult common tern survival of around 0.9 for this same common tern colony, but also pointed out that population trend in numbers related more to the influence of environmental variation, especially small pelagic fish abundance, on demography than to the distance from environmental carrying capacity. However, none of these analyses quantified effects of environmental factors on survival rates. Craik (1995, 1997 and 2007) showed that invasive alien predators such as mink can strongly affect adult common tern survival as well as colony productivity, so predation impacts are likely to alter survival rates, as may pelagic fish abundance as suggested by Ezard et al. (2006). However, lacking quantitative assessments of the impacts of these factors, it is only possible to make qualitative or semi-quantitative estimates of consequences of management on survival rates of common terns.

17.1.4 *Breeding success in the British Isles*

In 1986-1989, predation by mink in the west of Scotland and by gulls and skuas in Shetland (combined with a shortage of sandeels around Shetland) were identified as factors reducing common tern productivity in those areas; by far the lowest productivity occurred in Shetland (0.035 chicks per pair in 1986-1989 compared with levels between 0.3 and 1 chick per pair elsewhere) (Walsh et al. 1990). In 1990, Walsh et al. (1991) concluded '*Reduced foraging success for [common] terns and direct losses of chicks through exposure may have contributed to poor success rates*' but indicated that predation was the most obvious cause of reduced productivity in this species. At least 12 out of 29 Scottish colonies monitored suffered total breeding failure, with mammalian predators (mink, otters, foxes) responsible in most cases. Poor weather and predation by gulls were also considered to be contributing factors. In 1991, productivity of common terns was low (averaging 0.4 chicks per pair in Scotland and 0.5 chicks per pair in England). In parts of Scotland food shortage was an issue, with abnormally small clutch sizes and adults deserting colonies. Predation affected many colonies: foxes, gulls, otters and mink were identified as predators at many sites in Scotland. In England, flooding, predation and human disturbance were listed as the main problems at most colonies. However, it was noted that success was highest at sites where artificial nesting rafts were provided for common terns (Walsh et al. 1992). In 1992, food shortage was reported at several colonies, including Lady's Island Lake (0.2 chicks per pair) and Ribble Marshes (almost total failure of >500 pairs). Predation was also reported: mink affected productivity at many colonies in west Scotland, and fox predation affected productivity at Hodbarrow (Walsh et al. 1993). Walsh et al. (1993) also noted that productivity was much higher at artificial sites than at natural colonies (on average about 30% higher) because predation and disturbance levels were much lower at artificial sites. In 1993, food shortages were quoted as a factor in reduced productivity at some colonies. Predation reduced productivity at several colonies: mink were the main problem in west Scotland, rats were responsible for taking many chicks at Shotton (Wales), although productivity there was still high at 1.22, but lower than the 1.45 achieved there in 1992. Predation by stoats, gulls and oystercatchers continued to be a problem at several colonies in NW England. At artificial sites throughout Britain success was again relatively high, as these sites provide improved protection against disturbance and predation compared to natural sites (Walsh et al. 1994). In 1994, mink caused some whole-colony failures, but mink control at several colonies in Argyll allowed productivity of 1 to 1.6 chicks fledged per pair (Walsh et al. 1995). In 1995, the main factor reducing productivity at many colonies was predation: mink at several colonies in Argyll, feral cats and mustelids at Nigg, mink and gulls at Dungeness, kestrels at North Solent NNR and peregrines at Cemlyn (Thompson 1996). In 1996, five colonies in Argyll where mink were trapped out had productivity of 1.3 fledglings per pair whereas 14 colonies where there was no mink control had a mean productivity of only 0.1 fledglings per pair). Mink removal boosted productivity by a factor of 13. However, several colonies were also affected by birds of prey, fox and otter predation. Predators affected productivity at several other colonies: stoats and polecat ferrets were trapped at the colony at St Fergus, the Isle of May colony suffered heavily from gull predation, while low water levels due to dry weather allowed mammal predators access to several colonies in SE England. In contrast, flooding reduced productivity at Rockliffe Marsh (Cumbria). At St Fergus, productivity was reduced by combined impacts of shortage of food, poor weather and human disturbance (Thompson et al. 1997). In 1997, five colonies in Argyll where mink were trapped out produced 722-942 fledglings from 873 pairs (productivity 0.83 to 1.08) whereas six colonies where there was no mink control fledged only 13-23 young from 156 pairs (productivity 0.08 to 0.15). Mink removal boosted productivity by a factor of 5.5 to 13.5. However, several of these eleven colonies were also affected by peregrine and otter predation. At St Fergus, productivity was only 0.1, as a consequence of lack of food, bad weather and disturbance. High tides washed out the majority of nests at six colonies in East England (productivity 0.3), while predation by badgers and foxes affected Dungeness and Rye Harbour while mink affected the colony at Dungeness (Thompson et al. 1998). In 1998, mink control at several colonies in Argyll increased productivity (814 pairs in six colonies fledged 469 chicks) by a factor of 2.5 to 3.4 above levels in colonies where mink were not trapped (139 pairs at ten colonies fledged 28 to 34 chicks) (Thompson et al. 1999). Elsewhere, weather affected several colonies: high tides and waves affected 15 pairs at Newton which only fledged two chicks, high tides and strong winds on the Isles of Scilly resulted in 130 pairs fledging between 20 and 30 chicks (in colonies possibly also affected by presence of brown rats), and bad weather at Rockabill was blamed for relatively poor productivity there (though the 0.96 chicks fledged per pair was well above the average for this species) (Thompson et al. 1999). In 1999, productivity averaged 0.82 chicks per pair. Productivity at Argyll colonies where mink were controlled was 6 times higher (0.66) than at colonies without mink control (0.11). At McDermott's Yard, feral cats caused total breeding failure. The Farnes colony suffered from food shortage and produced only 0.07 chicks per pair. Fox and badger predation reduced productivity at Dungeness (0.03) and Rye Harbour (0.0). Poor weather washed away nests at Lough Swilly but no productivity estimate was provided (Mavor et al. 2000). In 2000, productivity was generally good, averaging 1.09 chicks per pair. Otters greatly reduced productivity at Glas Eileanan (0.2) and Torinturk (0.16). Flooding reduced productivity at two sites in SW Scotland. Human disturbance at Alness Point reduced productivity there to 0.18, while a combination of predation, bad weather and food shortage resulted in productivity of only 0.14 at St Fergus, and predation by great black-backed gulls reduced productivity at Brownsea to 0.51 chicks per pair (Mavor et al. 2001). In 2001, overall productivity was 1 chick per pair, but lower in Scotland

(0.58 chicks per pair), apparently to a large extent due to sandeel scarcity (Mavor et al. 2002). In addition, several colonies were affected by predation and a few by weather. In Argyll, colonies were affected by mink, otter and peregrine predation. However, in north Scotland, productivity (0.71) was close to average despite some predation and disturbance (Mavor et al. 2002). Predation by black-headed gulls reduced productivity at Loch of Strathbeg to 0.16, poor weather and increased gull predation reduced productivity on the Farnes to 0.3, predation by great black-backed gulls reduced productivity at Newton, Isle of Wight, and predation (predator not identified) caused big losses at Hodbarrow where productivity was 0.0 and at Rockliffe Marsh (0.33). High winds reduced hatching success at Seaforth (Mavor et al. 2002). In 2002, overall productivity was similar to the average for earlier years. Reductions in productivity were attributed to predation, weather, and food shortage. Predation affected Glas Eileanan (otters reduced productivity to 0.0), Loch Melfort (peregrines reduced productivity to 0.71), St Fergus (carrion crows reduced productivity to 0.07), Blakeney (gulls reduced productivity to 0.27), Brownsea (predation and flooding reduced productivity to 0.06), and colonies in north Scotland (otters). Weather reduced productivity at colonies in north Scotland, Langston Harbour (0.08), Hayling Island (0.29), Brownsea (0.06), and Shotton (1.05). Food shortage affected colonies in north Scotland (Mavor et al. 2003). In 2003, productivity was above average. In Argyll, the highest productivity was achieved by a colony nesting on a converted mink-free mussel raft, the only monitored site to achieve over 2.1 chicks per pair (Mavor et al. 2004). Food shortage reduced productivity at Nigg (0.0) and at Langstone Harbour (0.23). Predators reduced productivity at many colonies: peregrines caused breeding failure at Sgeir nan Caillich (0.0), and at Dubh Sgeir (0.0), and reduced productivity at Rockabill to 1.17 which is lower than normal at that site (Mavor et al. 2004). Foxes and cats affected Alness and McDermotts, stoats and gulls affected Sands of Forvie (0.09), gulls affected Boultham Mere (0.12), Holkham (0.0), Blakeney (0.27), and Isles of Scilly. Mink killed over 200 chicks at Shotton to reduced productivity to 0.99 at this normally productive colony (Mavor et al. 2004). The colony at Breydon Water was affected by a helicopter, but birds relaid and productivity was close to normal (Mavor et al. 2004). Weather conditions had little impact, although they contributed to reduction in productivity on the Isles of Scilly and flooded a few nests at Chichester Harbour (Mavor et al. 2004). In 2004, poor weather combined with food shortages affected productivity in most regions (Mavor et al. 2005). At ten sites in Argyll where mink were successfully removed, productivity was 1.42 chicks per pair, compared to only 0.11 at five sites with no mink control (Mavor et al. 2005). At Sands of Forvie, avian predation and food shortage resulted in only one chick fledging from 19 pairs (Mavor et al. 2005). On the Isle of May, productivity of only 0.1 chicks per pair was attributed to scarcity of the preferred prey. Storms and food shortage depressed productivity at Coquet Island, and at all monitored colonies on the north Norfolk coast. Mink affected productivity at Alton Water with only four young fledged from 35 pairs on natural sites, but 17 pairs nesting on a raft fledged at least 30 young (Mavor et al. 2005). Storms, tidal inundation and fox predation affected productivity at colonies in SE England, while colonies in SW England were affected by storms, and predation by rats (Mavor et al. 2005). Storms caused egg losses at Shotton, Wales, but birds relaid and overall productivity was good. Low productivity in NW England was attributed to food shortage (Mavor et al. 2005). In 2005, presumed food shortage depressed productivity in west and north Scotland, while predation and food shortage reduced productivity at colonies in England (Mavor et al. 2006). Depredation by otters, herring gulls and mink, and starvation of chicks caused productivity in Argyll of less than half the long-term average (Mavor et al. 2006). Tidal inundation depressed productivity at Blakeney Point, Holkham and Scolt Head colonies and at the last of these sites food shortage was also evident (Mavor et al. 2006). On the Isles of Scilly, productivity was reduced by human disturbance as well as by predation by gulls and oystercatchers (Mavor et al. 2006). At Seaforth, depredations by moorhens and bad weather affected productivity. At Big Copeland, disturbance by black-headed gulls reduced productivity (Mavor et al. 2006). In 2006, particularly high productivity figures were recorded at common tern colonies in N Scotland, SE Ireland and Wales, but predation depressed productivity at several sites across Britain (Mavor et al. 2008). In SW Scotland, otters ate chicks at Sgeir na Caillich, the largest colony in the region, and greatly reduced productivity at Eilean nan Gabhar. Controlling mink at two colonies resulted in productivity above 1 chick per pair, whereas at colonies where mink were not trapped, productivity averaged below 0.5 chicks per pair (Mavor et al. 2008). At several colonies, including the Isle of May SE Scotland, Eilean an Ruisg, SW Scotland, and Havergate Island, E England, depredations by herring gulls depressed productivity. At Langstone Harbour, productivity was low due to predators, while at the Isles of Scilly productivity was low due to tides flooding the colony. Food shortage affected productivity at Seaforth, NW England, and at Coquet Island, NE England (Mavor et al. 2008). Human recreation activities (and dogs) can cause inadvertent disturbance to tern colonies accessible to walkers, and such disturbance was identified at several of the colonies monitored in the JNCC annual monitoring of seabird numbers and breeding success (Table 17.1.1). Human disturbance can act in combination with other factors, for example pushing colonies down the beach towards high tide line and therefore increasing risk of flooding or exposure to predation.

Ratcliffe et al. (2006) showed that eradiation of mink in the Western Isles improvced hatching success of terns, but in their study the fledging success was no higher than in control colonies where mink were not removed, owing to overwhelming impacts of food shortage and poor weather on chick survival. Craik (2008) suggested that gull productivity provides a better measure of the impact of mink because gull productivity is less sensitive to weather and food supply than that of terns, a point accepted by Ratcliffe (2008).

Table 17.1.1 Meta-analysis of main factors contributing to reduced productivity of Common terns at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Food shortage	47
Mink predation	42
Gull predation	24
Extreme weather conditions	22
Tidal flooding	19
Otter predation	15
Fox predation	14
Peregrine predation and disturbance	8
Human disturbance	8
Feral cat predation	5
Rat predation	5
Stoat predation	5
Badger predation	4
Great skua predation	3
Oystercatcher predation	3
Ferret predation	2
Buzzard predation	2
Crow predation	2
Kestrel predation	1
Moorhen predation	1
Helicopter disturbance	1

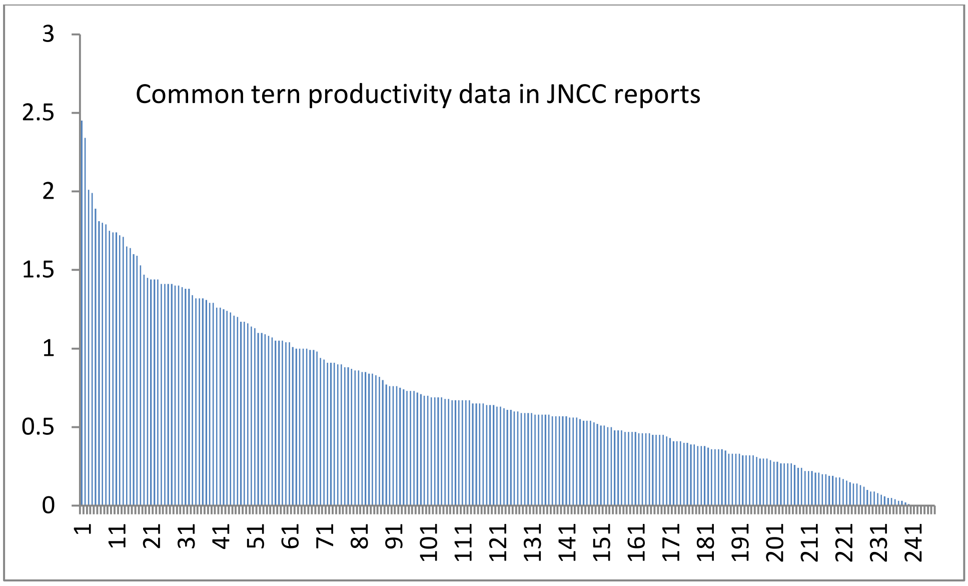

Figure 17.1.4. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of common terns at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

17.2 Management options

	17.2.1 Closure of sandeel and sprat fisheries close to colonies	17.2.2 Eradicate mink	17.2.3 Eradicate feral cats	17.2.4 Eradicate rats	17.2.5 Exclude foxes	17.2.6 Control stoats	17.2.7 Nest platforms	17.2.8 Exclude large gulls
Evidence of success for this species	Low C=Low	High C=High	Low C=Low	High C=High	High C=High	High C=High	High C=High	High C=High
Evidence of success for similar species	High C=Mod	High C=High	High C=High	High C=High	High C=High	High C=High	High C=High	High C=High
Cost-effectiveness	Moderate C=Low	High C=High	High C=High	High C=High	Mod C=High	Mod C=High	High C=High	High C=High
Feasibility	Moderate C=Low	High C=High	Mod C=High	High C=High	Mod C=High	High C=High	High C=High	Mod C=High
Practicality	Moderate C=Low	Moderate C=High	Mod C=High	Mod C=High	Mod C=High	High C=High	High C=High	Mod C=High
Applies at SPA populations	Yes C=High	No C=High	Few C=High	Few C=High	Few C=High	Few C=High	Yes C=High	Some C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

17.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs

Food shortage is implicated as a cause of reduced productivity at some colonies in some years. In the meta-analysis of JNCC data this factor came out top of the list, being identified as the cause of reduced productivity of common terns on 47 occasions. Mean foraging range of common terns is around 9 km and maximum foraging range is up to 37 km (Langston 2010), so closure of sandeel and sprat fishing within 30 km of SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

An area off the east coast of Scotland was closed to sandeel fishing from 2000 because kittiwake breeding success in the area had fallen to very low levels after intensive fishing for sandeels for several years by the Danish industrial fleet on the sand banks close to the east coast of Scotland. The aim of this fishery closure was to allow sandeel stocks in the area to recover (which sampling of sandeels in the area confirmed to be the consequence; Greenstreet et al. 2006), and kittiwake productivity was used as the proxy for sandeel abundance in the absence of data on sandeel stock biomass (Camphuysen 2005). Frederiksen and Wanless (2006) reviewed the evidence that this closure increased productivity of kittiwakes and other seabirds (but not including common tern). They found clear evidence that the closure resulted in increased productivity of kittiwakes within the study area compared with a control area outside the closure zone, results which agreed with the earlier findings of Frederiksen et al. 2004) that the sandeel fishery in this area reduced productivity of kittiwakes on the Isle of May during the years of the fishery. Productivity of kittiwakes did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (the difference was 0.28 chicks per nest and statistically highly significant ($p<0.0001$). Analysis of productivity data for monitored colonies of other seabirds was based on much smaller sample sizes, and showed less clear results due to chance variations. However, for Sandwich tern the pattern was similar, within the study zone there was higher productivity in no fishery years (0.62 chicks per pair) than in fishery years (0.38 chicks per pair) whereas productivity in the control area was consistently high. However, high variation and small sample size (there were only two colonies within the closure zone) resulted in this difference not being statistically significant. Data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie was more or less completely deserted from 1992-1998 and that this species did not breed on the Isle of May until 1999. Combined with the observed (non-significant) reduction in productivity in the closure zone during the fishery years, Frederiksen and Wanless (2006) concluded that '*this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes*'. For Arctic tern, European shag, common guillemot, razorbill and Atlantic puffin, the closure could not be shown to have any strong effect on productivity by analysis of data for each individual species in turn, but a one-way MANOVA with data for all species showed a significant effect of the fishery closure on productivity of the seabird community, but this was predominantly driven by the strong and clear effect on kittiwakes.

In Shetland, sandeel stock biomass fell rapidly in the late 1980s during the period of a local industrial fishery for sandeels, and that fishery was closed in 1990. Breeding success of kittiwakes in Shetland in 1990 was 0.12 chicks per pair. In 1991, the first year of the sandeel fishery closure, sandeel recruitment was higher than it had been in any of the previous seven years during heavy fishing, and kittiwake productivity in 1991 at 0.57 chicks per pair was the highest at monitored Shetland colonies since 1986 (JNCC data and ICES sandeel stock data). However, it is unclear whether the high recruitment in 1991 was caused by the fishery closure or whether the timing was coincidental.

Closure of all sandeel and sprat fisheries in UK waters would bring the UK into the same management position as exists in the USA, where fishing on the small pelagic fish such as sandeels that are keystone species for marine food webs (including large predatory fish of high commercial importance) is prohibited. The body of evidence strongly suggests that Sandwich tern productivity and adult survival would benefit from higher average abundances of these small prey fish, but there is no suitable evidence base to test this for common tern, so the assumed benefit is based on the similar ecology of these two species. Closure of sandeel fisheres in UK waters would have little direct cost for British fishermen, since the fishery is almost entirely carried out by the Danish fishing fleet. Closure of sprat fisheries in UK waters would affect small numbers of fishermen in NW Scotland (there is a small fishery for sprats carried out by a few local fishing vessels near to Mull in Argyll and there is a small fishery for sprats carried out by a few local fishing vessels in the English Channel (ICES 2013). Closure of these fisheries would be likely to have some beneficial effects for commercially important fish stocks such as cod and whiting (Greenstreet et al. 2006).

Closure of sandeel and sprat fisheries close to common tern colonies (or to common tern SPAs) would close a smaller total area, but since there are 22 sites across the UK designated as SPAs for common terns, closures within 30 km of all of these 22 sites would be a complex management arrangement and it might be simpler to close all UK waters to these fisheries, with consequent benefits for other wildlife and predatory fish.

17.2.2 Eradication of American mink

Mink are a factor reducing productivity at several colonies, and their depredations reduce adult survival although that impact has not been quantified (Ratcliffe et al. 2008). Eradication of invasive alien mink would allow common tern productivity to increase at colonies where this predator is present.

Eradication of mink from islands can be achieved relatively easily by trapping. However, mink swim well, and are likely to re-colonise islands that are within 2 km of habitat with mink present (and most common tern colonies are on islands very close to the mainland or are on the mainland). Eradication of mink from mainland areas is more difficult, although there is evidence from mink eradication projects in the highlands of Scotland and in Iceland that mink can be eradicated from river catchments and in such cases re-colonisation by mink from adjacent river catchments may be inhibited by upland areas in between. The Hebridean Mink Project run by Scottish Natural Heritage provides a very relevant example of mink eradication benefitting terns (Ratcliffe et al. 2006).

Where eradication of mink is difficult to achieve due to re-colonisation from adjacent areas, Craik (2007) showed that trapping mink in the immediate vicinity of seabird colonies in late winter and early spring normally leads to a ‘mink-free’ status for the seabird colony through the summer, resulting in considerably improved productivity of gulls (from around 0.2 chicks per pair up to around 0.8 chicks per pair). However, trapping mink every year represents a long-term commitment that will be more expensive in the long term than eradication if that can be achieved. An alternative to annual control of mink may be mink-proof fencing to protect colonies, such that the need for annual trapping is removed once the area has been fenced and mink within the fenced area have been removed. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012) (see 17.2.5 below).

17.2.3 Eradication of feral cats

Feral cats are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of feral cats would allow common tern productivity to increase at colonies where this predator is present.

Eradication of feral cats can be more complex than eradication of rats, especially where there is a resident human population with pet cats, as experienced on Ascension Island (Ratcliffe et al. 2009). In that example, large numbers of pet cats were killed by the poison deployment (38% of 168 domestic cats were accidentally killed) despite efforts to keep domestic pet cats safe (which had all been neutered as part of the project). Nevertheless, that eradication successfully removed all of the feral cats and has resulted in re-establishment of seabirds that had been extirpated from the main island (Ratcliffe et al. 2009). This project cost £650,000 over three years, with 24,200 person-hrs of cat control work and 14,480 person-hrs of monitoring work on an island of 97 km^2 (9,700 ha so a cost of £67 per ha which is not far from the costs per unit area of rat eradication programmes on UK islands) but where fieldwork is made relatively easy by the climate, terrain and existence of roads (Ratcliffe et al. 2009). Cats have been successfully eradicated from many islands, the largest being

Marion Island (290 km^2) in the sub-Antarctic Indian Ocean, where cats were eradicated in 2000, but this required a 15-year programme including shooting and poisoning and hunting with dogs and introduction of disease (feline panleucopaenia virus) to eradicate a population of around 3,400 cats at the start of the programme that were estimated to be killing over 450,000 seabirds per year (van Aarde 1980, Bloomer and Bester 1992, Bester et al. 2000). The second-largest island from which cats have been eradicated is Macquarie Island (120 km^2) in the sub-Antarctic which had a population of around 2,400 cats, and where the programme of trapping, hunting and deployment of the poison 1080, took 25 years before success was achieved around 2000 (Copson and Whinan 2001). Obviously these very long programmes are much more expensive, and the long timescale required can be attributed to relatively limited expertise in this type of work when those two projects started, and major logistical difficulties of working on uninhabited sub-Antarctic islands far from the mainland. Nogales et al. (2004) provide a useful tabulation of data about 48 islands where eradication of cats has been carried out and summary information on these eradication programmes, indicating that most programmes are now completed within 1-3 years, although none of the projects reviewed in that paper were in the British Isles. Although trapping, shooting, hunting with dogs, poisoning with baits and introducing disease have been the main methods used to eradicate cats, virus-vectored immune-contraception may hold promise in the near future (Courchamp and Cornell 2000). An alternative to eradicating feral cats may be cat-proof fencing to protect colonies. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012) (see 16.2.5 below).

17.2.4 Eradication of rats

Brown rats are a factor reducing productivity at a few colonies. Eradication of invasive alien rats would allow common tern productivity to increase at colonies where this predator is present.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a professional New Zealand company with considerable experience, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

17.2.5 Exclusion of foxes from colonies

Foxes are a factor reducing productivity at several colonies, particularly in England, and their depredations appear to reduce adult survival although that impact has not been quantified. Exclosure fencing to keep foxes out of colonies would allow common tern productivity to increase at colonies where this predator is present.

Foxes can be controlled on small islands by shooting, although recolonisation from the mainland may be an issue for islands situated within 1 or 2 km of the mainland.

Fencing of colonies to exclude foxes would allow common tern productivity to increase at colonies where this predator is present. In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require some maintenance. A more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka' ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Such completely predator-proof fencing may be unnecessary to protect colonies just from foxes, but might be especially appropriate for colonies subject to predation by rats or mink as well as by foxes.

By 2006, in total, around 109 km of predator-proof fencing had been erected in various areas of mainland New Zealand to exclude predators from sites with important populations of native animals and birds (Scofield et al. 2011, Innes et al. 2012, Scofield and Cullen 2012). This predator-proof fencing is expensive, costing around £100 per m to construct, and around £1 per m per year to maintain, with a life-span in New Zealand of around 25 years, so a considerable rate of depreciation (Scofield et al. 2011). However, maintenance costs and life span will depend very much on the environment where the fencing is set up. In New Zealand, where much of the fencing is in forested habitat, trees falling onto the fence can cause expensive damage, as can cyclones (Scofield et al. 2011). In the predominantly open habitat of UK seabird colonies such fencing would be under less risk of damage, although corrosion from salt spray would be a consideration. There are several companies in New Zealand providing predator-proof fencing, for example see:
http://www.xcluder.co.nz/
http://www.meshindustries.com/Home
http://www.pestprooffences.co.nz/

There are several examples of the use of predator-proof fences to protect seabirds from mammals. A predator-proof fence completed in 2007 stretches 10.6 km across the neck of the peninsula from coast to coast at Cape Kidnappers Peninsula, North Island, New Zealand. This fence protects a privately owned and financed seabird restoration project where grey-faced petrels and Cook's petrels are being re-introduced (Cooper 2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels).

A good example of successful deployment of a predator-proof fence to protect a seabird colony is one erected in 2001 to protect 36-ha on Pitt Island (Chatham Islands, New Zealand) from feral cats and pigs. Between 2002 and 2005, 200 endangered Chatham petrel chicks from the only known breeding site on South East Island (Chatham Islands) were moved into the fenced reserve. In 2012, 17 pairs from these translocated birds returned to breed, producing 15 chicks http://www.chathams.co.nz/index.php/naturalheritage/116-chatham-petrel-recovery

Cooper (2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels lists a further ten examples of deployment of predator-proof fencing around seabird colonies in New Zealand, Hawaii (USA) and Azores (Portugal).

17.2.6 Control of stoats close to colonies

Stoats are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Stoat removal or exclusion would allow common tern productivity to increase at colonies where this predator is currently present and reducing productivity. It may be necessary to deploy trapping to reduce stoat numbers at colonies or to eradicate them from islands. However, a more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka' ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Deploying such predator-proof fencing around colonies affected by stoats would allow common tern productivity to increase at colonies where this predator is currently present and reducing productivity.

17.2.7 Construction and deployment of predator-proof nesting rafts

Evidence indicates that common terns readily take to nesting on purpose-built rafts, and that productivity on these artificial colonies is considerably higher than at natural colonies. Provision of predator-proof nesting rafts for common terns appears to be the most effective way to increase productivity in this species in the UK, and is almost certainly the most cost-effective approach. Such rafts are not inaccessible to avian predators of course, but evidence from their deployment is that avian predators rarely have a significant impact on productivity of colonies on rafts (Clive Craik pers. comm.; see also text below reviewing achieved productivity on rafts).

The outstanding case study of provision of nesting rafts for common terns is work being done by Dr Clive Craik in Argyll. Initially, he developed a derelict mussel farming raft on which a few pairs of common terns had started nesting. This raft was a wooden rectangular structure moored in a sea loch in Argyll which had been used for many years to grown mussels on ropes hanging from the raft. Clive enhanced the raft for terns, adding gravel substrate on which they could nest, and fences around the perimeter to deter mammal predators. Numbers of common terns nesting on the raft increased over the years, and with protection from mammal predators this colony has been highly productive. In 2003, this raft produced 2.1 chicks per pair, making it the most productive common tern colony out of the dozens monitored throughout the UK (Mavor et al. 2004), and this pattern has been replicated in most years since then (Clive Craik pers. comm.). In 2010 a decision was made to build more rafts at this site to provide a larger surface area for nesting terns. Construction of new rafts (developed to be predator-proof), and their licenced Crown Estate moorings, cost about £20,000. Half of this was provided by Argyll Bird Club, and half by Clive Craik himself. In 2011, the year these new rafts were constructed, over 300 pairs of common terns nested, and fledged over 300 chicks. In 2012, 600 pairs of common terns nested on these rafts making this the 2nd largest common tern colony in Scotland in 2012, and fledged more common tern chicks from this one site (over 600) than were produced by all of the natural common tern colonies in the whole of the west of Scotland. Common terns appear to be very quick to adopt artificial nesting platforms and experience high success rates on them. Similar success, though on a smaller scale, has been achieved by rafts placed for common terns at other locations. Herts and Middlesex Wildlife Trust placed rafts on Wilstone Reservoir in the 1990s, a site where common terns had not previously nested but occurred on migration in small numbers. Small numbers of common terns colonised these rafts, breeding successfully in most years. In 2008, twelve chicks fledged from the rafts http://www.tringreservoirs.org.uk/cmntern.html. Lincolnshire Wildlife Trust constructed a fibreglass raft to deploy on Grebe Lake, Whisby Nature Park, near Lincoln, to replace an older raft that was decomposing http://lincstrust.org.uk/conservation/article.php?id=21 and offer these rafts on a commercial basis. Nesting rafts for common terns are also promoted by the RSPB http://www.rspb.org.uk/ourwork/conservation/advice/rafts/.

A small, replicated, controlled study from May-August in 1982 on a concrete breakwater in Port Colborne, Canada (Richards and Morris 1984), found that common terns nested at higher densities on two plots enhanced with clumps of mossy stonecrop and driftwood (62% of 166 clutches in these plots), compared to plots layered with gravel (29% of clutches) or control plots of bare concrete (9% of clutches). Enhanced plots were also colonised earlier. Average clutch size and hatching rates were similar between plots (2.4-2.5 eggs/clutch and 76-86% hatching success), but the average number of chicks fledged per pair was significantly higher in enhanced (1.6) and control (1.3) plots than in gravel-layered plots (0.6).

A 1992 review of the use of artificial islands and floating platforms in 17 wetland nature reserves across the UK (Burgess and Hirons 1992) found that all seven species of gull and tern investigated used sparsely-vegetated islands and platforms at southern, coastal sites, but that nesting sites elsewhere were not used by four of the species. Sandwich terns used vegetated nesting sites at southern coastal sites, whilst common terns nested at all sites. At one site in Kent, the provision of 20 shingle islands attracted 350 pairs of Sandwich and common terns (Burgess and Hirons 1992).

A replicated study in 1987-1990 of a managed wetland in Macedonia, Greece (Pyrovetsi 1997) found that the target species, Dalmatian pelicans, did not benefit consistently from artificial habitats although other waterbirds did. Two constructed rafts and one artificial island were used extensively by a variety of waterbirds as resting and foraging sites. Common terns colonised the rafts in both years (average 12 nests and 14 fledglings per raft).

Dunlop et al. (1991) reported that rafts they built in spring 1990 in Totonto Outer Harbour were used by about 130 pairs of common terns, which achieved productivity of 1.3 chicks per pair, whereas this colony had been failing and declining in previous years due to predation, competition with gulls and human disturbance.

Although not based on deployment of rafts, a before-and-after study on Praia Islet (12 ha), off Graciosa in the Azores (Bried et al. 2009), found that the breeding population of common terns increased dramatically (from no pairs, to over 1,000 pairs) following the installation of nest boxes in 1996, combined with the eradication of rabbits and habitat restoration. It is unclear how much of this increase can be attributed to eradication of rabbits or to habitat restoration (which mainly involved removal of alien vegetation), or to provision of nestboxes. Although terns occupied many of the nestboxes, the breeding numbers increased far more than the numbers of nestboxes that were provided, suggesting that creation of more open ground without invasive vegetation may have been mostly responsible. However, provision of nestboxes on rafts may further aid increases in common tern numbers by providing shelter from the weather and some protection from avian predators.

17.2.8 Exclude large gulls from nesting close to colonies

Predation by large gulls affects productivity at some colonies in some years (this was identified by the meta-analysis of JNCC data to be the third most frequently identified cause of reduced productivity, reported 24 times). Removal of gull territories adjacent to these key colonies could improve productivity of common terns. It is likely that most of the common tern colonies affected by gull predation are affected by small numbers of gulls that have developed specialised habits of feeding on tern eggs and chicks, and so the numbers of gulls that would need to be removed to increase tern productivity are likely to be small. However, this measure might not be necessary if other measures can be put in place (such as providing nesting platforms which although they would not be immune to impacts from gulls would be likely to increase productivity sufficiently to make removal of gulls unnecessary.

18. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR COMMON GUILLEMOT

18.1 Common guillemot ecology

A widespread and numerous species, with a high-medium latitude breeding distribution around the northern hemisphere. The world population is around 7.3 to 7.4 million pairs, with around 2.8 to 2.9 million of these in the North Atlantic (Mitchell et al. 2004).

18.1.1 The species in the British Isles

Seabird 2000 found 890,000 pairs in Great Britain the Isle of Man and Channel Islands, and 160,000 pairs in Ireland (Mitchell et al. 2004). Common guillemots breed in dense colonies on cliff ledges, on sea stacks, or under boulders below cliffs. They lay a single egg onto bare rock, but will replace lost eggs around 14 days later. While breeding, common guillemots forage by diving to catch fish by underwater pursuit, feeding chicks especially on sandeels and sprats.

The UK SPA suite for this species holds around 693,000 pairs (95% of the UK total) across 34 sites: Ailsa Craig, Buchan Ness to Collieston Coast, Calf of Eday, Canna and Sanday, Cape Wrath, Copinsay, East Caithness Cliffs, Fair Isle, Farne Islands, Firth of Forth Islands, Flamborough Head and Bempton Cliffs, Flannan Isles, Foula, Fowlsheugh, Handa, Hermaness Saxa Vord and Valla Field, Hoy, Marwick Head, Mingulay and Berneray, North Caithness Cliffs, North Colonsay and Western Cliffs, North Rona and Sula Sgeir, Noss, Rathlin Island, Rousay, Rum, Shiant Isles, Skomer and Skokholm, St Abb's Head to Fast Castle, St Kilda, Sule Skerry and Sule Stack, Sumburgh Head, Troup Pennan and Lion's Heads, West Westray.

Adult survival rate averages 0.885 (del Hoyo et al. 1996) and mostly breed for the first time when 5 years old (Cramp and Simmons 1977-1994). British common guillemots disperse away from colonies in late summer, but mostly overwinter in British waters. Small numbers may overwinter further afield, from southern Norway to Iberia. Few birds from overseas populations migrate through, or overwinter in, British waters, although small numbers from Scandinavian, Faroese and Icelandic colonies reach northern Britain in winter (Wernham et al. 2002).

18.1.2 Present, and likely future trends

Seabird 2000 recorded about 1,600,000 individual common guillemots at colonies in Britain and Ireland, with 75% in Scotland, 15% in Ireland, 6% in England the Isle of Man and Channel Islands, and 4% in Wales (Mitchell et al. 2004). This was 32% higher than found by the SCR Census in 1985-88, which itself was 81% higher than the number recorded in Operation Seafarer in 1969-70 (Mitchell et al. 2004). The JNCC index of breeding numbers of common guillemots at UK colonies (Figure 18.1.1) indicates a further slight increase in numbers from 2000 to 2011, but the trend differs dramatically between northern and southern colonies. In Scotland, the index has declined (Figure 18.1.2), whereas in Wales it has continued to increase strongly (Figure 18.1.3). Foster and Marrs (2012) estimated a 24% decrease in the index of breeding numbers of common guillemots in Scotland from 1986 to 2011, but Figure 18.1.2 shows that the decrease has been even stronger than this if measured from the peak numbers recorded in 2001. The index shows a 40% drop in numbers from 2001 to 2011 at monitored Scottish colonies. This decrease within Scotland also varies regionally, being much greater at Shetland colonies than in SE or SW Scotland. It is difficult to predict future changes in common guillemot numbers, but the pronounced current trends of decreases in the north and increases in the south show no evidence at present of changing.

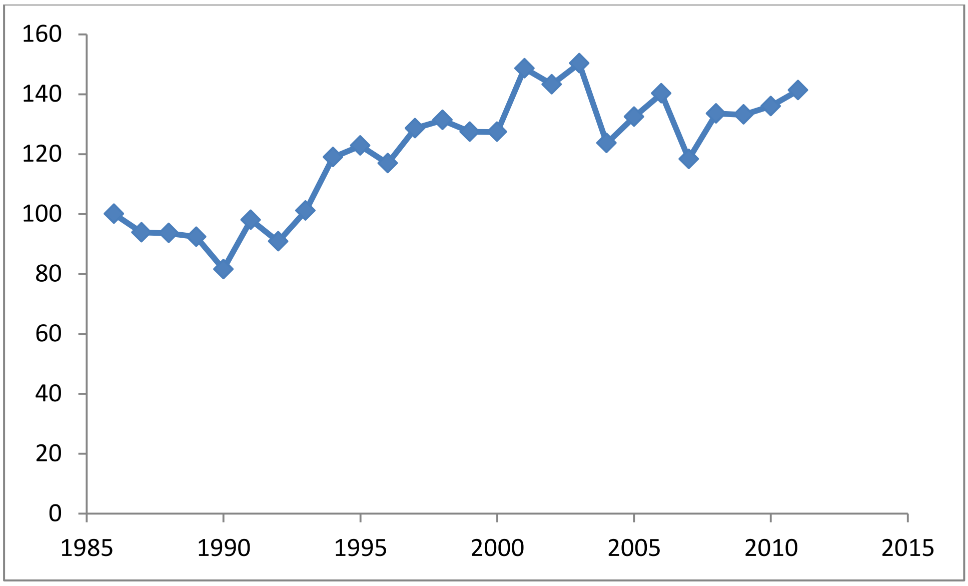

Figure 18.1.1. Common guillemot breeding population index for the UK. Data from JNCC online database.

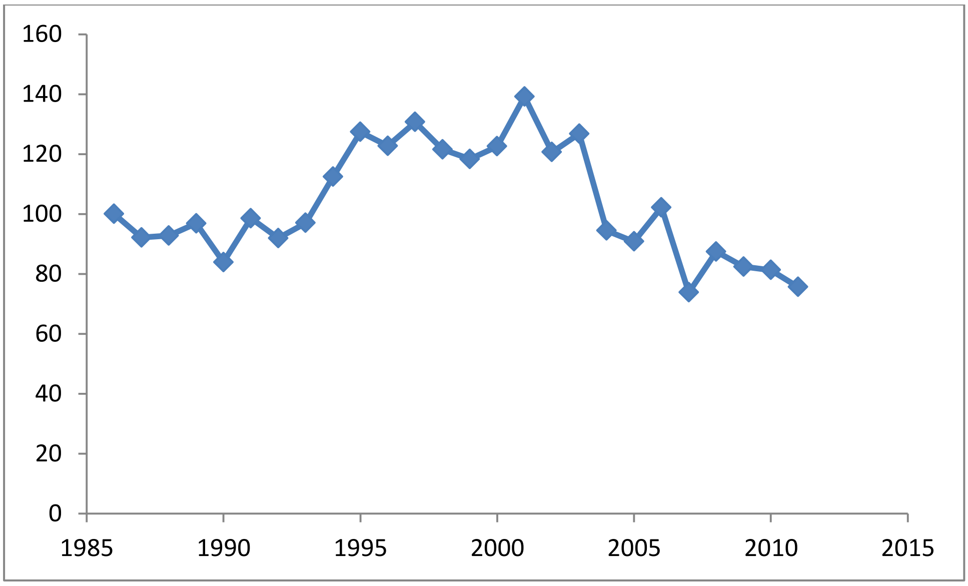

Figure 18.1.2. Common guillemot breeding population index for Scotland. Data from JNCC online database.

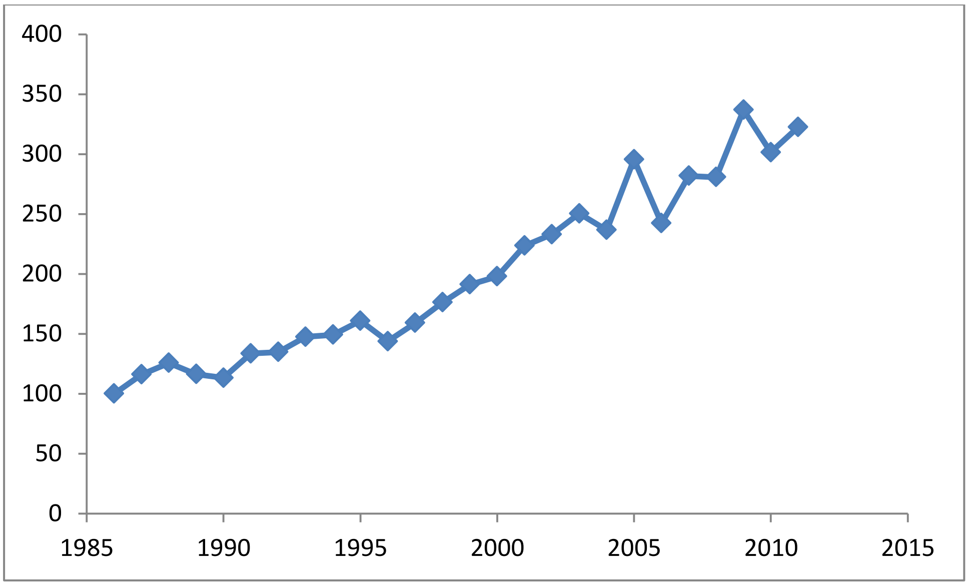

Figure 18.1.3. Common guillemot breeding population index for Wales. Data from JNCC online database.

18.1.3 Factors affecting survival rates

There have been numerous studies of adult survival rates of common guillemots, in the British Isles (Southern et al. 1975 0.87 NE Scotland, Birkhead 1974 British ringing recoveries 0.879, Mead 1974 British ringing recoveries 0.937, Birkhead and Hudson 1977 Skomer 0.904, Harris and Bailey 1992 Isle of May 0.94, Harris et al. 1992, Harris and Wanless 1995, 1996, Wernham et al. 1997, Poole et al. 1998, Harris et al. 2000 0.952 Isle of May 0.964 Colonsay 0.939 Canna, 2007, Votier et al. 2005, Reynolds et al. 2008, Votier et al. 2008), elsewhere in Europe (Olsson et al. 2000 Sweden 0.859, Sandvik et al. 2005 North Norway 0.961) and in North America (Sydeman 1993 California 0.939). Adult survival rate can vary between years and colonies. Most of these studies report survival rate estimates without including environmental factors as covariates. However Sandvik et al. (2005) showed a strong negative correlation between common guillemot survival at a colony in north Norway and SST, and a suggestion of a weak relationship with herring abundance. Votier et al. (2005) analysed the influences of oil spill incidence, North Atlantic Oscillation (NAO), sea surface temperature (SST) and indices of forage fish abundance on survival rates of adult common guillemots from Skomer, Wales. They found that survival rates varied between years, from 0.975 to 0.879, with the four lowest survival rates all occurring in years with major oil spills in the region. As a result, oil spill incidence significantly affected survival rate, with mortality doubling from 4.43% in years with no oil spill to 8.96% in years with a major oil spill. The NAO also influenced survival rate. Forage fish indices had a weak influence, possibly because the data on forage fish in the area are poor (there are no fisheries for sprats or sandeels in the region), and SST had a small influence but much less than the NAO. Harris and Bailey (1992) reported a strong correlation (0.8) between sprat abundance in the North Sea and the survival of first year guillemots from the Isle of May, but no detectable relationship for adult common guillemot survival rate across the same period (which varied only from 0.97 to 0.93), suggesting that inexperienced guillemots may be much more susceptible to reduced food abundance than are experienced adults.

18.1.4 Breeding success in the British Isles

In 1991, mean productivity at 11 monitored colonies was 0.73 chicks per pair, with colonies showing lower productivity mainly ones affected by predation (Herring gulls took guillemot eggs at Sumburgh Head and productivity there averaged 0.62) (Walsh et al. 1992). In 1992, productivity averaged 0.77 chicks per pair, with little geographical variation (Walsh et al. 1992). In 1993, productivity averaged 0.72 chicks per pair. Increased adult attendance at colonies suggested improved food supply compared to 1988-90 (Walsh et al. 1994). In 1994, overall productivity was high, averaging 0.76

chicks per pair, but poor weather was thought to have been the reason for a reduction in success at some Orkney colonies (e.g. from 0.77 chicks per pair at some Orkney colonies to 0.65 at Mull Head) (Walsh et al. 1995). In 1997, productivity was reduced at many colonies as a consequence of storms; mean productivity was 0.62 chicks per pair, suggesting that the exceptional weather had reduced productivity by about 10% from 'normal' levels (Thompson et al. 1998). In 1998, productivity averaged 0.73 chicks per pair, but there was evidence of food shortage at some colonies. At the Isle of May, chicks fledged 25% lighter than normal and productivity there was below the mean for the previous 17 seasons (Thompson et al. 1999). In 1999, breeding success at the Isle of May was the lowest recorded (0.66 chicks per pair) with evidence of food shortage (lowest recorded weights of chicks) (Upton et al. 2000). At North Sutor, predation by Great black-backed gulls contributed to low productivity (Upton et al. 2000). In Shetland in 1999, thousands of eggs were washed into the sea on west-facing coasts in a gale in May, and although a proportion of birds relaid, survival of late-hatched chicks was low due to predation by Great black-backed gulls (Upton et al. 2000). In 2000, productivity across 10 colonies averaged 0.74 chicks per pair, marginally above the average of 0.73 for 1986-1999 (Mavor et al. 2001). However, at Sumburgh some 8,000-9,000 chicks were washed into the sea by a severe storm on 13 June 2000 (Mavor et al. 2001). It was a relatively poor season in 2001, with 8 out of 10 monitored colonies recording below average productivity, this low productivity being attributed to adults experiencing difficulties finding food (Mavor et al. 2002). That supposition was supported by the lowest chick provisioning rates yet recorded on Fair Isle, and a high level of chick neglect on the Isle of May and at Sumburgh (Mavor et al. 2002). In 2002, productivity averaged 0.75 chicks per pair; lowest productivity was in Shetland (0.64) where attendance of chicks by adults was noted to be very low, suggesting food shortage. Further evidence for food shortage at Shetland in 2002 was lower chick weights that year compared to a sample taken in 1999 (Mavor et al. 2003). In Shetland, breeding success in 2003 was lower than in any previous year, with low growth rates of chicks and low attendance by adults indicating food shortage. Low attendance of adults also allowed increased rates of predation of eggs and chicks by large gulls (Mavor et al. 2004). On Canna in 2003, eggs depredated by brown rats were found and it appeared that rats had been responsible for redistribution of breeding common guillemots into areas inaccessible to rats (Mavor et al. 2004). The breeding success of common guillemots was lower in 2004 than in any previous year of the monitoring programme started in 1986. Many chicks were left unattended suggesting that adults had difficulty finding food (Mavor et al. 2005). Mavor et al. (2005) specifically stated '*Low availability of high quality food (e.g. sandeels) was likely the cause of the much reduced breeding success at all the aforementioned colonies, whether directly – due to chick starvation – or indirectly – due to hypothermia, predation, or killing of unattended young by neighbouring adults' 'Wing/weight ratios of chicks were nearly identical to that recorded in 2003, also a year of low success, but chicks of all wing lengths were, on average, 50g lighter than in 1999, a year of relatively high success*'. Productivity was only marginally better in 2005, but still very poor and apparently reflecting low availability of food fish. Chicks on Fair Isle and at Compass Head were around 50g lighter than normal for their ages, strongly implicating food shortage (Mavor et al. 2006). Although food was considered to be the main factor affecting productivity, predation of eggs and chicks by gulls was observed at North Sutor in 2005, especially in smaller sub-colonies (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for common guillemots as follows. Productivity was generally low in 2006 in all regions of Britain and Ireland. Mean success was 0.48 chicks per pair, well below the long-term average of 0.69 chicks per pair for 1986-2005. Many losses occurred at the egg stage, but in addition, chick weights were lighter than normal, suggesting that food shortage was the main factor reducing productivity. On the Isle of May, chick survival was estimated at 49%, whereas 90% had previously been typical. Many chicks were left unattended, indicating that adults were finding it hard to find food. Chick losses were due mainly to starvation or attacks from neighbouring adults, with gull predation a minor factor. In contrast, on Skomer, predation by great black-backed gulls was thought to be the main cause of chick loss in 2006 (Mavor et al. 2008). This was also the case in 2004 at this colony, where productivity was relatively high (0.66 chicks per pair) and not apparently affected by food shortage (Mavor et al. 2005).

Table 18.1.1 Meta-analysis of main factors contributing to reduced productivity of Common guillemots at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	**Cases reported**
Food shortage	22
Gull predation	7
Extreme weather conditions	5
Rat predation	1

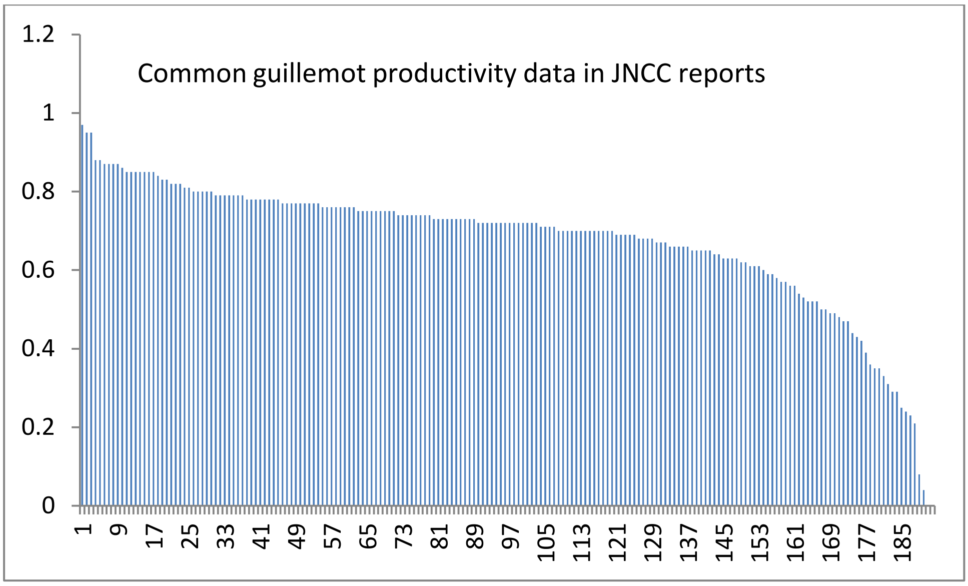

Figure 18.1.4. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of common guillemots at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

18.2 Management options

	18.2.1 Closure of sandeel and sprat fisheries in all UK waters	18.2.2 Closure of sandeel and sprat fisheries in wintering areas	18.2.3 Eradicate rats	18.2.4 Prevent oil spills
Evidence of success for this species	Low C=Low	Low C=Low	Low C=Mod	High C=Mod
Evidence of success for similar species	High C=Mod	Low C=Low	High C=High	Low C=Low
Cost-effectiveness	Uncertain C=Low	Uncertain C=Low	High C=Low	Uncertain C=Low
Feasibility	Moderate C=Low	Moderate C=Low	High C=High	Low C=High
Practicality	Moderate C=low	Moderate C=Low	Moderate C=High	Low C=High
Applies at SPA populations	Yes C=High	Uncertain C=High	Few C=High	Yes C=High

*Confidence in each score is listed as ‘High’ ‘Moderate’ or ‘Low’ based on amount and consistency of evidence

18.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs or in all UK waters

Food shortage is implicated as a cause of reduced productivity at some colonies in some years. Indeed, food shortage was identified in the meta-analysis of JNCC monitoring as the most important factor reducing common guillemot productivity at colonies in Britain and Ireland (22 cases out of 35). Mean foraging range of common guillemots is around 38 km and maximum foraging range is up to 200 km (Langston 2010, Thaxter et al. 2012), so closure of sandeel and sprat fishing within 100 km of SPAs should increase productivity and adult survival, but that would amount to closure of sandeel and sprat fishing in British waters given the wide distribution and large number of SPAs for these species. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

An area off the east coast of Scotland was closed to sandeel fishing from 2000 because kittiwake breeding success in the area had fallen to very low levels after intensive fishing for sandeels for several years by the Danish industrial fleet on the sand banks close to the east coast of Scotland. The aim of this fishery closure was to allow sandeel stocks in the area to recover (which sampling of sandeels in the area confirmed to be the consequence; Greenstreet et al. 2006), and kittiwake productivity was used as the proxy for sandeel abundance in the absence of data on sandeel stock biomass (Camphuysen 2005). Frederiksen and Wanless (2006) reviewed the evidence that this closure increased productivity of kittiwakes and other seabirds. They found clear evidence that the closure resulted in increased productivity of kittiwakes within the study area compared with a control area outside the closure zone, results which agreed with the earlier findings of Frederiksen et al. 2004) that the sandeel fishery in this area reduced productivity of kittiwakes on the Isle of May during the years of the fishery. Productivity of kittiwakes did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (the difference was 0.28 chicks per nest and statistically highly significant ($p<0.0001$). Analysis of productivity data for monitored colonies of other seabirds was based on much smaller sample sizes, and showed less clear results due to chance variations. However, for Sandwich tern the pattern was similar, within the study zone there was higher productivity in no fishery years (0.62 chicks per pair) than in fishery years (0.38 chicks per pair) whereas productivity in the control area was consistently high. However, high variation and small sample size (there were only two colonies within the closure zone) resulted in this difference not being statistically significant. Data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie was more or less completely deserted from 1992-1998 and that this species did not breed on the Isle of May until 1999. Combined with the observed (non-significant) reduction in productivity in the closure zone during the fishery years, Frederiksen and Wanless (2006) concluded that '*this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes*'. For Arctic tern, European shag, common guillemot, razorbill and Atlantic puffin, the closure could not be shown to have any strong effect on productivity by analysis of data for each individual species in turn, but a one-way MANOVA with data for all species showed a significant effect of the fishery closure on productivity of the seabird community, but this was predominantly driven by the strong and clear effect on kittiwakes. So, despite the clear evidence from the JNCC monitoring that food shortage has a strong impact on common guillemot productivity, analysis of the data from the Isle of May and east Scottish sites from the 1990s did not show this effect. Possibly food fish abundance needs to fall to lower levels to affect common guillemots than it does to affect kittiwakes and terns (Furness and Tasker 2000).

In Shetland, sandeel stock biomass fell rapidly in the late 1980s during the period of a local industrial fishery for sandeels, and that fishery was closed in 1990. Breeding success of kittiwakes in Shetland in 1990 was 0.12 chicks per pair. In 1991, the first year of the sandeel fishery closure, sandeel recruitment was higher than it had been in any of the previous seven years during heavy fishing, and kittiwake productivity in 1991 at 0.57 chicks per pair was the highest at monitored Shetland colonies since 1986 (JNCC data and ICES sandeel stock data). However, it is unclear whether the high recruitment in 1991 was caused by the fishery closure or whether the timing was coincidental.

Closure of all sandeel and sprat fisheries in UK waters would bring the UK into the same management position as exists in the USA, where fishing on the small pelagic fish such as sandeels that are keystone species for marine food webs (including large predatory fish of high commercial importance) is prohibited. The considerable body of evidence shows that kittiwake productivity and adult survival would benefit from higher average abundances of these small prey fish, and the JNCC data together with the detailed research at the Isle of May (Frederiksen et al. 2004, Frederiksen and Wanless 2006) show that the depletion of sandeel stocks has been the single most influential factor reducing productivity of kittiwakes at colonies in the British Isles, but evidence that this would affect common guillemot productivity is less strong. Closure of sandeel fisheries in UK waters would have little direct cost for British fishermen, since the fishery is almost entirely carried out by the Danish fishing fleet. Closure of sprat fisheries in UK waters would affect small numbers of fishermen in NW Scotland (there is a small fishery for sprats carried out by a few local fishing vessels near to Mull in Argyll and there is a small fishery for sprats carried out by a few local fishing vessels in the English Channel (ICES 2013). Closure of these fisheries would be likely to have some beneficial effects for commercially important fish stocks such as cod and whiting (Greenstreet et al. 2006).

18.2.2 Closure of sandeel and sprat fishing in areas where these species are aggregated in winter

Crashes of common guillemot populations have occurred in winter as a result of food shortage, and ‘wrecks’ of common guillemots occur in autumn/winter in some years, suggesting that winter may be a critical season determining survival rates. If so, protection of prey fish stocks that are important as winter food may be of greatest importance. Common guillemots feed extensively on sprats during winter, and also take some sandeels (even though sandeels are mostly buried in the sand in winter, guillemots will dig them out of the sand; M.L. Tasker pers. comm.) and sandeels were well represented in guillemot stomachs from birds killed in the Braer oil spill in Shetland which occurred in mid-winter. British common guillemots mainly overwinter in British waters, though often south of their breeding areas, so closure of sprat and sandeel fisheries in southern Britain may be most effective in protecting their food supply. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider.

There are localised fisheries for sprats in UK waters, by Scottish fishermen off west Scotland (usually close to Mull), by English fishermen in the English Channel (Lyme Bay), and occasionally as a result of the activity of foreign industrial fishing fleets (ICES 2013). There have in the past been fisheries on sprats in the Moray Firth and in the Firth of Forth, fisheries which appear to have caused local depletion of those stocks and then been closed (Jennings et al. 2012). There has in the past been a fishery by Scottish fishermen on sandeels in Shetland which was closed in 1991 due to the depletion of that stock which has still not recovered, and the large industrial fishery for sandeels in the North Sea has moved around over the years to exploit different stocks within the North Sea, resulting in depletion of many of the distinct sandeel stocks in the northern North Sea, but continued exploitation by Danish fishermen of sandeel stocks in English waters which appear to have been more resilient than stocks further north in the North Sea (ICES 2010, ICES 2012). All of these sandeel and sprat stocks are likely to represent important preferred food of common guillemots in UK waters in winter. Closure of sandeel fishing in UK waters would have low cost for UK fishermen, as almost the entire fishery is carried out by Danish fishermen. Closure of sprat fishing in UK waters would affect a small number of fishing vessels in west Scotland and in Lyme Bay, and limit potential development of sprat fisheries on stocks that are currently unfished (e.g. in Irish Sea, Clyde, Thames, Firth of Forth, Moray Firth).

18.2.3 Eradication of rats

Brown rats are a factor reducing productivity at a few colonies. Eradication of invasive alien rats would allow common guillemot and razorbill productivity to increase at colonies where this predator is present. However, this applies at a very small proportion of colonies and it is very unlikely that rat eradication would be an effective conservation measure for common guillemots.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

18.2.4 Prevent risk of major oil spills near to SPAs

The effect of oil pollution on seabird populations is surprisingly uncertain. Votier et al. (2005) showed that major oil spill incidents doubled mortality of common guillemots from a Welsh colony in the years in which these occurred. Major incidents such as the oil spill accidents considered by Votier et al. (2005) probably kill fewer seabirds than the chronic discharges of small quantities of oil from illegal discharges (Dunnet 1982). So the high influence of these major oil spills on common guillemot survival rates probably understates the influence of oil pollution. Nevertheless, previous attempts to relate changes in numbers of common guillemots attending colonies to oil spill mortality have often failed to show any detectable impact (see for example the lack of impact of the Braer oil spill in Shetland on seabird breeding numbers or breeding success http://www.nature-shetland.co.uk/birdclub/braer/Part10.html), suggesting that the consequence of increased mortality is not seen at the level of breeding numbers at colonies, so relationships between survival rates and breeding numbers may be quite complex. According to JNCC '*In all of the oil spill disasters that have occurred during the SMP, guillemots and razorbills have predominated in the seabirds recovered. But despite large numbers of birds being killed, there does not appear to have been any substantial lasting effect on UK seabird populations*' http://jncc.defra.gov.uk/pdf/SMP%20Review%20Report%20Appendix%202%20-%20Impacts%20of%20pressures%20on%20Seabirds.pdf . However, this statement, based on changes in numbers, is not entirely consistent with evidence that recruitment rates of immature birds increased in years following oil-related mortality of adults (Votier et al. 2005, 2008). That observation indicates that although breeding numbers did not change, there is a reduction in the size of the nonbreeding pool that may play an important role in buffering effects of environmental change.

Management to reduce amounts of oil pollution in UK waters would contribute to increasing survival rates of common guillemots. While it is reasonable to assume that strenuous efforts are taken to minimize risk of serious oil spills, survival rates of common guillemots (and presumably of razorbills) could be increased if incidence of oil spills could be reduced in the wintering areas used by common guillemots and razorbills. Preventing major accidents may be very difficult since there is already a very considerable effort to avoid such disasters. Preventing illegal discharges of oil that contribute to chronic oil pollution might be improved. Enforcement of laws prohibiting discharge of oil into the sea, with increased detection of culprits and increased effort into successful prosecution of offenders could reduce the amount of oil illegally entering UK waters.

19. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR RAZORBILL

19.1 Razorbill ecology

SPEC 4 Favourable conservation status (secure) but concentrated in Europe. The world population is around 610,000 to 630,000 pairs, with two subspecies, *torda* and *islandica*. The nominate subspecies breeds in Denmark, Norway, Russia, Finland, Sweden, Greenland, Canada and USA. The subspecies *islandica* breeds in the British Isles, Faroes, Iceland, France and Germany. The largest population is in Iceland (380,000 pairs) (Mitchell et al. 2004).

19.1.1 The species in the British Isles

Seabird 2000 found 110,000 pairs in Great Britain the Isle of Man and Channel Islands, and 35,000 pairs in Ireland (Mitchell et al. 2004). Razorbills breed in colonies on cliff ledges, on sea stacks, or under boulders below cliffs. Nest sites differ from those of common guillemots, with razorbill nests more often among boulders and more often scattered rather than in high-density groups. They lay a single egg onto bare rock, but will replace lost eggs around 14 days later. While breeding, razorbills forage by diving to catch fish by underwater pursuit, feeding chicks especially on sandeels and sprats, and generally taking smaller fish than caught by common guillemots.

The UK SPA suite for this species holds around 81,000 pairs (76% of the British total and 26% of the all-Ireland total) across 19 sites: Cape Wrath, East Caithness Cliffs, Fair Isle, Firth of Forth Islands, Flamborough Head and Bempton Cliffs, Flannan Isles, Foula, Fowlsheugh, Handa, Mingulay and Berneray, North Caithness Cliffs, North Rona and Sula Sgeir, Rathlin Island, Shiant Isles, Skomer and Skokholm, St Abb’s Head to Fast Castle, St Kilda, Troup Pennan and Lion’s Heads, West Westray.

Adult survival rate averages 0.905 (del Hoyo et al. 1996) and most breed for the first time when 4 or 5 years old (Cramp and Simmons 1977-1994). British razorbills generally move south after breeding, wintering from British waters to Iberia including the western Mediterranean (Wernham et al. 2002). However, many British razorbills, especially adults, remain in British waters all year round. Rather few foreign-ringed razorbills have been recovered in Britain, but these together with biometrics of tideline corpses indicate that some birds from Scandinavia (from the subspecies *torda*) winter in British waters and some birds from Iceland do too (Wernham et al. 2002).

19.1.2 Present, and likely future trends

Seabird 2000 recorded about 216,000 individual razorbills at colonies in Britain and Ireland, with 64% in Scotland, 24% in Ireland, 6% in England the Isle of Man and Channel Islands, and 6% in Wales (Mitchell et al. 2004). This was 23% higher than found by the SCR Census in 1985-88, which itself was 5% higher than the number recorded in Operation Seafarer in 1969-70 (Mitchell et al. 2004). The JNCC index of breeding numbers of razorbills at UK colonies (Figure 19.1.1) indicates a further increase in numbers of about 15% from 2000 to 2011, but the trend differs between northern and southern colonies. The index for Scotland indicates a small decrease from 2000 to 2011 (Figure 19.1.2), whereas in Wales numbers have continued to increase (Figure 19.1.3). Predicting future changes in numbers is very difficult, but the conditions appear to be more favourable in the south and less so in the north.

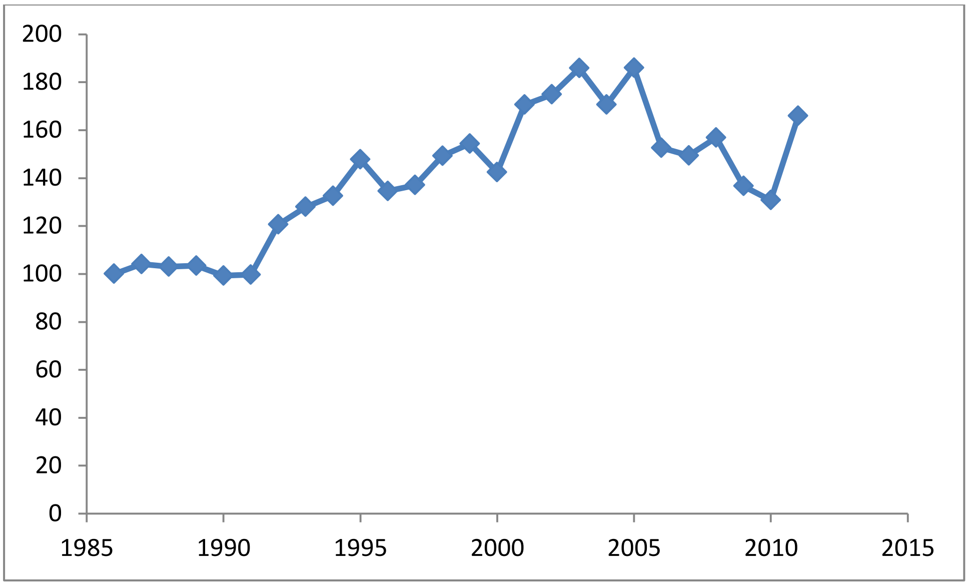

Figure 19.1.1. Razorbill breeding population index for the UK. Data from JNCC online database.

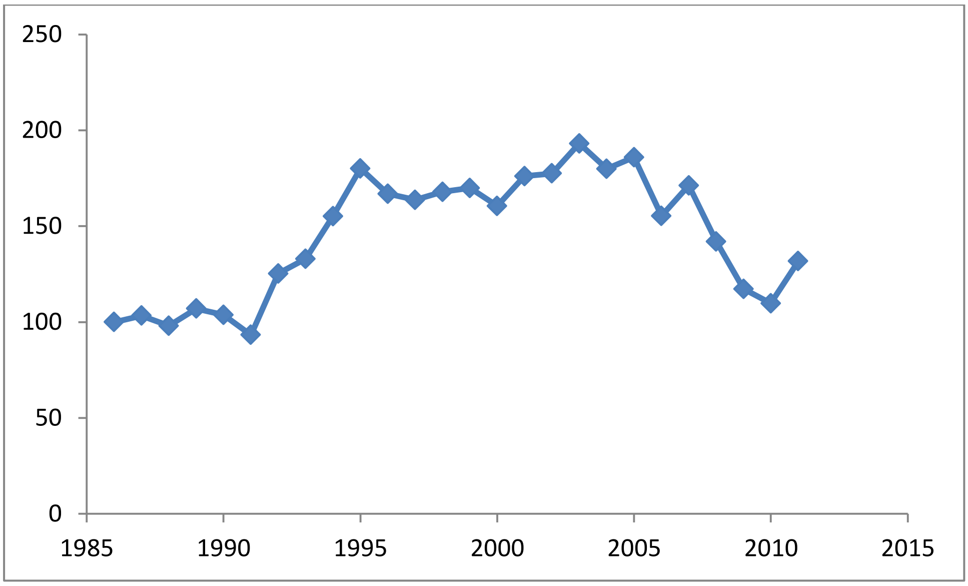

Figure 19.1.2. Razorbill breeding population index for Scotland. Data from JNCC online database.

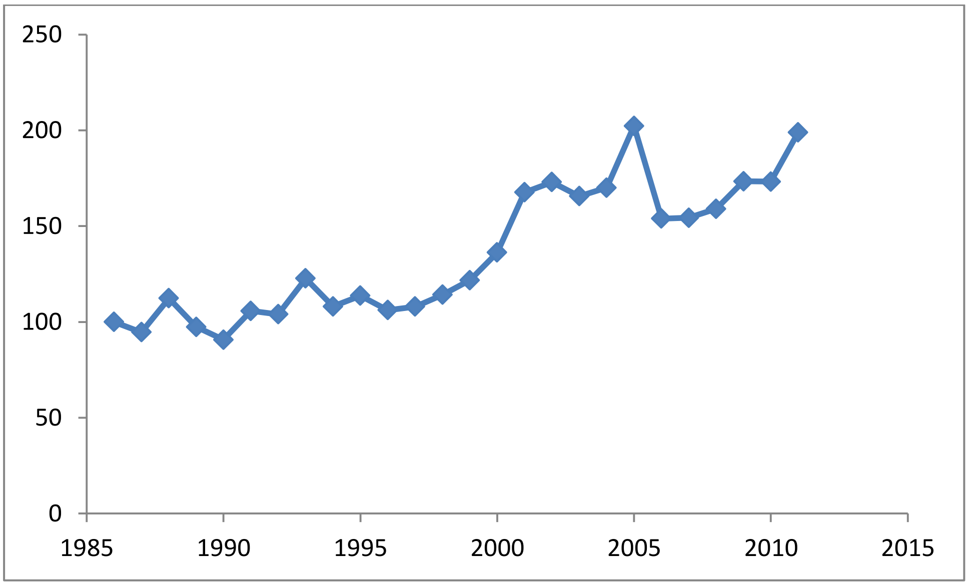

Figure 19.1.3. Razorbill breeding population index for Wales. Data from JNCC online database.

19.1.3 Factors affecting survival rates

Mean survival rates of adult razorbills have been reported as 0.919 at Hornoya north Norway (Sandvik et al. 2005), 0.905 at the Isle of May (Harris et al. 2000), 0.898 at Skomer (Hudson 1979), 0.901 at Skomer (Poole et al. 1998), 0.895 in Canada (Chapdelaine 1997), 0.92 at the Shiants (Steventon 1979), 0.96 at Clo Mor (Lloyd and Perrins 1977), 0.81 to 0.89 at Skokholm (Lloyd and Perrins 1977), 0.89 from British ring recovery data (Lloyd 1974) and 0.914 from British ring recovery data (Mead 1974). Sandvik et al. (2005) showed that razorbill survival rate decreased with increasing SST at Hornoya, but none of the other studies included environmental covariates, so the influences of environmental factors on survival rates of British razorbills remain uncertain. Estimation of the extent to which management might alter survival rates is therefore extremely difficult. However, it is recognised that razorbills can be subject to mass mortality incidents ('wrecks') in autumn and winter which seem to be mainly due to food shortage (scarcity of small pelagic fish). For example, a wreck in autumn 2007 involved thousands of dead razorbills, especially in the Skagerrak and Kattegat, was attributed to food shortage (Heubeck et al. 2011). A similar wreck in 1983 was attributed to low abundance of sprats in the north-western North Sea (Blake 1984).

19.1.4 Breeding success in the British Isles

Although productivity in 1988-90 was poor in Shetland, it improved in 1991, and growth rates of chicks at Hermaness were higher in 1991, indicating a better food supply. It is known that there was high recruitment of sandeels around Shetland in 1991(Walsh et al. 1992). In 1992, productivity was high at all study colonies, and chick weights at Hermaness were above those in 1988-90 which suggests that this improved productivity was due to improved food supply (Walsh et al. 1993). Walsh et al. (1995) noted that Razorbill nest sites on open ledges on Skomer were noticeably less successful than enclosed sites, probably reflecting exposure to predation. In 2001, Razorbill productivity was 0.7 chicks per pair across the 6 monitored colonies, similar to the average for 1986-2000 (Mavor et al. 2002). On Canna in 2003, eggs depredated by brown rats were found and it appeared that rats had been responsible for redistribution of breeding Razorbills into areas inaccessible to rats (Mavor et al. 2004). In 2004, there was low productivity of Razorbills in the Northern Isles, but higher productivity further south. Food samples indicated a scarcity of large sandeels in the north, but no evidence of food shortage in the south, strongly suggesting that reduced productivity was caused by lack of suitable food. In 2005, chicks at Fair Isle were noted to be underweight for their ages and some lost weight rather than growing, providing strong evidence for food shortage being the cause of low productivity (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for razorbills as follows. Breeding success was generally poor, averaging 0.48 chicks per pair. In Shetland, chick survival was very poor, attributed to a lack of food (sandeels). Chicks showed low rates of weight gain, and some cases of weight loss, consistent with severe food shortage. On the Isle of May, productivity at 0.64 chicks per pair was close to the long-term average, and adults appeared to be having little difficulty finding food around that colony.

Table 19.1.1 Meta-analysis of main factors contributing to reduced productivity of Razorbills at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Food shortage	8
Gull predation	1
Rat predation	1

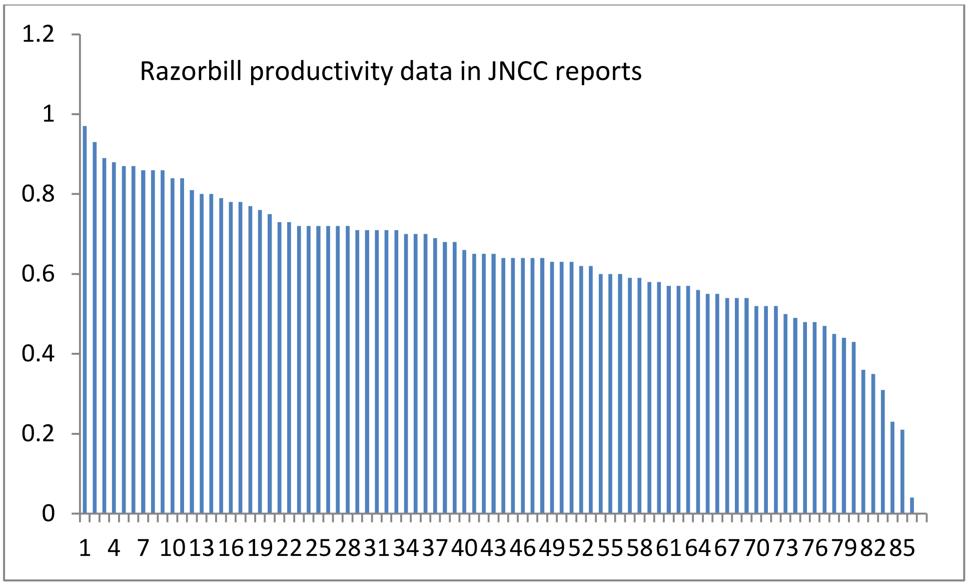

Figure 19.1.4. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of razorbills at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

19.2 Management options

	19.2.1 Closure of sandeel and sprat fisheries in all UK waters	19.2.2 Closure of sandeel and sprat fisheries in wintering areas	19.2.3 Eradicate rats	19.2.4 Prevent oil spills
Evidence of success for this species	Low C=Low	Low C=Low	Low C=Low	Low C=Low
Evidence of success for similar species	High C=Mod	Low C=Low	High C=High	High C=Mod
Cost-effectiveness	Uncertain C=Low	Uncertain C=Low	High C=Mod	Uncertain C=High
Feasibility	Moderate C=Low	Moderate C=Low	High C=High	Low C=High
Practicality	Moderate C=Low	Moderate C=Low	Moderate C=High	Low C=High
Applies at SPA populations	Yes C=High	Uncertain C=High	Few C=High	Yes C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

19.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs or in all UK waters

Food shortage is implicated as a cause of reduced productivity at some colonies in some years. Mean foraging range of razorbills is around 24 km and maximum foraging range is up to 95 km (Langston 2010, Thaxter et al. 2012), so closure of sandeel and sprat fishing within 100 km of SPAs should increase productivity and adult survival, but that would amount to closure of sandeel and sprat fishing in British waters given the wide distribution and large number of SPAs for these species. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

An area off the east coast of Scotland was closed to sandeel fishing from 2000 because kittiwake breeding success in the area had fallen to very low levels after intensive fishing for sandeels for several years by the Danish industrial fleet on the sand banks close to the east coast of Scotland. The aim of this fishery closure was to allow sandeel stocks in the area to recover (which sampling of sandeels in the area confirmed to be the consequence; Greenstreet et al. 2006), and kittiwake productivity was used as the proxy for sandeel abundance in the absence of data on sandeel stock biomass (Camphuysen 2005). Frederiksen and Wanless (2006) reviewed the evidence that this closure increased productivity of kittiwakes and other seabirds. They found clear evidence that the closure resulted in increased productivity of kittiwakes within the study area compared with a control area outside the closure zone, results which agreed with the earlier findings of Frederiksen et al. 2004) that the sandeel fishery in this area reduced productivity of kittiwakes on the Isle of May during the years of the fishery. Productivity of kittiwakes did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (the difference was 0.28 chicks per nest and statistically highly significant ($p<0.0001$). Analysis of productivity data for monitored colonies of other seabirds was based on much smaller sample sizes, and showed less clear results due to chance variations. However, for Sandwich tern the pattern was similar, within the study zone there was higher productivity in no fishery years (0.62 chicks per pair) than in fishery years (0.38 chicks per pair) whereas productivity in the control area was consistently high. However, high variation and small sample size (there were only two colonies within the closure zone) resulted in this difference not being statistically significant. Data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie was more or less completely deserted from 1992-1998 and that this species did not breed on the Isle of May until 1999. Combined with the observed (non-significant) reduction in productivity in the closure zone during the fishery years, Frederiksen and Wanless (2006) concluded that '*this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes*'. For Arctic tern, European shag, common guillemot, razorbill and Atlantic puffin, the closure could not be shown to have any strong effect on productivity by analysis of data for each individual species in turn, but a one-way MANOVA with data for all species showed a significant effect of the fishery closure on productivity of the seabird community, but this was predominantly driven by the strong and clear effect on kittiwakes. So, despite the clear evidence from the JNCC monitoring that food shortage has a strong impact on razorbill productivity, analysis of the data from the Isle of May and east Scottish sites from the 1990s did not show this effect. Possibly food fish abundance needs to fall to lower levels to affect razorbills than it does to affect kittiwakes and terns (Furness and Tasker 2000).

In Shetland, sandeel stock biomass fell rapidly in the late 1980s during the period of a local industrial fishery for sandeels, and that fishery was closed in 1990. Breeding success of kittiwakes in Shetland in 1990 was 0.12 chicks per pair. In 1991, the first year of the sandeel fishery closure, sandeel recruitment was higher than it had been in any of the previous seven years during heavy fishing, and kittiwake productivity in 1991 at 0.57 chicks per pair was the highest at monitored Shetland colonies since 1986 (JNCC data and ICES sandeel stock data). However, it is unclear whether the high recruitment in 1991 was caused by the fishery closure or whether the timing was coincidental. This fishery re-opened in 1994 based on high recruitment in the early 1990s, but local fishermen did not resume sandeel fishing to a significant extent, and the fishery effectively became extinct when sandeel abundance fell again in the early 2000s.

Closure of all sandeel and sprat fisheries in UK waters would bring the UK into the same management position as exists in the USA, where fishing on the small pelagic fish such as sandeels that are keystone species for marine food webs (including large predatory fish of high commercial importance) is prohibited. The considerable body of evidence shows that kittiwake productivity and adult survival would benefit from higher average abundances of these small prey fish, and the JNCC data together with the detailed research at the Isle of May (Frederiksen et al. 2004, Frederiksen and Wanless 2006) show that the depletion of sandeel stocks has been the single most influential factor reducing productivity of kittiwakes at colonies in the British Isles, but evidence that this would affect razorbill productivity is less strong. Closure of sandeel fisheres in UK waters would have little direct cost for British fishermen, since the fishery is almost entirely carried out by the Danish fishing fleet. Closure of sprat fisheries in UK waters would affect small numbers of fishermen in NW Scotland (there is a small fishery for sprats carried out by a few local fishing vessels near to Mull in Argyll and there is a small fishery for sprats carried out by a few local fishing vessels in the English Channel (ICES 2013). Closure of these fisheries would be likely to have some beneficial effects for commercially important fish stocks such as cod and whiting (Greenstreet et al. 2006).

19.2.2 Closure of sandeel and sprat fishing in areas where these species are aggregated in winter

So-called ‘wrecks’ of razorbills occur in autumn/winter in some years (Blake 1984), suggesting that winter may be a critical season determining survival rates. If so, protection of prey fish stocks that are important as winter food may be of greatest importance. British razorbills mainly overwinter in British waters, though often south of their breeding areas, so closure of sprat and sandeel fisheries in southern Britain may be most effective in protecting their food supply. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider.

There are localised fisheries for sprats in UK waters, by Scottish fishermen off west Scotland (usually close to Mull), by English fishermen in the English Channel (Lyme Bay), and occasionally as a result of the activity of foreign industrial fishing fleets (ICES 2013). There have in the past been fisheries on sprats in the Moray Firth and in the Firth of Forth, fisheries which appear to have caused local depletion of those stocks and then been closed (Jennings et al. 2012). There has in the past been a fishery by Scottish fishermen on sandeels in Shetland which was closed in 1991 due to the depletion of that stock which has still not recovered, and the large industrial fishery for sandeels in the North Sea has moved around over the years to exploit different stocks within the North Sea, resulting in depletion of many of the distinct sandeel stocks in the northern North Sea, but continued exploitation by Danish fishermen of sandeel stocks in English waters (ICES 2010, ICES 2012). All of these sandeel and sprat stocks are likely to represent important preferred food of razorbills in UK waters in winter. Closure of sandeel fishing in UK waters would have low cost for UK fishermen, as almost the entire fishery is carried out by Danish fishermen. Closure of sprat fishing in UK waters would affect a small number of fishing vessels in west Scotland and in Lyme Bay, and limit potential development of sprat fisheries on stocks that are currently unfished (e.g. in Irish Sea, Clyde, Thames, Firth of Forth, Moray Firth).

19.2.3 Eradication of rats

Brown rats are a factor reducing productivity at a few colonies. Eradication of invasive alien rats would allow common guillemot and razorbill productivity to increase at colonies where this predator is present. However, this applies at a very small proportion of colonies and it is very unlikely that rat eradication would be an effective conservation measure for razorbills.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and

Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

19.2.4 Prevent risk of major oil spills near to SPAs

The effect of oil pollution on seabird populations is surprisingly uncertain. Votier et al. (2005) showed that major oil spill incidents doubled mortality of common guillemots from a Welsh colony in the years in which these occurred. The same is likely to be true of razorbills, given the similarity in their at sea behaviour. Major incidents such as the oil spill accidents considered by Votier et al. (2005) probably kill fewer seabirds than the chronic discharges of small quantities of oil from illegal discharges (Dunnet 1982). So the high influence of these major oil spills on common guillemot survival rates probably understates the influence of oil pollution. Nevertheless, previous attempts to relate changes in numbers of common guillemots or razorbills attending colonies to oil spill mortality have often failed to show any detectable impact (see for example the lack of impact of the Braer oil spill in Shetland on seabird breeding numbers or breeding success http://www.nature-shetland.co.uk/birdclub/braer/Part10.html), suggesting that the consequence of increased mortality is not seen at the level of breeding numbers at colonies, so relationships between survival rates and breeding numbers may be quite complex. According to DEFRA '*In all of the oil spill disasters that have occurred during the SMP, guillemots and razorbills have predominated in the seabirds recovered. But despite large numbers of birds being killed, there does not appear to have been any substantial lasting effect on UK seabird populations*' http://jncc.defra.gov.uk/pdf/SMP%20Review%20Report%20Appendix%202%20-%20Impacts%20of%20pressures%20on%20Seabirds.pdf.

Nevertheless, management to reduce amounts of oil pollution in UK waters would contribute to increasing survival rates of razorbills. While it is reasonable to assume that strenuous efforts are taken to minimize risk of serious oil spills, survival rates of razorbills could presumably be increased if incidence of oil spills could be reduced in their wintering areas. Preventing major accidents may be very difficult since there is already a very considerable effort to avoid such disasters. Preventing illegal discharges of oil that contribute to chronic oil pollution might be improved. Enforcement of laws prohibiting discharge of oil into the sea, with increased detection of culprits and increased effort into successful prosecution of offenders could reduce the amount of oil illegally entering UK waters.

20. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR ATLANTIC PUFFIN

20.1 Atlantic puffin ecology

SPEC 2 (Unfavourable conservation status (vulnerable) and concentrated in Europe). The Atlantic puffin is found around much of the North Atlantic and adjacent seas, with three sub-species. The nominate race *F. a. arctica* breeds at mid-latitudes, in eastern North America, Iceland, north Norway to southern Novaya Zemyla. *F. a. naumanni* breeds further north in high Arctic waters in Greenland, Svalbard and northern Novaya Zemyla. *F. a. grabae* breeds in southern areas of the species' range; in southern Norway, the Faeroes, the British Isles, and France. The total population of *F. a. grabae* (the biogeographical population) is estimated at 901,000 pairs (Cramp 1977-1994; Lloyd et al. 1991). The combined British and Irish population of 469,500 pairs in 1985–1987 made up 52% of this biogeographical population (Stone et al. 1997).

20.1.1 The species in the British Isles

Puffins are crevice and burrow-nesting seabirds, with colonies often found in highly inaccessible locations, such as on grassy slopes half way down cliffs. They can be very difficult to census, and their activity at colonies fluctuates dramatically, often showing large variations in colony attendance over periods of a few days throughout the breeding season. As a consequence trends in puffin colony sizes are often uncertain. The majority of the British and Irish population breed in Scotland with the St. Kilda archipelago holding the largest colony (Lloyd et al. 1991). Other principal breeding areas include Shetland (particularly Fair Isle and Foula), Orkney, Sule Skerry, the Isle of May and the Western Isles including the Shiant Islands. The main breeding areas in England are the Farne Islands, Coquet Island and the Bempton-Flamborough cliffs. Major colonies in Wales include Skomer, Skokholm and Ynys Gwylans. Rathlin Island (Antrim) is the main breeding location in Northern Ireland (Lloyd et al. 1991).Puffins feed on small pelagic fish (Hislop and Harris 1985), mostly caught in the upper 10 m of the sea by shallow dives from the surface. They carry multiple fish to the burrow to feed their chick (unlike common guillemots which carry a single fish), Breeding numbers at some colonies apparently declined considerably in early to mid 20th century. The very large colony on Ailsa Craig was apparently extirpated by brown rats (Zonfrillo 2001). Tens of thousands of pairs apparently disappeared from Skomer and St Kilda (Cramp et al. 1974). More recent studies have suggested roughly stable numbers at many colonies since the 1970s, although large increases occurred at the Isle of May, Farne Islands and Coquet Island. Food availability appears to be a major factor influencing population change. The cessation of increase in the Isle of May colony coincided with a reduction in the numbers of sprats in the North Sea and a doubling of the annual mortality rate of breeding adults (Harris and Wanless 1991). The large decrease in the northern Norway population in the 1970s and 1980s was linked to a crash in Atlantic/Scandinavian herring stocks (Anker-Nilssen and Barrett 1991). Declines have often been attributed to local factors such as oiling or mammalian predation (Harris et al. 1997). Conditions in the wintering areas also appear to be critical for puffins (Harris et al. 2005, 2010). Studies of puffin egg size over the last 30 years also indicate that climate change, through its effect on puffin prey stocks, has been causing a long-term reduction in egg size (Barrett et al. 2012). Finney et al. (2003) showed that puffin recruitment into a colony is influenced by the density of breeding gulls around the puffin colony. Puffin recruitment was reduced in areas with larger numbers of gulls. Furthermore, puffins breeding in areas of the colony that were gull free achieved higher rates of chick-feeding (because fewer fish were stolen by gulls), although there was no overall impact on productivity in the year of the study (Finney et al. 2001).

The UK's SPA suite for puffin supports around 470,000 pairs. This amounts to the vast majority of the British breeding population, and about 12% of the all-Ireland population. The SPA suite contains 21 sites where puffin has been listed as a qualifying species; Canna and Sanday, Cape Wrath, Coquet

Island, East Caithness Cliffs, Fair Isle, Farne Islands, Firth of Forth Islands, Flamborough Head and Bempton Cliffs, Flannan Isles, Foula, Hermaness Saxa Vord and Valla Field, Hoy, Mingulay and Berneray, North Caithness Cliffs, North Rona and Sula Sgeir, Noss, Rathlin Island, Shiant Isles, Skomer and Skokholm, St Kilda, and Sule Skerry and Sule Stack.

Adult survival rate averages 0.95 (del Hoyo et al. 1996) and most breed for the first time when 4 or 5 years old (Cramp and Simmons 1977-1994). After breeding, puffins disperse away from colonies. Birds breeding on North Sea coasts are thought to remain predominantly within the North Sea during winter, while birds breeding to the north and west of Britain disperse mainly into the North Atlantic, apparently ranging widely across the ocean at low densities (Harris 2011).

20.1.2 Present, and likely future trends

Seabird 2000 recorded about 600,000 apparently occupied burrows (approximately equivalent to pairs) in Britain and Ireland, of which 82% were in Scotland, 13% in England, 4% in Ireland and 2% in Wales (Mitchell et al. 2004). This represented a substantial (104%) increase in England and a small (13%) increase in Scotland since 1985. The greatest increase has occurred in SE Scotland and NE England (Isle of May, Inchkeith, Farne Islands and Coquet Island), Despite sandeel shortages in Shetland and puffin breeding failures at many Shetland colonies, numbers in Shetland have not shown clearly detectable decreases, except at the formerly very large colony at Foula (Mitchell et al. 2004). The JNCC index of breeding numbers of puffins at UK colonies does not provide a reliable population trend because the number of colonies where puffins are monitored is small, and count accuracy is considered to be relatively low (JNCC database). Foster and Marrs (2012) were unable to estimate the current puffin population trend for Scotland for the same reasons. Puffin populations appear to be rather robust to environmental pressures such as food shortage (Mitchell et al. 2004) and while impacts of climate change and shortage of sandeels are likely, these may be difficult to detect in view of the large uncertainty and variability in counts of puffin numbers at colonies. Predicting future trends is difficult, but it is also likely that trends will be quite difficult to detect from empirical data except at the very few intensively monitored colonies such as the Isle of May.

20.1.3 Factors affecting survival rates

Mean survival rates of breeding adult puffins averaged 0.93 at five European colonies where long time series of data are available (Skomer, Isle of May, Fair Isle, Røst, Hørnoya), and did not differ significantly among colonies (Harris et al. 2005), though the rate is slightly lower than the previous estimate of 0.95 for Skomer and 0.975 for Isle of May (Harris et al. 1997). The survival rates were considered by Harris et al. (2005) to be 'virtually identical' among the five colonies, despite differences in rates of growth or decline between colonies, indicating that local demography was not determined primarily by adult survival rate. Annual survival rates at four of the five colonies showed a significant negative correlation with sea surface temperature (SST) two years earlier. The exception (Hørnoya) was the northernmost colony (in Arctic Norway), where survival did not relate to local SST. Harris et al. (2005) point out that most mortality of adult puffins appears to occur during winter, so may be driven by environmental conditions in the wintering range of the birds. However, puffins from the Isle of May wintered predominantly within the North Sea, puffins from Skomer wintered predominantly west and south of the British Isles, while puffins from Røst wintered predominantly north of the British Isles, so the similarity of adult survival rates is despite these populations showing largely non-overlapping winter distributions. The influence of SST on adult survival rates was interpreted by Harris et al. (2005) as indicating 'bottom-up' effects of sea temperature on food abundance (such as sandeel stocks, and herring recruitment) affecting puffin survival.

20.1.4 Breeding success in the British Isles

Breeding success of Atlantic puffins is monitored at a rather small number of colonies around the British Isles; factors affecting productivity may differ between monitored sites and those that are not monitored (which tend to be less accessible locations). However, the JNCC monitoring reports identify food shortage as the most frequently reported factor causing breeding failure of puffins at monitored colonies. This has been especially frequent at Shetland, but has also affected colonies further south in the UK in some years. Flooding by intense rainfall events during May or June was also recorded several times (Table 20.1.1). Several research studies have reported on the influence of food availability on puffin productivity (reviewed in Harris 2011), reaching a clear conclusion from the evidence that scarcity of small pelagic fish (such as sandeels, juvenile herring, sprats or capelin) tends to result in breeding failure of puffins, with chicks coming out of burrows and then being taken by predators or dying of starvation or exposure. Productivity may be slightly influenced also by the presence of large numbers of kleptoparasites (great skuas, Arctic skuas, gulls, jackdaws), and in some colonies may be influenced by predators (including rats, feral cats, ferrets, great black-backed gulls and great skuas).

Table 20.1.1 Meta-analysis of main factors contributing to reduced productivity of Atlantic puffins at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Food shortage	16
Flooding of burrows	8
Feral cat predation	2
Ferret predation	1

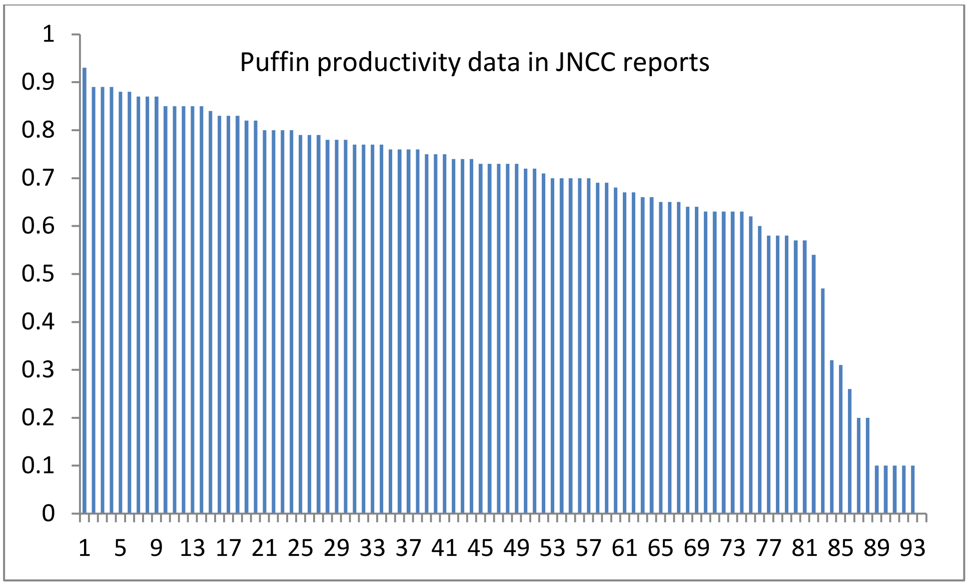

Figure 20.1.1. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of Atlantic puffins at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

20.2 Management options

	20.2.1 Closure of sandeel and sprat fisheries in UK waters	20.2.2 Eradicate rats	20.2.3 Reduce oil spills
Evidence of success for this species	Low C=Low	Low C=Low	Low C=Low
Evidence of success for similar species	High C=Mod	High C=High	High C=Mod
Cost-effectiveness	Uncertain C=Low	High C=High	Uncertain C=Mod
Feasibility	Moderate C=Low	High C=High	Low C=High
Practicality	Moderate C=Low	Moderate C=High	Low C=High
Applies at SPA populations	Yes C=High	Some C=High	Yes C=High

*Confidence in each score is listed as ‘High’ ‘Moderate’ or ‘Low’ based on amount and consistency of evidence

20.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs or in all UK waters

Food shortage is implicated as a cause of reduced productivity at some colonies in some years. Mean foraging range of Atlantic puffins was only 4 km at the one colony where this has been measured, but the maximum foraging range averaged about 100 km across several studies (Thaxter et al. 2012), so closure of sandeel and sprat fishing within 100 km of SPAs should increase productivity and adult survival, but that would amount to closure of sandeel and sprat fishing in British waters given the wide distribution and large number of SPAs for these species. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

An area off the east coast of Scotland was closed to sandeel fishing from 2000 because kittiwake breeding success in the area had fallen to very low levels after intensive fishing for sandeels for several years by the Danish industrial fleet on the sand banks close to the east coast of Scotland. The aim of this fishery closure was to allow sandeel stocks in the area to recover (which sampling of sandeels in the area confirmed to be the consequence; Greenstreet et al. 2006), and kittiwake productivity was used as the proxy for sandeel abundance in the absence of data on sandeel stock biomass (Camphuysen 2005). Frederiksen and Wanless (2006) reviewed the evidence that this closure increased productivity of kittiwakes and other seabirds. They found clear evidence that the closure resulted in increased productivity of kittiwakes within the study area compared with a control area outside the closure zone, results which agreed with the earlier findings of Frederiksen et al. 2004) that the sandeel fishery in this area reduced productivity of kittiwakes on the Isle of May during the years of the fishery. Productivity of kittiwakes did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (the difference was 0.28 chicks per nest and statistically highly significant (p<0.0001). Analysis of productivity data for monitored colonies of other seabirds was based on much smaller sample sizes, and showed less clear results due to chance variations. However, for Sandwich tern the pattern was similar, within the study zone there was higher productivity in no fishery years (0.62 chicks per pair) than in fishery years (0.38 chicks per pair) whereas productivity in the control area was consistently high. However, high variation and small sample size (there were only two colonies within the closure zone) resulted in this difference not being statistically significant. Data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie was more or less completely deserted from 1992-1998 and that this species did not breed on the Isle of May until 1999. Combined with the observed (non-significant) reduction in productivity in the closure zone during the fishery years, Frederiksen and Wanless (2006) concluded that '*this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes*'. For Arctic tern, European shag, common guillemot, razorbill and Atlantic puffin, the closure could not be shown to have any strong effect on productivity by analysis of data for each individual species in turn, but a one-way MANOVA with data for all species showed a significant effect of the fishery closure on productivity of the seabird community, but this was predominantly driven by the strong and clear effect on kittiwakes. So, despite the clear evidence from the JNCC monitoring that food shortage has a strong impact on puffin productivity, analysis of the data from the Isle of May and east Scottish sites from the 1990s did not show this effect. Possibly food fish abundance needs to fall to lower levels to affect puffins than it does to affect kittiwakes and terns (Furness and Tasker 2000).

In Shetland, sandeel stock biomass fell rapidly in the late 1980s during the period of a local industrial fishery for sandeels, and that fishery was closed in 1990. Breeding success of kittiwakes in Shetland in 1990 was 0.12 chicks per pair. In 1991, the first year of the sandeel fishery closure, sandeel recruitment was higher than it had been in any of the previous seven years during heavy fishing, and kittiwake productivity in 1991 at 0.57 chicks per pair was the highest at monitored Shetland colonies since 1986 (JNCC data and ICES sandeel stock data). However, it is unclear whether the high recruitment in 1991 was caused by the fishery closure or whether the timing was coincidental. This fishery re-opened in 1994 based on high recruitment in the early 1990s, but local fishermen did not resume sandeel fishing to a significant extent, and the fishery effectively became extinct when sandeel abundance fell again in the early 2000s.

Closure of all sandeel and sprat fisheries in UK waters would bring the UK into the same management position as exists in the USA, where fishing on the small pelagic fish such as sandeels that are keystone species for marine food webs (including large predatory fish of high commercial importance) is prohibited. The considerable body of evidence shows that kittiwake productivity and adult survival would benefit from higher average abundances of these small prey fish, and the JNCC data together with the detailed research at the Isle of May (Frederiksen et al. 2004, Frederiksen and

Wanless 2006) show that the depletion of sandeel stocks has been the single most influential factor reducing productivity of kittiwakes at colonies in the British Isles, but evidence that this would affect puffin productivity is less strong. Closure of sandeel fisheries in UK waters would have little direct cost for British fishermen, since the fishery is almost entirely carried out by the Danish fishing fleet. Closure of sprat fisheries in UK waters would affect small numbers of fishermen in NW Scotland (there is a small fishery for sprats carried out by a few local fishing vessels near to Mull in Argyll) and there is a small fishery for sprats carried out by a few local fishing vessels in the English Channel (ICES 2013). Closure of these fisheries would be likely to have some beneficial effects for commercially important fish stocks such as cod and whiting (Greenstreet et al. 2006).

20.2.2 Eradication of rats

Rats are a factor which may be reducing productivity at a few colonies. Eradication of invasive alien rats could allow puffin productivity to increase at colonies where this predator is present. This applies at a very small proportion of colonies, but these include the Shiants where black rats have been present for around 100 years, co-existing with one of the largest puffin colonies in the British Isles. Whether black rats affect puffin productivity on the Shiants is unclear, as breeding success has not been monitored there. RSPB are currently seeking funding to eradicate black rats from the Shiants, in order to reduce the risk to the puffin colony on the Shiants and to increase prospects for storm petrels and Manx shearwaters to colonise.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

20.2.3 Prevent risk of major oil spills near to SPAs

The effect of oil pollution on seabird populations is surprisingly uncertain. Votier et al. (2005) showed that major oil spill incidents doubled mortality of common guillemots from a Welsh colony in the years in which these occurred. The same is likely to be true of razorbills, given the similarity in their at sea behaviour. Major incidents such as the oil spill accidents considered by Votier et al. (2005) probably kill fewer seabirds than the chronic discharges of small quantities of oil from illegal discharges (Dunnet 1982). So the high influence of these major oil spills on common guillemot survival rates probably understates the influence of oil pollution. Nevertheless, previous attempts to relate changes in numbers of alcids attending colonies to oil spill mortality have often failed to show any detectable impact (see for example the lack of impact of the Braer oil spill in Shetland on seabird breeding numbers or breeding success http://www.nature-shetland.co.uk/birdclub/braer/Part10.html), suggesting that the consequence of increased mortality is not seen at the level of breeding numbers at colonies, so relationships between survival rates and breeding numbers may be quite complex. According to DEFRA '*In all of the oil spill disasters that* *have occurred during the SMP, guillemots and razorbills have predominated in the seabirds recovered. But despite large numbers of birds being killed, there does not appear to have been any substantial lasting effect on UK seabird populations'* http://jncc.defra.gov.uk/pdf/SMP%20Review%20Report%20Appendix%202%20-%20Impacts%20of%20pressures%20on%20Seabirds.pdf.

There is no evidence to suggest that oil spills contribute extensively to puffin mortality. Nevertheless, management to reduce amounts of oil pollution in UK waters would contribute, if only to a small extent, to increasing survival rates of puffins. While it is reasonable to assume that strenuous efforts are taken to minimize risk of serious oil spills, survival rates of puffins could presumably be increased slightly if incidence of oil spills could be reduced in their wintering areas. Preventing major accidents may be very difficult since there is already a very considerable effort to avoid such disasters. Preventing illegal discharges of oil that contribute to chronic oil pollution might be improved. Enforcement of laws prohibiting discharge of oil into the sea, with increased detection of culprits and increased effort into successful prosecution of offenders could reduce the amount of oil illegally entering UK waters.

21. SPECIES SPECIFIC RECOMMENDATIONS

Many of the seabird populations around the British Isles are currently declining in numbers after reaching long-term peak numbers late in the 20th century. Current declines are thought to be due to a wide range of factors, but primarily to climate change impacts on food abundance (especially sandeels), effects of changes in fisheries management such as reducing volumes of discards, and presence of alien mammal predators. Breeding numbers of seabirds on many SPAs are now below levels present at site designation; site condition monitoring is likely to lead to a conclusion of unsatisfactory condition for these populations. It is recommended that consideration is given to the fact that great skua, herring gull, great black-backed gull and lesser black-backed gull numbers in the UK may be elevated above naturally sustainable levels by historical provision of fishery discards (and for the gulls also urban refuse), and therefore that reduced numbers of these species at some UK colonies might be a desirable conservation objective.

If management action to increase survival or productivity of red-throated divers was considered to be desirable, the provision of nest platforms on selected breeding lochs (7.2.1) is recommended as the most cost-effective approach.

If management action to increase survival or productivity of Manx shearwaters was considered to be desirable, the eradication of brown rats on Rum SPA, feral cats on Eigg, brown rats and feral cats on the Calf of Man, brown rats and feral cats on the Isles of Scilly, ferrets on Rathlin Island, feral cats on Fetlar, and feral cats on Foula (8.2.1) is recommended as the most cost-effective approach.

If management action to increase survival or productivity of northern gannets was considered to be desirable, thetermination of the licenced harvest of young gannets from Sula Sgeir (9.2.2) is the most cost-effective approach. Such action would have only a very limited buffering capacity, but no other feasible management options that would have a greater effect on gannet demography have been identified in this evidence review. Quantification of fishery bycatch of gannets would be useful to assess if reducing bycatch rate could provide a potential compensation.

If management action to increase survival or productivity of Arctic skuas was considered to be desirable, the provision of supplementary food to breeding pairs (10.2.2) is recommended as the most cost-effective approach, providing carried out with necessary care to avoid attracting larger scavengers, and may need associated control of nearby great skuas'.

It is recommended that consideration is given to the fact that great skua numbers in the UK are elevated above naturally sustainable levels by historical provision of fishery discards, and therefore that reduced numbers of this species at UK colonies might be a desirable conservation objective. If management action to increase survival or productivity of great skuas was considered to be desirable, we suggest that there is no truly cost-effective approach to achieve this. Two options that could be considered further are closure of areas within 50 km of great skua SPAs to sandeel and sprat fishing (11.2.1) and supplementary feeding of great skuas at SPA colonies (11.2.2). Quantification of fishery bycatch of great skuas would be useful to assess if reducing bycatch rate could provide a potential compensation.

If management action to increase survival or productivity of lesser black-backed gulls was considered to be desirable, the end of culling of breeding adults (12.2.3) is recommended as the most cost-effective approach, with mink eradication from islands with colonies (12.2.1) and establishment of predator-proof fencing around mainland colonies subject to mammal predation, especially foxes (12.2.2) as potential further measures.

If management action to increase survival or productivity of herring gulls was considered to be desirable, the end of culling of breeding adults (13.2.3) is recommended as the most cost-effective approach, with mink eradication from islands with colonies (13.2.1) and establishment of predator-proof fencing around mainland colonies subject to mammal predation, especially foxes (13.2.2) as potential further measures.

If management action to increase survival or productivity of great black-backed gulls was considered to be desirable, the end of culling of breeding adults (14.2.1) is recommended as the most cost-effective approach, with mink eradication from islands with colonies (14.2.2)

If management action to increase survival or productivity of kittiwakes was considered to be desirable, the closure of all UK waters to fishing directed at sandeels and directed at sprats (15.2.1) is recommended as the most cost-effective approach. A further action could be exclusion of great skuas from holding breeding territories in buffer zones around kittiwake colonies (15.2.6).

If management action to increase survival or productivity of Sandwich terns was considered to be desirable, theexclusion of foxes from colonies by predator-proof fencing (16.2.5) and protection of colonies from flooding (16.2.7) are recommended as the most cost-effective approaches.

If management action to increase survival or productivity of common terns was considered to be desirable, the eradication of mink from islands with natural colonies of common terns (17.2.2) and deployment of predator-proof nesting rafts (17.2.7) are recommended as the most cost-effective approaches, and possibly the control of local gulls that specialise in eating tern chicks (17.2.8).

If management action to increase survival or productivity of common guillemots was considered to be desirable, the closure of all UK waters to fishing directed at sandeels and directed at sprats (18.2.1) is recommended as the most cost-effective approach.

If management action to increase survival or productivity of razorbills was considered to be desirable, the closure of all UK waters to fishing directed at sandeels and directed at sprats (19.2.1) is recommended as the most cost-effective approach.

If management action to increase survival or productivity of Atlantic puffins was considered to be desirable, the closure of all UK waters to fishing directed at sandeels and directed at sprats (20.2.1) is recommended as the most cost-effective approach, and possibly eradication of rats from islands with large colonies of puffins (20.2.2).

Notwithstanding specific requirements of the Birds Directive, we believe that we should be seeking to intervene to ensure functioning ecosystems rather than attempting to manage individual populations as if these do not interact. If management actions are required for several species, some of these recommended approaches may be effective across several species (for example closure of sandeel and sprat fisheries could benefit many seabird species) while other measures could potentially have negative interactions (for example measures to increase productivity or survival of some large gull populations may have negative consequences for some tern populations). In such cases there may be a need to consider spatially resolved management to reduce undesirable interactions.

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