Scottish Government
Riaghaltas na h-Alba
gov.scot

Finding out the Fate of Displaced Birds

Scottish Marine and Freshwater Science Vol 9 No 8

K R Searle, D C Mobbs, A Butler, R W Furness, M N Trinder and F Daunt

marinescotland

Finding out the Fate of Displaced Birds (FCR/2015/19)

Scottish Marine and Freshwater Science Vol 9 No 08

K R Searle, D C Mobbs, A Butler, R W Furness, M N Trinder and F Daunt

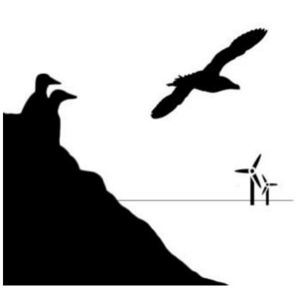
Published by Marine Scotland Science
ISSN: 2043-7722
DOI: 10.7489/12118-1

Marine Scotland is the directorate of the Scottish Gorvernment responsible for the integrated management of Scotland's seas. Marine Scotland Science (formerly Fisheries Research Services) provides expert scientific and technical advice on marine and fisheries issues. Scottish Marine and Freshwater Science is a series of reports that publishes results of research and monitoring carried out by Marine Scotland Science. It also publishes the results of marine and freshwater scientific work that has been carried out for Marine Scotland under external commission. These reports are not subject to formal external peer-review.

This report presents the results of marine and freshwater scientific work carried out for Marine Scotland under external commission.

Finding out the fate of displaced birds (FCR/2015/19)

K. R. Searle, D. C. Mobbs, A. Butler, R.W. Furness, M.N. Trinder & F. Daunt



Title	Fate of Displaced Birds
Client	Marine Science Scotland
Client reference	
CEH reference	NEC05978
CEH contact details	Dr Francis Daunt Bush Estate, Penicuik, Midlothian, EH26 0QB t: 0131 445 4343 f: 0131 445 3943 e: frada@ceh.ac.uk
Date	02/02/2018

Contents

1 Executive Summary

- The Scottish Government has the duty to ensure that the development of the offshore renewable sector is achieved in a sustainable manner. A key challenge in delivering sustainable development is the potential effects of offshore renewable developments (ORDs) on populations of seabirds. Seabirds breed in internationally important numbers in Scotland, and many colonies are designated as Special Protection Areas under the EU Birds Directive [2009/147/EC]. Offshore renewable developments may affect seabirds from collisions with turbine blades, displacement to less favourable habitats, barrier effects to the movement of birds, disturbance during construction and operation, contamination, noise and indirect effects via impact of developments on seabird prey.

- The aim of this project was to produce a tool to estimate the cost to individual seabirds, in terms of changes in adult survival and productivity, of displacement and barrier effects resulting from ORDs. The tool was developed for common guillemot, razorbill, Atlantic puffin, and black-legged kittiwake in the Forth/Tay region during the chick-rearing period. The tool has been constructed as a MATLAB Application ("SeabORD") deployed with 'MATLAB Runtime', which is freely available, enabling users to use the tool without the need for MATLAB. The tool provides a user-friendly interface for setting up simulation runs and user-provided inputs, and for displaying model outputs. A guidance document and worked example are provided with the tool.

- The tool uses a simulation model, which extends and improves that developed by Searle et al. (2014), to predict the time/energy budgets of breeding seabirds during the chick-rearing period, and translates these into projections of adult annual survival and productivity for each individual and at the population level. The model simulates foraging decisions of individual seabirds under the assumption that they are acting in accordance with optimal foraging theory, minimising time away from offspring whilst maximising energy gain. In the model, foraging behaviour of individual seabirds is driven by prey availability, travel costs, provisioning requirements for offspring, and at-sea density of conspecifics. The model estimates productivity and adult survival, the latter resulting from estimates of adult mass at the end of the breeding season. To determine ORD effects, baseline scenarios are compared with scenarios containing one or more ORDs.

- The model estimates the demographic fate of individual birds, partitioned into different categories of affected individuals, including those that experience only displacement, only barrier effects, both or neither (i.e., those that never interact with the ORD). The model also quantifies the impact on observed birds - it looks at the relationship between the number of birds seen in a "snapshot" at-sea survey in the ORD footprint and the mortality associated with the subsequent development from the ORD. This provides a mechanism for translating at-sea survey data from ORD footprints into population-level demographic consequences.

- The model was parameterised from empirical values for time activity budgets, adult mass change during chick-rearing, chick growth and chick survival from studies of these or closely related species from CEH's long term study of seabirds on the Isle of May or from published studies elsewhere. In some instances it has been necessary to set parameter values based on expert opinion because relevant empirical data does not exist. These cases are clearly identified within the report, and the implications on model outputs are discussed.

- The tool requires the user to input a range of information on ORD footprints, displacement and barrier rates, colony locations, colony population size, bird foraging distribution and density, and prey distribution and density. Users also specify the proportion of the total species population to include in simulations, and the number of matched pairs of baseline and ORD simulations, both of which affect assessments of uncertainty in model outputs. Users are advised to identify a range of median prey densities over which to run multiple paired simulations to provide a range of estimated ORD impacts, which are then synthesized into a single value with associated uncertainty for each impact metric.

- Local tracking data represents the 'gold standard' for estimating bird densities for use with the tool. The method in which these data are analysed to derive estimated foraging densities may have an impact on effect sizes. This is particularly the case in terms of whether the statistical analysis has included or removed flight locations from GPS tracking data prior to estimating bird densities, but is also important in terms of whether non-flight fixes have been partitioned into foraging and resting, whose distributions may differ. The report provides guidance on data suitability for input into the model.

- Results from these four study species in the Forth/Tay region were in accordance with theory and past work; effect sizes of ORDs depended on their size and shape, proximity to SPA colonies and the species in question. Furthermore, for any particular ORD scenario, SPA and species, the impacts on displacement and barrier effects on productivity and adult survival varied with assumed prey levels. It is, therefore, recommended that users identify the range of prey levels that constitute moderate conditions, as determined from empirical data on adult body condition at the end of the season, and a series of matched pairs of model runs at different prey levels within that range are undertaken to obtain the estimated range of potential effects under moderate conditions.

- We have developed a method that uses stratified random sampling to estimate effect sizes across the range of ‘moderate’ conditions experienced by birds in the baseline (no ORDs present) and generate an average for each model metric. This method has the advantage that it incorporates uncertainty in model outputs deriving from uncertainty in prey levels, to produce both an overall mean estimate for each metric and a corresponding 95% prediction interval that includes prey uncertainty.

- This is, to our knowledge, the first analytical tool for estimating the population level consequences of displacement and barrier effects for seabirds. It is a user-friendly framework for assessing the impacts of ORDs on breeding seabirds, estimating the fate of individual birds, and translating these effects into policy-targeted metrics at the population level to improve precision of assessment and facilitate the transition of Scotland’s energy sector to a more environmentally sustainable suite of platforms. The tool can be adapted to estimate these effects on any seabird species in any part of the UK during chick-rearing, if empirical data are available. With further development, the model could operate on other periods of the annual cycle, subject to the availability of suitable data.

- There remain important caveats associated with the estimated effects arising from this work. Individual-based simulation models (IBMs) are a useful tool for assessing the cumulative effects of behavioural decisions and energetics in animals, particularly in situations where empirical data are lacking meaning correlative methods may not be used. However, as for all models, the outputs of IBMs are subject to the accuracy of model structure and parameterisation and the inputs used to drive models. Whilst a broad understanding of the behaviour and energetics of foraging and breeding seabirds exists, this understanding is by no means comprehensive, and as in any model, we have had to use the best available evidence and expert judgement to formulate some of the mechanisms and to set values for some of the parameters within the model. Importantly, within the timeframe of this project there has not been the opportunity to perform a full sensitivity analysis for how model parameters, particularly those derived from expert opinion, affect model outputs. Moreover, there are a lack of empirical data on the proportion of birds that are susceptible to displacement and barrier effects, and for those that are, how they alter their behaviour in response to ORDs. Furthermore, there is limited information on whether seabird prey show a behavioural response to ORDs, with subsequent effects on seabirds. There is also, as yet, no empirical data available to assess how affected birds may habituate to ORDs over time, thereby potentially reducing the impacts of developments as birds become accustomed to their presence. Similarly, there has not been a comprehensive analysis of foraging site fidelity within these species, preventing the incorporation of this behaviour within the tool. Finally, bird habitat use varies considerably over space and time, potentially associated with spatio-temporal variation in prey availability. Therefore, the precise interaction of ORDs with the foraging and flight patterns of breeding birds at any particular location or time will inevitably vary from model predictions, and remains a key driver of ORD impacts on SPA populations.

2 Introduction

2.1 Background

Scotland is committed to meeting 100% of its electricity demands from renewable sources by 2020. Marine renewables are a significant contributor to these targets, but the industry must comply with strict legislation (EU Birds Directive [2009/147/EC]) that protects internationally important seabird populations. This is a critical challenge because marine renewable developments have the potential to impact on seabird populations. The two effects that have received the most attention are collisions with turbine blades and displacement from developments (Drewitt & Langston 2006; Larsen & Guillemette 2007; Masden et al. 2010; Grecian et al. 2010, Langton et al. 2011, Scottish Government 2011). Other factors that may be important include barrier effects to the movement of migrating or commuting birds, disturbance during construction and operation, contamination, noise and indirect effects via impact of developments on seabird prey.

This project will focus on the means by which marine developments may affect seabirds from displacement and barrier effects. If the development is located in a foraging area then birds can be displaced, a process that may equate to habitat loss. If the development is on a route used by birds for daily or seasonal commuting then this may create a barrier to movement, potentially forcing affected individuals to travel further to meet their requirements.

This project first undertook a literature review of displacement and barrier effects (see Appendix A). This review demonstrated that there is considerable uncertainty about the potential impacts on seabird populations from these effects. To assess the state of knowledge, displacement can be split into two components for assessment: the proportion of the population that are displaced from the development and the magnitude of the demographic consequences on colony SPAs that arises from displacement and barrier effects.

Estimates of the former have been obtained primarily from at-sea surveys. Studies have shown marked variation in barrier effects and displacement among species. However, they have also shown substantial variation within species among studies, suggesting that effects are highly context dependent. One potential cause of this variation is intrinsic differences among individuals in different populations studied, for example the extent to which they are central place foraging or operating as independent individuals. However, another potential cause is the challenge in obtaining robust quantification of displacement and barrier effects. In particular, these quantifications rely on key assumptions, arguably the most important of which is that the relative differences in density of birds inside and outside the wind farm are due to displacement or barrier effects, when alternative explanations such as the relative distribution of prey are equally plausible.

Such studies based on monitoring surveys are also not structured to address the second component outlined above - the magnitude of demographic consequences. The absence of empirical data has necessitated the use of a precautionary approach to assessments. When based on well-grounded assumptions and reliable parameter estimates, individual-based simulation models can provide a valuable framework for estimating the demographic consequences of a variety of environmental perturbations. As such, this approach can be used to estimate demographic effects for seabirds of offshore renewable developments (ORD) mediated via the costs of barrier effects and displacement, whilst incorporating the uncertainty surrounding these estimates. This project sought to develop an analytical tool that uses this approach to quantify the demographic consequences of displacement and barrier effects, thereby improving the precision of assessment of ORD impact on seabirds so that regulatory bodies and their statutory advisors can reach informed, evidence-based decisions on the potential impacts of renewable developments whilst also encouraging sustainable use of the sea, consistent with the Scottish Government's twin goals of meeting its obligations with respect to climate change mitigation and safeguarding the marine environment.

2.2 Framework

Individual-based simulation models (IBMs) are a useful tool for assessing the cumulative effects of behavioural decisions and energetics in animals, particularly in situations where empirical data is lacking meaning correlative methods may not be used. However, as for all models, the outputs of IBMs are subject to uncertainties in model parameterisation and the inputs used to drive models. Whilst a broad understanding of the behavioural and energetics of foraging and breeding seabirds exists, this understanding is by no means comprehensive, and as in any model, we have had to use the best available evidence and expert judgement to formulate some of the mechanisms and set some of the parameters within the model.

We developed an analytical tool to simulate the individual behaviour, energetics and demography of four species of seabirds (black-legged kittiwake, common guillemot, razorbill and Atlantic puffin) during the chick-rearing period of the breeding season in the context of user-specified ORDs. The tool uses a simulation model, which extends that developed by Searle et al. (2014), to predict the time/energy budgets of breeding seabirds, and translates these into projections of adult annual survival and productivity (i.e., chick survival/mortality). The model represents a considerable improvement from the previous version (Searle et al. 2014) in terms of the sophistication of biological realism in foraging behaviour and bird flight, model processing speed, and the range of output metrics used to assess effects of ORDs on individual and population level demography.

The model simulates foraging decisions of individual seabirds under the assumption that they are acting in accordance with optimal foraging theory. Each individual selects a suitable location for feeding during each foraging trip from the colony based on bird density maps derived from a range of methods, and the subsequent behaviour of birds is then simulated, incorporating realistic assumptions and constraints derived from observed behaviour. Fundamentally, the model assumes that the foraging behaviour of individual seabirds is driven by prey availability, travel costs, provisioning requirements for offspring, and behaviour of conspecifics. The resulting outputs for adult mass at the end of the breeding season are then translated into an estimate of population level adult survival for each colony, with and without one or multiple ORDs present. The model provides individual and population level estimates for the change in adult mortality and breeding success for individual colonies affected by one or more ORDs, providing a direct link from observed or estimated spatial foraging patterns of breeding birds through to population demographics. The model also enables the behaviour and fate of individual birds to be tracked and summarised in a range of different ways. This permits a direct quantification of the demographic consequences of displacement for individual birds.

2.3 Quantification of ORD Effects on Breeding Seabirds

There are a number of useful ways in which these effects may be quantified. The assessment of Searle et al. (2014) was concerned with assessing the demographic consequences of ORDs at the population level. This project, in contrast, is concerned with assessing consequences for individual birds. The obvious simple metric for calculating the “fate of displaced birds” (individual level effects) would be to look at the proportion of birds that have been displaced or barrier affected by the ORD, at any point during the breeding season, which subsequently die over the following winter. However, this more simplistic metric would not distinguish between whether those birds that suffer overwinter mortality as a result of the ORD impacts would in fact still have suffered overwinter mortality even if the ORD had not been built. Therefore, a more defensible metric would be to look at the proportion of “affected” birds (i.e., those that are displaced or barrier affected at any point during the chick-rearing period) that subsequently die when the ORD is present *but not* when it is absent - the “excess” mortality associated with the ORD. This metric forms the basis of this project, and is equal to:

$$\frac{\text{\# of affected birds that die when ORD is present} - \text{\# affected birds that die without ORD}}{\text{Total number of affected birds}}$$

This metric can be applied to either adult or chick mortality. The same metric definition can also be used with alternative, more detailed, definitions of ORD “affected” birds: this could relate solely to whether individual birds are ever displaced, ever barrier affected, or to the frequency with which barrier and displacement effects occur over the entire chick-rearing period. The most detailed form involves calculating this metric separately for all possible combinations of the cumulative number of days on which a bird is displaced and the cumulative number of days on which it is barrier affected over the whole breeding season; this allows us to look in detail at the relationship between the frequency of ORD impacts and mortality, and to distinguish between barrier and displacement effects. This approach also allows us to look at the impact on mortality for birds that never interact with the ORD. This is important because the ORD may impact upon birds indirectly through increased intra-specific competition in the area around the ORD leading to reduced prey intake, and thereby increased mortality, depending on the strength and form of density-dependence within the population (Horswill et al. 2016).

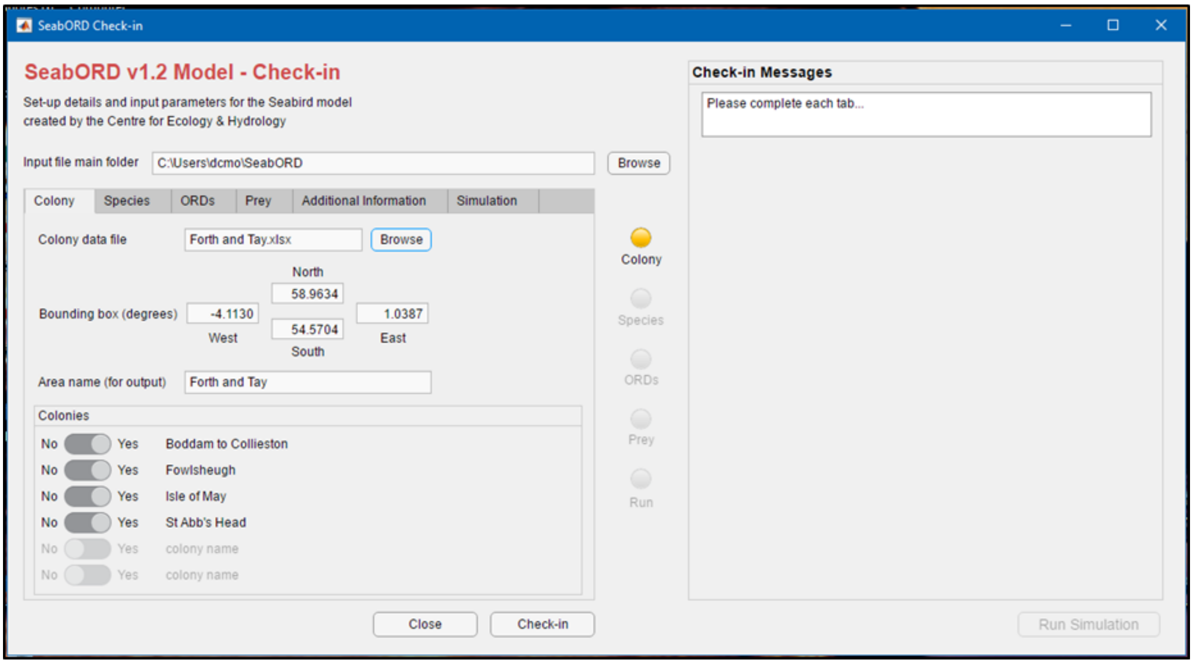
Considering the impact of the ORD upon the set of displaced birds does not, however, relate directly to the decision-making process because it does not provide a direct link to the type of data typically acquired for this purpose, namely at-sea surveys of birds in proposed ORD footprints. Therefore, as well as using the model to quantify the impact on *displaced* birds, we also quantify the impact on *observed* birds - i.e., we use the model to look at the relationship between the number of birds seen in a “snapshot” (pre-consent) survey of the ORD and the mortality associated with the subsequent development of the ORD (using paired simulation runs with and without the ORD). This provides a mechanism for translating spatial survey data from ORD footprints into population-level demographic consequences, and integrates information on the fate of individual birds with information on turnover of individuals using a specific area.

Together, these components create a tool capable of estimating the fate of individual breeding seabirds affected by ORDs for a range of species throughout UK waters. The tool has been constructed as a user-friendly MATLAB Application (“**SeabORD**”) deployed with 'MATLAB Runtime', which is freely available, enabling users to use the tool without the need for MATLAB.

3 Methodology for Developing the Tool for Assessing Impacts of ORDs on Breeding Seabirds

3.1 Overview of Tool

The tool requires the user to input information on ORD footprints (shapefiles), colony locations (WGS84 latitude and longitude), colony population size, bird foraging density and prey density. The last two inputs may, alternatively, be selected from a range of options within the tool ('maps' or 'distance-decay' for bird density; and a 'uniform' prey density). The user then specifies from a range of options to set the scenario parameters for the model run, for instance specifying the number of ORDs to include in the run, the percentage of the population to include, overall prey level, and the probability of displacement and barrier effects. The analytical tool is a Matlab RunTime executable, which can be installed on a computer without needing a Matlab licence. The tool provides a user-friendly interface for setting up simulation runs and user-provided inputs, and for displaying model outputs during and post completing of model runs:



SeabORD v1.2 Model - Check-in

Set-up details and input parameters for the Seabird model
created by the Centre for Ecology & Hydrology

Input file main folder: C:\Users\dcmo\SeabORD

Colony data file: Forth and Tay.xlsx

Bounding box (degrees): North 58.9634; West -4.1130; South 54.5704; East 1.0387

Area name (for output): Forth and Tay

Colonies:
- No / Yes: Boddam to Collieston
- No / Yes: Fowlsheugh
- No / Yes: Isle of May
- No / Yes: St Abb's Head

Check-in Messages: Please complete each tab...

3.2 Key Model Processes and Assumptions:

Here we summarise the key model mechanisms, assumptions and their impact on demographic output for assessing effects of ORDs on breeding seabirds. The model can be condensed into a series of sub-models associated with different stages of simulation (Figure 3-1; e.g., estimating spatial distribution of birds, simulating foraging behaviour and provisioning, estimating survival from mass change of adults).

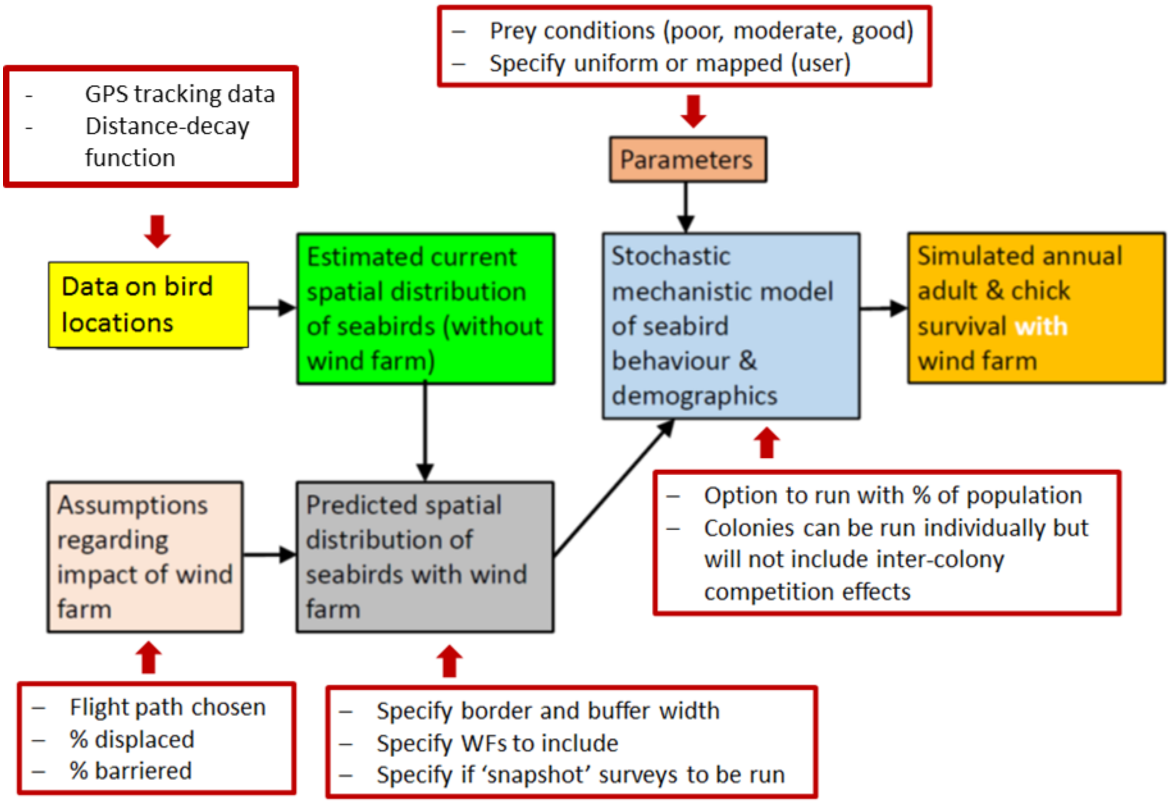


Figure 3-1: Schematic of model structure and data and user-specified inputs.

The specific mechanisms within the model are based upon the best available evidence or expert opinion for how breeding seabirds are likely to behave in terms of time-activity budgets and specific behaviours relating to their own energy acquisition, provisioning of energy for chicks, and breeding behaviours such as attendance at nests. Below we summarise all of the main mechanisms within the model, listing their assumptions and stating the likely impact of each on subsequent model output for demographic parameters (Table **3-1**).

Table 3-1: Summary of mechanisms, assumptions and likely effect in demographic output for all processes operating within the model (SeabORD) for estimating the impact of ORDs on breeding seabirds.

Process	Mechanism	Assumptions	Likely effect on demographic output
Bird Foraging locations:			
Local GPS maps	Birds from each colony choose a foraging location in proportion to the intensity of usage estimated using a GAM model of GPS points. Importantly, there is no assumption that birds attempt to meet an ideal free distribution (IDF). The IDF has restrictive assumptions, including that birds have perfect information of resource supply and distribution of conspecifics, and that there are no constraints to patch choice. The first two assumptions are clearly unrealistic, and the third is likely to be violated when central place foraging occurs, as for breeding seabirds. Furthermore, the balance between competition and facilitation in foraging seabirds is not properly understood. Finally, several studies have shown that seabird and prey distribution seldom conform with IDF predictions (e.g., see Fauchald 2009).	Foraging locations are chosen independently at each simulated time step with no influence of site fidelity. The order of individuals choosing foraging locations is random at each time step so the likelihood of an individual choosing a location with high or low bird density is also random. The available GPS data provide an accurate and unbiased estimate of the underlying spatial distribution of foraging birds utilised by birds from the colonies of interest.	Bird location influences demographic output through determining distance travelled, intraspecific competition and potential encounter with ORDs (displacement and barrier effects). The relationship between the spatial distribution of birds and the impact of the ORD upon survival is potentially very complicated, so it is not straightforward to anticipate the likely sign or magnitude of effects that would arise from altering the spatial distribution.
Distance-decay	Intensity of usage declines exponentially with distance from colony according to pre-specified parameters by the model user. It is the responsibility of the user to best determine that simulated bird distributions match those expected in reality. Birds from each colony choose a foraging location in proportion to the predicted intensity of usage from the distance-decay algorithm. Importantly, there is no assumption that birds attempt to meet an ideal free distribution (IDF). The IDF has restrictive assumptions, including that birds have perfect information of resource supply and distribution of conspecifics, and that there are no constraints to patch choice. The first two assumptions are clearly unrealistic, and the third is likely to be violated when central place foraging occurs, as for breeding seabirds. Furthermore, the balance between competition and facilitation in foraging seabirds is not properly understood. Finally, several studies have shown that seabird and prey distribution seldom conform with IDF predictions (e.g., see Fauchald 2009).	The spatial distribution of birds is unaffected by either environmental heterogeneity or competition. Foraging locations are chosen independently on each simulated time step with no influence of site fidelity. The order of individuals choosing foraging locations is random at each time step so the likelihood of an individual choosing a location with high or low bird density is also random.	Bird location influences demographic output through determining distance travelled,intraspecific competition and potential encounter with ORDs (displacement and barrier effects).

Prey availability			
Median prey density	The user specifies a median prey density across cells in the region of interest which is used to specify the overall available prey density per grid cell in the model.	Adults protect their own survival to safeguard future reproduction (via a threshold in acceptable mass loss in relation to provisioning of chicks and abandoning the breeding attempt) over that of their chick's survival	This trade-off between current reproduction and future survival means that relative effects on adult survival and productivity will depend on prey level in complex ways. As prey levels decline, the effect will initially be stronger on adult survival as they safeguard current reproduction, but with further declines the effect on productivity will strengthen and those on adult survival stabilise as individuals abandon breeding. Yet further declines in prey levels are then likely to affect both demographic rates simultaneously.
Uniform spatial distribution of prey	Prey is uniform throughout the available foraging area	All locations have equal prey availability	Prey availability at each bird location influences demographic output through determining intake rates, and therefore the required time spent foraging to achieve a set energy requirement. Model output under uniform prey is likely to be less variable than that under prey derived from local GPS data because all birds encounter the same prey availability at their chosen foraging location.
Local GPS spatial distribution of prey	Prey is estimated from a GAM model of bird GPS locations assuming that once the accessibility (distance from source colony) and competition (distance from next nearest colony) effects are accounted for, the remaining spatial distribution in the intensity of usage is due to prey availability.	No knowledge of empirical prey distribution and density is assumed, prey is derived solely from bird locations. Locations far from the source colony with high densities of birds assume high prey availability. Foraging locations simulated by SeabORD are determined by bird densities (see above section) and are not related to prey directly (i.e., no assumption of Ideal Free Distribution)	Prey availability at each bird location influences demographic output through determining intake rates, and therefore the required time spent foraging to achieve a set energy requirement. Model output under uniform prey is likely to be less variable than that under prey derived from local GPS data because all birds encounter the same prey availability at their chosen foraging location.
Displacement and barrier effects			
Displacement effects	The user defines a proportion of the total population that are susceptible to displacement effects. Displacement susceptible birds are displaced from the ORD footprint (footprint + border) when their chosen foraging location lies within this region. Upon displacement, birds select a new foraging location within the buffer area around the ORD in proportion to the modelled bird density within the buffer area. It is assumed that birds fly straight to the new foraging location from the colony (i.e., they do not attempt to first fly to the displaced location). As a result birds may either	The user must set the displacement rate for each modelled species. This defines the proportion of the total population that are susceptible to displacement. Individual birds are randomly assigned to the displacement-susceptible category until this proportion is met at the population level. As a result all individuals in the displacement susceptible category will always seek a new foraging location in the ORD buffer zone when their chosen	Displacement effects can be both positive and negative in terms of their impact on demographic output. If a bird is displaced closer to the source colony it will have lower flight costs and shorter flight times, subsequently benefitting from displacement both energetically and in gaining more time for other activities (foraging or time at the nest). If a displaced bird is displaced into a part of the buffer zone where prey availability is higher (after taking into

	incur additional flight costs due to the new location being on the far-side of the ORD (and due to barrier effects if the individual is also barrier-susceptible), or may have reduced flight costs because their new foraging location in located on the near-side of the ORD in relation to the source colony.	foraging location lies within the ORD footprint or border region.	consideration the interference effects of other birds foraging at that location), it will benefit from a higher intake rate allowing it to more rapidly meet its energetic costs, therefore benefitting energetically and in gaining more time to devote to other activities (time at nest). If a displaced bird is displaced to a location further from the source colony it will incur increased flight costs and flight time, negatively affecting its energy budget and reducing the time available for other activities (foraging and time at nest). If a displaced bird is displaced into a part of the buffer zone where prey availability is lower (after taking into consideration the interference effects of other birds foraging at that location), it will suffer reduced intake rate, thereby negatively affecting its energetic budget through increased time spent foraging, potentially affecting its ability to meets its energetic requirements and devote time to attending its nest.
Barrier effects	Because the model assumes barrier-affected birds must also be displacement-susceptible (it is likely not plausible for a bird to not be prepared to fly within an ORD footprint, but to be prepared to forage within an ORD footprint) all barrier-affected birds are also assigned to the displacement-susceptible category. The proportion of the population in the barrier-affected category is set by the user when setting the barrier rate. This proportion may only be as great as the displacement rate proportion set above (because all barrier-affected birds must also be displacement-susceptible). Should a barrier-affected bird chose a foraging location obstructed by the ORD footprint it incurs additional flight costs determined by the barrier flightpath method ('perimeter' or 'A-star').	The user must set the barrier rate for each modelled species. This defines the proportion of the total population that are susceptible to barrier effects. Individual birds are randomly assigned to the barrier-affected category until this proportion is met at the population level. As a result all individuals in the barrier affected category will always fly around the ORD footprint + border zone when their straight-line path to the chosen foraging location is obstructed.	Barrier effects are negative, unless they cause a bird's chick to suffer mortality from unattendance or low provisioning as a result of its partner giving up the breeding attempt when an ORD is present, releasing both adults from restrictive central place foraging conditions resulting in the bird that did not reach the mass loss threshold losing less mass over the course of the breeding season. When a bird is obstructed by the ORD it incurs extra flight costs (energy and time) due to avoiding the ORD footprint + border. This will negatively affect the individual's energy budget and will reduce time available for other activities (foraging and time at nest). Some individuals may choose to reduce the number of foraging trips made per simulated time step to reduce the time costs associated with the extra flight distance. However, due to the optimisation procedure within the foraging component of the model, birds will never benefit from reducing the number of trips in relation to their time-energy budget in the paired baseline run within an ORD present.
Intake rate and number of trips			

Intake rate	Intake rate is determined by prey availability at the bird's foraging location, whereby a Type II functional response returns the estimated intake rate after accounting for effects of interference competition arising from the number of additional birds also foraging at that location during the simulated time step. Prey depletion occurs, determined by the shape of the Type II functional response curve for each species.	The effects of conspecifics foraging at the same location is assumed to create interference competition, reducing the intake rate of each forager in relation to the total number of other birds foraging at that location over the duration of the simulated time step. No facilitation by conspecifics is assumed to occur. Each individual experiences prey depletion whereby their intake rate drops with time spent foraging at a location, determined by the shape of the Type II functional response. Prey depletion occurs during each foraging trip, but prey is then replenished to the original level before any further foraging is undertaken. This means that each time a bird visits the same location within a simulated time step it is assumed that it encounters the same initial prey availability, and therefore experiences the same initial intake rate each time.	Intake rate is strongly and directly related to provisioning of food to chicks, and changes to adult and chick body mass over the chick-rearing period. Higher intake rates (due to greater prey availability or the presence of very few conspecifics) result in birds being able to meet their energy requirements more quickly, thereby increasing the amount of time available for other activities (time at nest).
Number of trips	Birds optimise the number of foraging trips to make during each simulated time step based on the prey availability at their chosen foraging location and the associated flight time accrued travelling between the foraging location and the source colony. If birds are able to meet their required daily energetic requirement at the chosen foraging location they select the number of trips that minimises the total time requirement (foraging + flying) required to meet the DER. If birds are unable to meet their required DER at the chosen foraging location (because prey availability is sufficiently low that the realised intake rate does not allow the bird to reach its DER within the time available) then the bird selects the number of trips that leads to the greatest total prey intake by the bird (i.e., that which minimises their shortfall in intake relative to the DER). Bird state (adult mass) also plays a role in determining the number of trips when the adult's DER cannot be met. If	The mechanisms underlying the selection of the number of trips to make per simulated time step assume that birds attempt to meet their DER within the shortest amount of time, thereby maximising nest attendance. If birds are unable to do meet their DER, the model assumes that they select the number of trips which minimises the energy deficit (i.e. the difference between DER and daily energy intake)	The model selected number of trips affects demographics by determining the time-energy budgets of each adult bird, and therefore its change in mass per simulated time step, and the change in mass of its chick. The 90% adult mass threshold that triggers a shift in behaviour when selecting the optimum number of trips to allow unattendance of chicks has a strong and direct impact on chick survival. Raising this threshold would increase unattendance and subsequent chick mortality, but would also allow adults the opportunity to better protect their own survival by minimising mass loss through additional energy gained by unattending chicks; lowering it would have the reverse effect.

	the adult's mass is >90% of its initial mass at the onset of chick rearing, it will avoid non-attendance of its chick, and will select the number of trips that minimises its energy deficit. However, if an adult's mass is >80% but <90% of its initial mass, it will fail to attend its chick, and will therefore select the number of trips that either allows it to meet its DER within the simulated time step (by increasing foraging time and unattending its chick), or that which minimises its energy deficit (by increasing foraging time and unattending its chick – although still not having sufficient time in the time step to meet its DER).		
Allocation between adult and chick			
Daily Energetic Requirements and provisioning	If an adult successfully collects all the food it needs (its DER plus half its chick's DER) then it provides exactly one half of what the chick needs to the chick during the simulated time step. If an adult is not able to collect enough food to satisfy its own DER plus half of its chick's DER then the intake of both chick and adult will be reduced so that each reach the same percentage of their energy requirements. If a bird is only simulated to receive 50% of the total energy needs for itself and (half of) the chick at a particular time step, for example, then the adult will receive only 50% of its DER, and the chick will receive 50% of one half of their DER (the other parent may still be able to provide 100% of its contribution to the chick's DER, it which case the chick would actually receive 75% of its total DER that time step, but this will not always be the case).	Adults do not take in to account the state (body mass or age) of their chick when deciding how to adjust their time-energy budgets to best meet energetic requirements and successful rearing of their chick. Adults do not account for the provisioning or unattendance of their partner when making decisions regarding time-energy budgets to best meet their own energetic requirements and successful rearing of their chick. Nor do adults take into account the provisioning of their chick by their mate when determining how much food to collect, therefore there is no compensation within a pair where one adult can acquire more food for the chick to compensate for its mate not being able to collect enough food.	The acquisition of DER and subsequent provisioning to chicks directly affects both the mass change of adults and chicks, and therefore their subsequent survival. The lack of compensation between adults in a breeding pair means that any deficit in DER for the chick arising from one parent failing to capture enough food cannot be mitigated by the other parent, should that parent have additional time available for foraging (after all other activities, including attendance at nest). This means the effect of an ORD on the foraging of one parent cannot be compensated for by the other parent, increasing the negative impact of an ORD upon chick survival over a model where such compensation is allowed to occur. It would be possible to change this mechanism within the model so that, for instance, the adult always attempts to provide 100% of one half of the chick's DER before provisioning itself. However, due to the lack of empirical data on which to parameterise this process, the division of acquired energy is simply split equally between the parent and the chick.
Bird states and consequences			
Unattendance and abandonment of breeding attempt	When an adult's body mass is greater than 90% of its starting body mass at the onset of chick-rearing (based on empirical data) it will avoid unattending its chick, even if it had not met its DER during the simulated time step.	Adults do not take in to account the state (body mass or age) of their chick when deciding how to adjust their time-energy budgets to best meet energetic requirements and successful rearing of their chick.	The rules governing unattendance and abandonment of the breeding attempt have a strong and direct impact on chick survival, as well as on energy acquisition by adults and consequently their body condition and survival prospects.

	If an adult's body mass is between 90% and 80% of its initial mass it will favour its own needs over those of its chick, and will leave its chick unattended in order to achieve its required DER. Adults with a body mass of less than 80% of their starting mass abandon the breeding attempt. This necessarily means that their partner also gives up the breeding attempt, resulting in chick death. Finally, if an adult's body mass falls below that deemed critical for survival (60% of its initial body mass at the onset of chick-rearing), the adult is assumed to have died and is removed from the simulation. This causes its partner to abandon the breeding attempt for the remainder of the simulation.	Adults do not account for the provisioning or unattendance of their partner when making decisions regarding time-energy budgets to best meet their own energetic requirements and successful rearing of their chick. The model assumes adults will prioritise their own survival (by protecting their energy gain and minimising mass loss over the chick-rearing period) over that of their chick. However, the model also assumes that adults avoid unattendance when their mass is still reasonably high (>90%) in comparison to their starting mass at the onset of chick-rearing.	The effect of unattendance is to increase the risk of chick death through exposure or predation. The risk of chick death increases linearly with time unattended, until reaching a certainty after 18 consecutive hours of unattendance. Both the thresholds for mortality from unattendance (18 hours) and abandonment of the breeding attempt (adult body mass <80% of initial mass at onset of chick-rearing) has a strong impact on demographic output from the model. Raising the unattendance threshold (e.g., from 18 hours to 24 hours) would decrease overall chick mortality, and would lower the impact of an ORD on chick mortality as fewer chicks would die from a result of unattendance. A similar effect would be seen on model output if the adult mass threshold for abandonment (<80% of initial mass) were lowered.
Chick death	Chick death occurs when the chick's mass reaches 60% of that of an idealised chick provided with its total DER on each time step of the simulation up to the current point in time.	The model assumes that chick's DER do not change with age or body mass.	Varying the mass threshold (60% of idealised chick's body mass) at which mortality occurs has a direct and strong impact on chick survival of the population. A lower threshold would reduce the impact of an ORD on chick survival because chicks would be able to buffer a greater reduction in provisioning (and therefore mass loss) before dying. Chick death is also linked to adult body condition and survival prospects because of behavioural changes that occur to adult foraging when freed from provisioning for offspring.
Mass Change			
Adults	Adult birds update their body mass at the end of every simulated time step in response to the balance between the energy expended and gained during the time step. When the bird's DER is met, it loses no mass. When the bird's DER is not met, it loses body mass according to a linear relationship with the ratio of the energy deficit to the energy density of the bird's tissue (parameter value set a priori). The upper limit to adult mass loss is set by this ratio between the energy deficit and the energy density of tissue.	The model assumes adults may only remain at the same weight as they enter the chick-rearing period, or lose mass over the chick-rearing period – it is assumed to be impossible for them to gain weight	Adult mass loss over the chick-rearing period determines both its own subsequent survival, as well as affecting its behavioural decisions affecting the survival of its chick through provisioning and unattendance.

Chicks	The model assumes a simple linear function for daily mass change of chicks in relation to food provisioned by its parents. If a chick receives its total DER from its parents its mass changes by the maximum possible mass gain (g/day; parameter value set a priori). If a chick receives only a proportion of its total DER, its increase in mass declines linearly with the decrease in total DER provided by its parents. The model contains a threshold parameter (set a priori) that represents the proportion of the chick's DER provided at which zero growth occurs.	The model assumes chicks may not lose mass during the model simulation, however if insufficient energy is provided by its parents it will fail to gain mass at the rate required to maintain good health, and eventually die from starvation.	Mass change in chicks is strongly and directly related to provisioning of food to chicks, and is the ultimate determinant of chick survival over the chick-rearing period and, therefore, the chick survival of its parents. Chick death is also linked to adult body condition and survival prospects because of behavioural changes that occur to adult foraging when freed from provisioning for offspring.
Mass-Survival relationship			
Converting adult mass at end of chick-rearing into subsequent survival	For each individual adult bird the model assumes a logistic relationship between the adult mass at the end of the breeding season and the probability of over-winter survival. The logistic model contains two unknown parameter values: in the way we have parameterized the models these parameters quantify (a) the "baseline" survival and (b) the slope associated with the impact of a change in adult mass upon the change in logit(survival probability).	The model assumes: a) that the shape of the relationship between adult-mass and over-winter survival can be described by a logistic curve; b) that the baseline survival probability has been specified correctly; and c) that the mass-survival slope parameter has been specified correctly. The value of the baseline survival probability is fixed to be the mean value across sites with observed data on annual adult survival.	The impact of ORDs upon adult survival will be directly related to the value of the slope parameter – the two quantifies are related in a strong but nonlinear way. The value of the baseline survival probability is also likely to be moderately strongly linked to ORD effects; the nonlinearity of the logistic curve means the impacts of the slope parameter vary depending on the level of baseline survival. The estimates of ORD effects are not likely to be strongly related to the assumption that the curve has a logistic shape.

3.2.1 Summary of Positive and Negative Impacts and Underlying Mechanisms:

Here we summarise the mechanisms by which there can be a positive or negative effect on adult mortality as a result of ORDs within the model (Table 3-2).

Table 3-2: Summary of positive and negative impacts on adult survival and their underlying mechanisms.

	Potential negative impact	**Potential positive impact**
Displacement effect	- Increased flight cost if displaced location is further away from colony - Increased competition in buffer zone around ORD thereby lowering intake rate of non-displaced birds in this zone - Displacement to a new location with lower prey availability and greater competition effects - Reduced foraging time - Potential for unattendance - Potential for increased energy requirement (DER) during following time step	- Decreased flight cost and flight time if displaced location is closer to colony - Displacement to a new location with higher prey availability and lower competition effects - Reduced competition for birds (not displacement-susceptible) remaining to forage within the ORD - Effect of ORD on a bird’s partner causes partner to abandon breeding attempt earlier in the season therefore the other parent loses less mass over the course of the chick-rearing period
Barrier effect	- Increased flight cost - Increased flight time - Reduced foraging time, leading to reduced energy intake at this time step - Potential for unattendance - Increased energy requirement (DER) during following time step – e.g. because of a reduction in the time spent undertaking low-energy activities (such as a resting at sea)	- Effect of ORD on a bird’s partner causes partner to abandon breeding attempt earlier in the season therefore the other parent loses less mass over the course of the chick-rearing period

3.3 Foraging Locations of Birds, Flights and Available Prey

The first stage involves specifying the foraging distribution of individual breeding seabirds from the colony of interest over the course of the breeding season, and the spatial footprint of the ORD. The foraging distribution that can be used as an input to the analytical tool will depend on the amount of GPS data available for the species and colony in question. Here, we consider a suite of methods classed within two standard alternatives:

1. If reasonable amounts of GPS data are available for this species at the focal colony then those data can be modelled (e.g., using a GAM, similar to the methods used in Searle et al. (2014);

2. If GPS data are unavailable or very limited for a species, then the foraging distribution can be specified more simply by assuming a simple relationship between the distance from the focal colony and the foraging density (e.g., that the density decays exponentially as distance increases, see Section 3.10).

In this report, we present a complete set of outputs and results for black-legged kittiwakes in the Forth-Tay region of SE Scotland using both various methods for Option 1 (local GPS tracking data) and Option 2 (no tracking data, simple exponential distance decay model), and a set of results for Option 1 for guillemots, razorbills and puffins.

3.4 Simulation Model for Daily Individual Time-Energy Budgets

3.4.1 Intake Rate

In this model, the intake rate achieved by a bird at its foraging location is described by the Michaelis-Menten equation for a Type II functional response (Holling 1959). The functional response describes the relationship between the density of prey and instantaneous intake rate for an individual. The form of the functional response for piscivorous seabirds is not known. Expert opinion generally expects seabirds to follow a Type II curve, whereby intake rate increases with prey density until reaching a maximum determined by handling time of the predator, although some studies of diving piscivores have suggested a Type III curve (Middlemas et al., 2006; Enstipp et al., 2007). The Type III functional response resembles the Type II in having an upper limit to prey consumption, but differs in that the response of predators to prey is depressed at low prey density, with the expectation that intake rate then rapidly accelerates (more than linearly as in the Type II response) due to predators reducing their learning time, prey switching, or a combination of both phenomena. However, to minimize the number of unknown parameters, for which little empirical data are available for calibration, we selected the widely used Type II form, which assumes that intake rate increases asymptotically with the density of prey at a foraging location and has a stronger theoretical underpinning:

$$\text{Intake rate at time } t = IR\_MAX * \text{ Prey at time } t / (IR\_HALF + \text{ Prey at time } t)$$

where *IR_MAX* is a parameter denoting the maximum possible intake rate (estimated from empirical data, see Appendix B) and *IR_HALF* is a parameter denoting the prey level that is associated with the intake rate reaching half of the maximum possible value. The value of *IR_HALF* was determined as part of the model calibration process and is specific to each species. The implications of using the Type II curve over the Type III are that at lower prey densities the Type II curve would tend to estimate higher intake rates than the Type III curve, but that this would be reversed at higher prey densities, until both curves reach the same asymptote or maximum intake rate at very high prey densities.

We use this relationship to simulate the decline in intake rate over time spent foraging at a location due to prey depletion, and to determine the amount of time an individual requires at a location to attain a certain cumulative intake of prey. This form of the functional response implies that the prey quantity remaining at the foraging location at time *t* is equal to:

$$x(t, x_0) = \{x: (x - x_0 + IR\_MAXt + IR\_HALF\log(x) - IR\_HALF\ \log(x_0)) = 0\} \qquad \textbf{[Eq 1.]}$$

where x_0 denotes the prey quantity at time 0.

This in turn implies that the total prey consumed by foraging up to time *t* is equal to:

$$y(t, x_0) = x_0 - x(t, x_0) \qquad [\textbf{Eq. 2}]$$

and this formula is used to calculate the total daily prey intake for a bird, given a) the number of foraging trips that the bird undertakes and b) the length of each trip. Note that “time” is assumed to return to zero at the start of each new foraging trip - we assume prey depletion at a location within a foraging bout or trip, but when the bird returns to a location for a subsequent foraging bout or trip the amount of prey in that location is reset to the original level. The solution to Equation 2 cannot be written down analytically, but it can be calculated numerically using a non-linear solver. Using numerical methods we pre-calculate the total prey consumed for a grid of times *t* and initial prey quantities x_0 at a foraging location.

3.4.2 Number of Trips

We select the number of trips per time step (for each species the chick-rearing period is divided into biologically relevant time steps - 24 hours for all species in this report except black-legged kittiwakes where the time step was 36 hours) that a bird undertakes by considering the possible outcomes that occur for each possible number of trips from one to six. An upper limit of six trips per time step was selected because the vast majority of empirical data on these species suggest that most individuals complete between two and four foraging trips per time step. Specifically, for each potential number of trips, r, we divide the daily energy requirements (DER; of both chick and adult) by r, and then numerically invert Equation 2 to determine the amount of foraging time per trip required to achieve this energy intake. We then calculate the total time required for all trips to be

$$\text{Total time required for all trips on a day} = \left(\frac{\text{Foraging time required to acheive DER}_r}{r} + \text{flying time required for each trip}\right) * r$$

where the 'Foraging time required to achieve DER_r' is dependent upon the number of trips (*r*) because of the link between foraging time at a location and decline in intake rate whilst foraging. We do this for all possible values of r. We then select the number of trips in one of two ways depending upon whether the bird is able to acquire its DER within the time available for foraging:

1. If the total time requirement is less than the total time available, for at least one possible value of r, then we select the number of trips to be the value that minimises the total time requirement required to achieve the DER;

2. If the total time requirement exceeds the total time available for all possible values of r then it is impossible for the bird to achieve their DER on this day. In this situation we select the value of r that leads to the greatest total prey intake by the bird (i.e., which minimises their shortfall in intake relative to the DER).

3.4.3 Intra-Specific Competition

We assume that intra-specific competition between individuals foraging at the same location acts to reduce the intake rate multiplicatively, so that:

$$\text{Intake rate with competition} = \text{Intake rate without competition}/\text{Intra-specific competition effect}$$

More specifically, the intra-specific competition effect is assumed to be a power-law model of the form (Hassell & Varley 1969):

$$\text{Intra-specific competition effect} = (\text{Total number of birds within the grid cell, summed across all colonies})^m$$

in which the unknown parameter m controls the magnitude of the intra-specific competition effect. The value of this parameter must, in terms of the biology, lie between zero and one: a value of zero corresponds to the special case in which there is no competition (i.e., the intake rate for each bird is unaffected by the number of other birds present), whilst a value of one corresponds to the special case in which competition is linearly related to the number of birds present.

Within the model we apply the intra-specific competition function to the inverse of the *IR_HALF* parameter in the functional response (which controls the shape of the curve relating intake rate to prey density). We assume that:

$$\text{IR\_HALF with competition} = \text{IR\_HALF without competition} * \text{Intra-specific competition effect}$$

The final specification of the intra-specific competition effect is detailed in Appendix C; our parameterisation improves the biological interpretability of the competition parameter and reduces the technical complications involved in model calibration.

3.4.4 Chick Mass Growth Rate

Chick growth in relation to food provisioning has not been well estimated under field conditions. Therefore, we assume a simple linear function for daily mass change of chicks in relation to food provisioned by its parents. One of the parameters for this function (‘*P*’, below) is derived from an energetics study on the growth and physiology of kittiwake chicks (Gabrielsen et al. 1992). Given the lack of empirical data on the other study species, we used the value of *P* obtained from kittiwakes, and it is challenging to judge the implications of this decision. The second parameter (‘*G*’, below) is estimated from observations of chick mass change from hatching to fledging for each species (CEH unpublished data; Harris & Wanless 2011). More specifically, we assume that

$$\text{Mass change} = G * (((\mathit{intake} / \text{DER}) - P) / (1 - P))$$

Where ‘*intake’* is the actual amount of food provided to the chick, the parameter “*G*” represents the maximum possible mass gain (g) per day if the chick receives 100% of its DER, and the parameter “*P*” represents the proportion of the daily energy requirement (DER) for the chick that corresponds to zero mass change: i.e., to neither an increase nor a decrease in mass.

3.4.5 Adult Mass Change

In the model all adult birds update their body mass at the end of each day based on the energy they gained and expended in foraging and other activities. We used the published equation from Langton et al. (2014) to calculate the body mass of all adults at the end of each time step:

$$Mass_{t+1} = Mass_t + \frac{Energy\_gained_t - DER_t}{K_G}$$

where $Mass_{t+1}$ is the body mass at the start of the next time step, $Mass_t$ is the body mass in the current time step, $Energy\_gained_t$ is the energy the individual acquired during the current time step, DER_t is the daily energy requirement for the adult for the current time step, and K_G is the energy density of the bird's tissue (kJ/g). Published values for the energy density of bird's tissue are available for guillemots (Gabrielson 1996) and gannets (Montevecchi et al. 1984), both of which are close to 38 kJ/g; therefore for all species in the model we use this value.

3.5 Behaviour and Bird State and Energetic Costs

3.5.1 Cost Model

We developed a cost model to accrue the amount of time and energy birds expended in reaching and foraging within their chosen location. This model is an expanded version of that used in Daunt & Wanless (2008) and Wanless et al. (1997) and separates the flight cost and foraging cost for each seabird to derive total energy expenditure.

3.5.2 Activity Costs

Foraging cost for each bird is defined by the energetic costs of foraging and the amount of time an individual is required to spend foraging to meet both its own DER and 50% of the DER of its offspring. On the first time step of the simulation, adult Daily Energy Expenditure (DEE) was drawn from a normal distribution parameterised using the mean and standard deviation of adult DEE from empirical data on the study species (Daunt et al. 2008 and references therein). On all subsequent days adult DEE was set to match the energy expended by each bird in the previous time step. Chick DEE remained constant throughout the simulation. We chose not to model increases in chick DEE with growth in order to constrain model processing time to reasonable limits, but species-specific mean daily energy requirement of chicks was based on provisioning rates recorded at colonies for each species, from a sample of chicks of a range of ages, so we do not think this simplification in the model would have had a large bearing on the results. This calculation implies both parents share the costs of provisioning equally. The resulting required daily energy expenditure (DEE) was divided by an assimilation efficiency (0.78, Hilton et al. 2000) to obtain the total DER of the birds.

Empirical daily time budgets of birds during chick-rearing demonstrate that adults divide their activities into four categories of behaviour - foraging, flight, time spent at the colony, and time spent resting on the sea surface (Daunt et al. 2002). For each bird, the foraging model returns the simulated flight time for each bird spent travelling to its chosen foraging location, and the simulated foraging time required to meet its required DER. The remaining time during each model time period is split into time spent at the colony and time spent resting at sea. A minimum of one hour spent resting at sea was required for each bird (Daunt et al. 2002), and each bird attempted to spend half of each time step at the colony thereby preventing the chick being left unattended at the nest. Any remaining time was split evenly between time at the colony and time resting at sea. If a bird could not meet its DER in the time available without leaving its nest unattended, a set of decision rules were implemented based on the energy state of the adult (see Behavioural Modes, below).

We derived the flight cost incurred by each bird by calculating the time taken to travel the distance both to and from the chosen foraging location assuming a mean flight speed for each species (Pennycuick 1997), upscaled to match the chosen number of trips per time step.

We then multiplied the time spent carrying out each of these activities by species- and activity-specific energy costs available from the literature (i.e. cost of flight, foraging, resting at and time at colony). In addition, we incorporated the energy cost of warming food to derive the total DER for each bird (Gremillet et al. 2003). These DER were converted into grams per day assuming a mean energy density of 6.1 kJg-1 (Harris et al. 2008).

3.5.3 Behavioural Modes for Adults and Chicks and Subsequent Decisions

At the end of each time step each adult was assigned to a behavioural mode that determined its behaviour in relation to chick rearing in the following time step. Behavioural modes for adults were determined by a critical mass threshold below which the adult is assumed to defend its own survival above that of its chick, based on expert judgement because of the lack of empirical data on these thresholds.

Therefore, when an adult’s body mass was greater than 90% of its starting body mass at the onset of chick-rearing it would avoid leaving its chick unattended, even if it had not met its DER. However, if its body mass was between 90% and 80% of its initial mass it would favour itself, and leave its chick unattended in order to attempt to achieve its required DER. There are no precise empirical data available to set these thresholds, therefore, in the model the thresholds were set based on similar logic to that used by Langton et al. (2014). Using guillemot as an example, the 90% threshold represents an average mass of around 820 g, which is well above the starvation mass (Golovkin 1963) and about 2/3 of the difference in mean initial mass (920g) and the minimum mass recorded in UK ringing data (770g; Robinson 2005), the majority of which probably came from birds at breeding colonies (Langton et al. 2014). The 80% threshold corresponds to around 735 g in guillemots, which is below the minimum mass recorded in UK ringing data for this species, and, therefore, likely to represent a reasonable mass at which individuals give up the breeding attempt to minimise further mass loss and safeguard their own survival. Therefore, for all species, we set behavioural models to switch using the 90% and 80% thresholds of the individuals starting mass.

Adults with a body mass of less than 80% of their starting mass abandon the breeding attempt. This necessarily means that their partner also gives up the breeding attempt, resulting in chick death. Golovkin (1963) found that unfed guillemot adults had lost approximately 60% of their body mass at death, therefore, should an adult's body mass fall below that deemed critical for survival (60% of the average pre-breeding season adult body mass for each species; Golovkin 1963), the adult is assumed to have died and is removed from the simulation. This causes its partner to abandon the breeding attempt for the remainder of the simulation. A similar assumption is made for chicks, which are assumed to have died if their body mass falls below 60% of that for a hypothetical chick that has received its DER on each model time step up to the current time.

If the time a chick's parents spend attending the nest falls below a critical threshold the chick is assumed to die through exposure. This parameter could not be set using empirical data ('*unattendance_hrs*'), therefore, we set it to be 18 hours for all species (except Atlantic puffins) based on expert judgement. If a chick suffers mortality its parents switch to 'nest abandonment' mode. We also incorporated an increased risk of predation if a chick was left unattended by both parents for an amount of time less than that which would result in its death through exposure. This was modelled as a probability of death that increased linearly with time left unattended, up until the time threshold was reached at which point the chick was assumed to have died from exposure or predation ('*unattendance_hrs*').

For burrow-nesting puffins, once the chick reached a certain energy deficit (80% of the body mass of a chick that have been provisioned with all its requirements at every previous time step, using the same threshold employed for adults abandoning young, given the absence of empirical data) it was assumed the chick ventured to the entrance of the burrow and suffered a linearly increasing predation risk with its body mass deficit as a consequence (between 60% and 80%). Above the threshold body mass value of 80% there was no risk to the chick from being left unattended by parents. Below the lower threshold of 60% the chick was assumed to have died.

3.6 Calibration of Model Parameters

Where available we set values for all parameters in the foraging model from published literature, expert opinion or unpublished CEH data from the long-term study on the Isle of May. The set of parameters that were selected in this way, and the values assigned to them, are listed in Appendix B with sources given in Appendix D.

There remained two parameters whose values could not readily be assigned in this way: the parameter '*IR_HALF_a*' (which relates to the way in which intake rate decreases with prey depletion, and the effect of conspecifics on intake rate of individuals through assumed interference competition ('*IR_HALF_b*'). The values for these parameters were chosen so that the model output matched empirical data on the mean number of foraging trips made per day, and the mean and range of time spent foraging per day for each species. The values of these parameters were calibrated before being used within SeabORD (Appendix B).

3.7 ORD Effects on Individuals within Simulation Model

Two main behavioural responses to ORDs are simulated in the model: displacement and barrier effects.

At the start of each simulation run, individuals were assigned as either birds that would be displaced if their foraging location fell within the ORD footprint ('displacement-susceptible birds'), and/or as birds that would choose to fly around the ORD footprint ('barrier-susceptible birds') if their chosen foraging location lay on the far side of a wind farm. These values were fixed for the lifetime of each bird meaning that no habituation to wind farms occurred. The proportion of birds that were assigned by the user to be displacement-susceptible and barrier-susceptible depended upon the species and scenario.

The assigned percentages for displacement and barrier levels are set as follows. If, for example, the displacement rate is set at 60% and the barrier rate is set at 20%, 60% of birds are first assigned as displacement-susceptible, and then of these 60%, 20% are assigned as barrier-susceptible. This assumes that birds may be in three categories: only displacement-susceptible (birds will fly through the ORD but not forage within it), both displacement and barrier susceptible (birds avoid ORDs completely), or not displacement nor barrier susceptible (birds are unaffected by ORDs). We did not consider the fourth possible category (birds would forage within the ORD but not fly through it) as biologically plausible, so no birds were assigned to that category. In practice, assessments typically propose a single percentage for “displacement”, intended to capture both displacement and barrier effects. So if, for example, the rate for displacement is set at 60%, 60% of birds are assigned as displacement-susceptible, and of these 100% are assigned as barrier-susceptible. However, the model allows for different percentages to be set should this be required in future.

Displacement and barrier effects were determined using a set of zones created around the footprint of each wind farm (Figure 3-2).

In the model the user specifies the width of an exclusion ‘border’ zone to be added to the ORD footprint supplied by the user as a shapefile. This border zone represents the area around an ORD footprint into which displacement or barrier susceptible birds will not enter due to assumed disturbance effects. If a displacement-susceptible bird chooses a foraging location within the ORD footprint plus the exclusion ‘border’ area (yellow areas, Figure 3-2) then it instead chooses a new foraging location within a 5 km buffer zone of the ORD footprint plus the selected border zone. This distance was chosen by the steering group of the previous project (Searle et al. 2014) based on expert judgement, and carried forward into this project. The precise foraging location in the 5 km buffer is selected with a probability proportional to the prey availability in the buffer zone. Displacement-susceptible birds only incur an additional travel cost if their new foraging location in the buffer zone lies on the far side of the ORD from their source colony (Figure 3-2, orange areas). Should their new foraging location lie on the nearside of the ORD in relation to the source colony, it is assumed that no additional travel cost is incurred as a result of displacement and that the bird flies directly to the new location using the shortest route. We did not impose an additional travel cost for these birds because we assume that over the course of the breeding season birds will have determined the location of the ORD and, therefore, fly directly to the location displaced to.

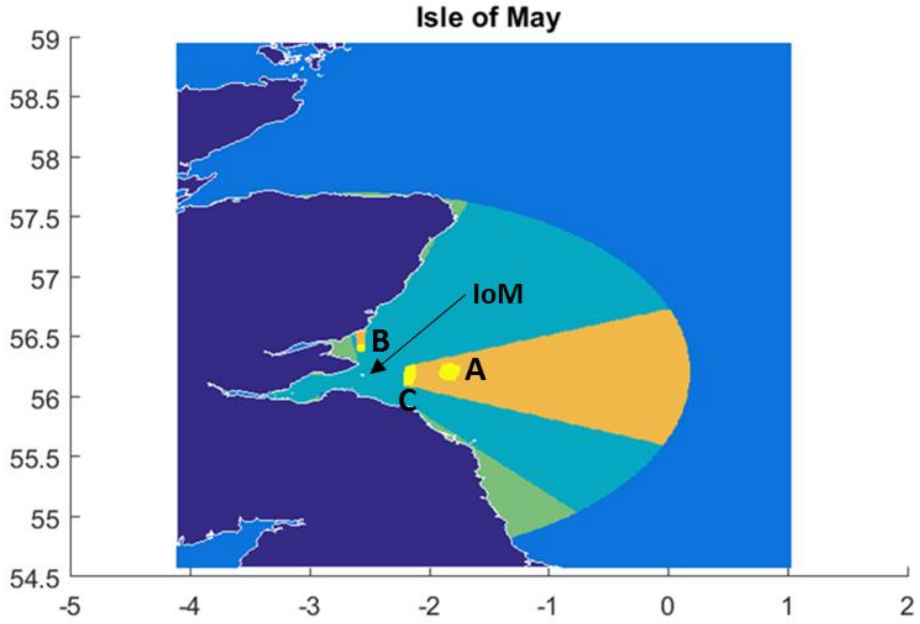


Figure 3-2: Example zones determining effects of three hypothetical ORDs (yellow polygons, labelled A, B, C) on birds flying and foraging from the Isle of May (IoM) in the Forth-Tay. Orange zones are foraging locations that induce barrier effects for birds choosing to forage within the zone from the Isle of May. Light blue circular zone defines the foraging extent for the species. The green zones indicate foraging locations where birds have to navigate around the coastline to reach their chosen destination

If the displaced bird is both displacement and barrier-susceptible, it incurs additional travel costs as a result of barrier effects if its new chosen foraging location lies on the far side of the ORD. The analytical tool has been parameterised to allow two options for how barrier affected birds navigate around ORD footprints. In the first option ('A* pathfinding' routine), a barrier affected bird takes the shortest route to the new foraging location, whilst navigating around the edge of the ORD footprint (Figure 3-3, *Flights 1 & 2b*). In the second option ('perimeter' routine), the barrier affected bird flies in a direct line to its foraging location right up to the front edge of the ORD footprint. It then skirts the edge of the ORD footprint until its path lines up with the straight line path from the source colony to the foraging location, where it resumes a straight line flight to its final destination (Figure 3-3, *Flight 3*).

The tool is parameterised with two options for determining the flight paths of barrier affected birds for several reasons. Primarily this is because there is no empirical evidence for how central-place foraging barrier affected breeding adult seabirds navigate around ORDs. We believe the two most biologically plausible navigation methods are captured by the two routines in the analytical tool. The 'A* pathfinding' routine finds the shortest route to the foraging location, and corresponds to the assumption that birds know the location of the ORD and adjust their flight path to minimise travel costs. This implies that birds have spatial memory for the position of the ORD as well as spatial memory for a pre-determined foraging location selected before leaving the colony. In contrast, the ‘perimeter’ routine assumes that whilst birds have spatial memory for a pre-determined foraging location before leaving the colony (i.e., they attempt to fly to the location in a straight line), they do not necessarily have perfect spatial memory for the location of the ORD. Here, the assumption is that birds attempt to fly in a straight line to their chosen location and only alter course when they encounter the edge of the ORD, then following the edge of the ORD until resuming their straight line course. This flight pattern could arise due to poor spatial memory for ORDs, or due to a preference for following edges of fixed objects to aid navigation.

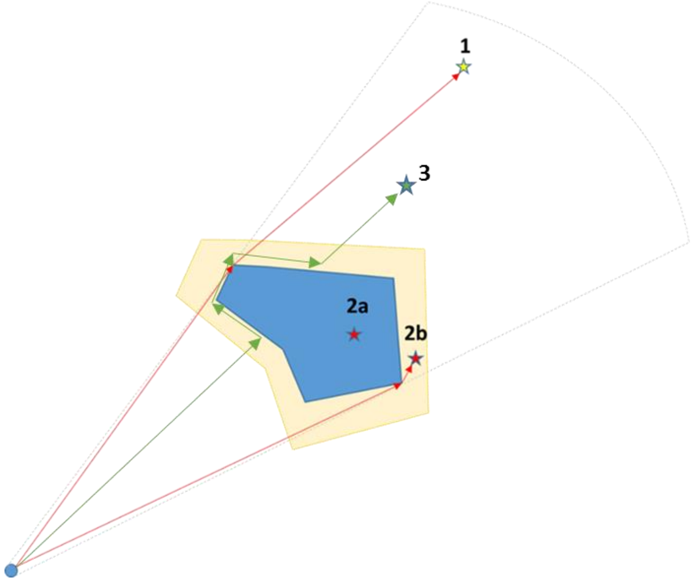


Figure 3-3: Examples of three flights from a source colony (blue dot) in relation to an ORD (blue polygon). Flight ‘1’ represents a barrier-susceptible bird that selects a foraging location on the far side of the ORD (1, yellow star) and incurs an additional travel cost navigating around the ORD footprint using the ‘A* pathfinding’ routine. Flight ‘2’ represents a displacement and barrier-susceptible bird that first selects a foraging location within the ORD footprint (red star, 2a) and then selects a new foraging location in the 5 km buffer (red star 2b, yellow zone). In this case its flightpath to the new foraging location (2b) follows the shortest route whilst avoiding entering the ORD footprint area, again using the ‘A* pathfinding’ option. Flight ‘3’ represents a barrier-susceptible bird that uses the ‘edge following’ routine to determine its route from the colony to the foraging location on the far side of the ORD (green star). All birds use the same flightpath to reach, and return from, their chosen foraging location

All birds that must navigate around a coastline use the A* Pathfinding routine to identify the shortest route around land.

3.8 Estimating Productivity and Adult Survival

There are three key outputs from each run of the foraging model relating to chick and adult survival:

1. the status of each chick (alive / dead) at the end of the breeding season;
2. the status of each adult (alive / dead) at the end of the breeding season;
3. the mass of each living adult (in grams) at the end of the breeding season.

The first two of these quantify the chick and adult survival rates during the breeding season. The final quantity provides an indirect way of quantifying the adult survival rate during the subsequent winter period. We used published relationships between adult mass and annual survival rates in order to convert simulated adult mass values into estimated survival rates. This is done in the same way for baseline simulations and for simulations that have been generated in the presence of ORDs such that we may assess the impact of the ORD upon the adult survival rate by contrasting the paired model runs.

The procedure for converting individual adult mass values into an overall estimate of adult survival for each simulation run is summarised here. Our approach is essentially based previously published studies (Oro and Furness 2002, Erikstad et al. 2009) where the general assumption is that mass and survival are linked through the equation:

$$\log\left(\frac{p_{ij}}{1-p_{ij}}\right)=\log\left(\frac{s_o}{1-s_o}\right)+bm_{ij}$$

where m_{ij} denotes the standardized mass of individual *i* in run *j* and p_{ij} denotes the (annual winter) survival probability of this individual. The value of *b* quantifies the strength of the relationship between mass and survival, and the value of s_0 denotes the ‘baseline’ survival (i.e,. the survival rate that would be associated with a bird of average mass in the absence of an ORD). The overall survival rate for a simulation run, P_i is simply assumed to be the average (mean) of the survival probabilities for all of the individuals within it, so that

$$P_i=\frac{1}{n}\sum_{i=1}^{n}p_{ij}$$

(where *n* denotes the total number of individuals).

The validity of this approach will depend primarily upon the validity of the values that are selected for *b* and s_0. It is worth noting that the approach also makes one substantive assumption - that the relationship between mass and survival is linear, on a logit-transformed scale - but it would be impossible in practice to check the validity of this assumption using currently available information.

The value of the baseline survival, s_0, is assumed to vary between species - the specific values are based upon the results of the population modelling performed by CEH for Marine Scotland (Freeman et al. 2014), and are given in Table 3-3 (below).

The strength of the relationship between mass and survival, *b*, is determined using values given in the published literature. For kittiwakes the value of *b* is based on the value given in Oro & Furness (2002), and for all other species it is based on the value given in Erikstad *et al*. (2009), published values do not exist for razorbill or guillemot, so we assume that they have the same value as that estimated for puffin in the Erikstad *et al*. (2009) paper. The kittiwake study was undertaken on a population in Shetland experiencing low food abundance, and the puffin study was based on a population in northern Norway. Both populations may have differed in terms of adult body mass and relationships between condition and survival from populations in the Forth/Tay region. Furthermore, mass/survival relationships in guillemots and razorbills may differ from puffins. The actual estimated values for *b* are 1.03 (Erikstad *et al*., 2009) and 0.038 (Oro & Furness 2002), but it is important to note that these values cannot be directly compared because they relate to mass values that are expressed on direct scales: for kittiwakes the mass is standardised solely by deducting the mean mass under the baseline scenario (because the paper by Oro & Furness 2002 expresses *b* in grams), whereas for other species the standardisation also involves dividing by the standard deviation under the baseline scenario (because Erikstad *et al*., 2009, expresses mass as a unit-free quantity).

Table 3-3: Baseline survival probabilities for birds under baseline conditions (no ORDs present) with poor, moderate and good prey availability (Freeman et al. 2014). The level of prey availability is determined by the percent mass loss of adult birds over the chick-rearing season (see Worked Example for more details).

	Poor	**Moderate**	**Good**
Kittiwake	0.65	0.80	0.90
Puffin	0.85	0.90	0.95
Guillemot	0.82	0.92	0.94
Razorbill	0.80	0.90	0.95

3.9 Metrics for quantifying the fate of affected birds

Within the analytical tool we produce a range of metrics to quantify the effect of the ORD upon the mortality of birds. These metrics are of two basic types: population-level metrics and individual-level metrics.

3.9.1 Population-Level Metrics

Two population-level metrics are produced.

Metric **P1** calculates the population-level impact of the ORD:

(mortality with ORD present - mortality in baseline) / (population size)

More specifically:

$$P1 = 100 * \frac{\begin{matrix}(\text{Total number of birds simulated to die when the ORD is present} - \\ \text{Total number of birds simulated to die when the ORD is absent})\end{matrix}}{\text{Total population size}}$$

This metric represents the overall impact of the ORD. This is the additional mortality that occurs as a result of the wind farm. Importantly, whilst this metric is identical to the one used in the Searle et al. (2014) report, it is presented here as the percentage change to mortality, not survival (as was used in Searle et al. 2014). Therefore, a positive value for this and all other metrics represents an increase in the mortality of birds (a decrease in survival), and a negative value represents a decrease in the mortality of birds (an increase in survival). These changes have been made to reflect the fact that assessments are primarily concerned with additional mortality effects on breeding birds.

The main population level metric (P1) can be calculated for either adults or chicks (with the formulae being identical in all cases: “birds” is simply replaced with either “adult birds” or “chicks”), whereas the other metrics (see below; P2, I1-I6) are only calculated for adult birds.

Metric **P2** divides the impact of the ORD upon mortality by the number of birds simulated to be within the footprint during a “snapshot” (pre-construction) survey of the population.

We assume that the “snapshot” surveys count all birds that are present within the ORD footprint at a specific instant in time, prior to ORD construction - the snapshot surveys are not equivalent to the kinds of spatial surveys that are actually undertaken, but those surveys can be used to approximate the “snapshot” by (a) adjusting the raw counts for non-detection (a range of statistical methods for doing this are already used within seabird ecology) and (b) apportioning birds to source colonies**.** The idea of the “snapshot” survey was used previously in Searle et al. (2015), and is discussed further there.

The model can be used to simulate the number of birds that forage within an ORD footprint (o_k) or that fly over an ORD footprint (l_k) during a particular snapshot, s. Within the tool the user can specify the days (based on the simulation time step number) on which snapshot surveys are conducted. The timing of the survey within the day is then chosen at random because the SeabORD model does not allocate specific times within a day (or time step) to the activities of individual birds; the allocation of specific times to activities would require more detailed knowledge of the temporal breakdown of time-activity budgets within a day than is currently available for these species. The algorithm for simulating the behaviour associated with the snapshot is presented in Appendix E.

3.9.2 Individual Level Metrics

Metrics **I1 - I6** use the same calculation as P1, but apply this to subsets of the population. These subsets are determined based on the frequency with which individuals interact with the ORD, and the nature of these interactions (barrier or displacement effects), as follows:

birds that never interact with the ORD at any point during the breeding season, via either displacement or barrier effects (I1);

i. birds that ever (at least once) interact with the ORD at any point during the breeding season, via either displacement or barrier effects (I2);
ii. birds that are displaced by the ORD at least once during the breeding season, but are never barrier affected (I3);

iii. birds that are barrier affected by the ORD at least once during the breeding season, but are never displaced (I4);

iv. birds that are displaced and barrier affected by the ORD at least once during the breeding season (I5);

v. birds that are displaced on exactly d time steps, and barrier affected on exactly b time steps (I6).

Note that these categories do not equate to the categories of individuals described in Section 2.6, which consider susceptibility to interaction, not actualised level of interaction. For example, an individual in subset iii may not have experienced barrier effects, even if it is susceptible to barrier effects should the wind farm be on route to their foraging destination. The last metric, I6, focuses on birds that experience specific patterns of barrier and displacement effects - e.g. that were displaced on five time steps and experience barrier effects on four time steps - and so provides the most detailed breakdown of the effects into population sub-groups. These are not provided in this report for the simulations described below, but are available from CEH upon request.

Metric P1 is of direct policy relevance, and formed the key output from the assessment within Searle et al. (2014). This metric combines information on the frequency and demographic consequences of displacement/barrier effects within a simple metric. Metrics I1-I6, in contrast, separate out the demographic consequences of displacement/barrier effects from the frequency with which these effects occur.

3.10 Distance-Decay Model

Within the model we allow users to specify a set of options for the bird and prey density to accommodate cases where there is no local GPS data. In this case, users may select a uniform prey density (prey density is uniform across the entire region) and estimate a bird density based on a simple distance-decay function.

3.10.1 Model for Distance-Decay

This approach assumes that the proportion of time that birds from the colony of interest spend within the i-th grid cell (the utilisation distribution) is u_i, where:

$$u_i \propto \frac{\exp(-\beta d_i)}{d_i}$$

if d_i represents the distance (in a straight line along a great circle) from the colony of interest to the midpoint of this grid cell. The values of u_i are normalised so that they sum to one across all grid cells. The parameter β determines the rate of distance decay.

This model is motivated as follows:

1. it assumes that the *total* number of birds within each distance band decays exponentially with distance - i.e., it assumes that the total number of birds that lie at distance d from the colony is proportional to $\exp(-\beta d)$;
2. it then assumes that birds within this distance band are distributed uniformly; because the distance band has a circumference equal to $2\pi d$ this leads the density in space to be divided by d (this is equivalent to dividing by $2\pi d$, because the normalisation step renders the constant, $2\pi,$ irrelevant).

The exponential decay model used in step (1) is not the only possible model for decay with distance, but it is a very widely used model and has the advantage of only containing a single unknown parameter. The decay involved in the second step ($1/d$) follows directly from geometric considerations: there are fewer grid cells within distance bands close to the colony than within distance bands far from the colony, so if birds are equally likely to visit each distance band the density of birds would decay with distance at a rate $1/d$.

If the parameter β were taken to be equal to zero then the model would assume that birds are uniformly distributed across distance to colony, so that the density decays with distance solely due to geometric considerations. If the parameter β is taken to be large then the model assumes a very rapid decay of density with distance: far more rapid than would be explained by geometry alone.

3.10.2 Specification of the parameters for distance-decay

In practice, the distance-decay model will typically be used for species for which relevant GPS data are unavailable. In such situations it will usually be necessary to specify the rate of distance decay based on biological judgement. The decay rate parameter, β, has no easy biological interpretation, and so would be a difficult quantity for an expert to meaningfully provide. We therefore re-parameterize the model so that it can be expressed in terms of quantities that could more easily be provided by a user. Specifically, we assume that the decay parameter is related to two other input parameters, r and p, via the formula:

$$\beta = -\log(1-p)/r$$

Substituting this into the equation for u_i leads to the equation:

$$u_i \propto \frac{(1-p)^{d_i/r}}{d_i}$$

Note that this is the same distance-decay model as that introduced above, even though the formula looks rather different: we have just represented the model in terms of different input parameters.

3.10.3 Specifying the Input Parameters

In terms of their biological interpretation, parameter r represents the range within which the proportion p of all foraging occurs. So if $r = 120$ and $p = 0.95$, for example, then we are saying that 95% of foraging locations lie within 120km of the colony.

In biological terms, the best strategy will probably be to specify r to be the foraging range of the species, because an estimate of this is already available for most species (Thaxter et al. 2015). The value of p still needs to be specified based on expert judgement; it should be plausible to assume that the published foraging ranges correspond to a large value of p, but whether the specific value is $p = 0.9$, $p = 0.99$, or $p = 0.999$, for example, remains a matter for expert judgement. Defensible choices for p are likely to vary between species, depending on the quantity and representativeness of data that were used to inform the published foraging ranges.

3.10.4 Derivation of the Re-Parameterisation

Assume that the distribution of the total number of birds with distance is exponential: i.e., it is proportional to $\exp(-\beta d)$. The PDF (i.e., normalized density) corresponding to this is $f(d) = \beta\exp(-\beta d)$, and the corresponding CDF is equal to $F(d) = 1 - \exp(-\beta d)$.

The values r and d are equal, by definition, to the values such that $F(r) = p$. This implies that $p = 1 - \exp(-\beta r)$, which in turn implies, by simple algebra, that $\beta = -\log(1-p)/r$.

3.11 Accounting for Uncertainty

It is important to quantify the uncertainty associated with the model outputs. We achieve this by running the model multiple times, and using the variation between runs to provide a measure of the level of uncertainty. We use the multiple runs to account for two key sources of uncertainty:

1. inherent stochastic variability between model runs; and
2. the uncertainty associated with the total amount of prey.

We account for the latter by selecting a different prey level for each model run. This approach has the advantage that it incorporates uncertainty in model outputs deriving from uncertainty in prey levels - which is likely to be a key source of uncertainty in practice - in order to produce both an overall mean estimate for each metric and a corresponding 95% prediction interval that includes prey uncertainty.

The steps involved in generating multiple runs from within SeabORD are detailed in the accompanying ‘Worked Example’.

3.11.1 Selecting Prey Levels

The model is run a number of times, $r = 1, \dots, R$, for a number of different simulated populations, to capture the stochastic variability inherent within the mechanisms that are described by the model, and the uncertainty associated with the level of prey. The prey levels for each run are selected by:

1. Identifying the median regional prey levels corresponding to the upper (U) and lower (L) boundaries of ‘moderate’ conditions, based on adult mass change and chick/nest survival in baseline runs for each species and colony of interest (Table 3-4);
2. Simulating the prey level for the r-th model randomly from within the range.

$$\left(L + \frac{(r-1)(U-L)}{R}, L + \frac{r(U-L)}{R},\right)$$

The random part of the process (Step 2) is a simple form of stratified random sampling; this provides a more accurate quantification of uncertainty, for a particular number of model runs R, than simply simulating R values randomly from within the range (L, U).

Table 3-4: Conditions used to set upper and lower boundaries for the median regional prey value corresponding to 'moderate' conditions for each species.

	Percent adult mass loss		Chick/nest survival
	Upper bound	**Lower bound**	**Lower bound**
Black-legged kittiwake	5	15	11
Common guillemot	3.5	10.5	49
Razorbill	3.5	10.5	50
Atlantic Puffin	3.5	10.5	50

Upper and lower boundaries for 'moderate' regional prey conditions are set by comparing model output in baseline runs (no ORDs present) for the change in adult mass (percent) and the nest or chick mortality (percent). Using empirical data (Harris 1979; Harris & Wanless 1988; Gaston & Hipfner 2006; Nelson 2013; Newell et al. 2016), we identified percent adult mass loss over the course of the chick-rearing period and nest survival rates (for kittiwakes) or chick survival rates (for auks) that reflect those observed during 'moderate' environmental conditions. Because adult mass loss (%) is the most reliable model output (less variable than chick survival) and of primary interest to population trends in long-lived species such as seabirds, because of its influence on over-wintering survival probability of adults (Oro & Furness 2002; Erikstad et al. 2009), we base the upper bound for moderate conditions solely using changes in adult mass loss. The lower bound is set using both adult mass loss and chick/nest survival (Table 3-4). If, however, chick or nest survival should fall below the 5th percentile observed in empirical data before the adult mass loss lower bound is reached, we consider that corresponding regional prey value to represent the lower bound of moderate conditions for that species. For instance, in black-legged kittiwakes, if at the end of a baseline run, adult mass loss was 8% but nest survival was 10%, this would be clasified as 'poor' conditions.

3.11.2 Calculating Prediction Intervals

Outputs are generated for each model run; for any particular output – e.g. the change in adult mortality that results from including the ORD. Outputs are generated for all of the metrics of interest (P1 – P6). For each metric we then calculate:

1. the mean of this value across runs, m (to provide our "best estimate" for this quantity); and
2. the standard deviation across runs, s, to capture the uncertainty associated with natural stochastic variation.

In order to present the uncertainty in a format that is of practical use, we calculate the 95% prediction interval associated with using these R simulated populations to predict the output that we would have obtained for the true but unobserved “real” population.

We assume that the outputs from the model runs follow a normal distribution; by standard formulae the prediction interval is then equal to

$$(m - ws, m + ws)$$

where $T_{R-!}$ represents the 97.5% quantile of t-distribution with $R-1$ degrees of freedom and

$$w = T_{R-!}\sqrt{1 + \frac{1}{R}}$$

3.11.3 Interpreting intervals

The intervals represent the uncertainty that arises from trying to predict what will occur within a finite population in a system that is subject to inherent stochastic variability, together with the uncertainty associated with determining the overall level of prey. The latter tends, in practice, to be a much larger source of uncertainty than the former. It is crucial to note that the intervals do *not* account for any other sources of uncertainty: e.g., for the uncertainty associated with estimating model parameters, for the uncertainty associated with the underlying structure of the model, or the uncertainty associated with the spatial distribution of birds. Since a number of these other sources of uncertainty – particularly the uncertainty in the adult mass-survival relationship – are likely to be large, the prediction intervals that we present should be treated with caution, and regarded as *lower* bounds on the actual level of uncertainty.

3.11.4 Selecting the Number and Size of the Simulations

The width of the prediction intervals is equal to $2ws$, and will be determined primarily by three things:

1. the size of the population of interest;
2. the range of plausible prey values $(U - L)$;
3. the size of each simulated population;
4. the number of simulated populations, R.

The first two of these are effectively fixed, but the latter two are features of the model simulations, rather than reality, and so need to be specified by the user.

If the simulated populations are specified to have the same size as the actual population of interest (e.g., the colony size), and the number of simulated populations R is taken to be very large, then w is approximately equal to 1.96 and the width of the prediction interval will be approximately equal to $1.96 * 2s = 3.92s$.

In practice, it will often only be computationally feasible to run a fairly small number of simulations, R, and it may be necessary to run the model for simulated populations that are smaller than the population of interest. Both of these will tend to increase the width of the prediction interval: the former will systematically increase the value of w and the later will tend to systematically increase the value of s. Both will also reduce the reliability of the estimates values of m and s.

Uncertainty can be reduced by increasing the size of each simulated population (up until it reaches the size of the population of interest: it should never be specified to be larger than that size), or by increasing the number of simulated populations. However, because there is a component of uncertainty that reflects the level of intrinsic natural stochastic variability associated with this population size, neither of these strategies can ever reduce the uncertainty to zero.

What is an appropriate value to use for the number of simulations, R? In Figure 3-4 we plot the relationship between R and w; it can we seen that the intervals become very wide when R is very small (especially for $R < 5$) but begin to stabilise towards 1.96 as the value of R increases. For small values of R, however, the value of s (the standard deviation across runs) is likely to be poorly estimated, so even if the value of w is reasonable the associated prediction interval may still have poor performance.

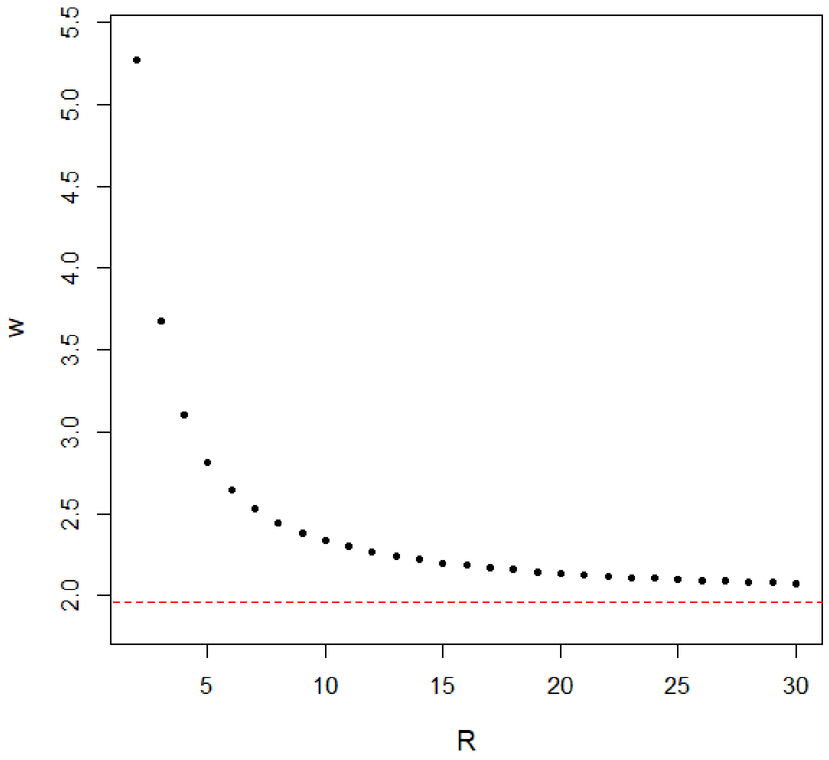


Figure 3-4: The relationship between R and w; it can be seen that the intervals become very wide when R is very small (especially for R<5) but begin to stabilise towards 1.96 as the value of R increases.

4 Results

We use the Forth-Tay as a case study to demonstrate use of the tool because of the high quality and quantity of GPS tracking data, and because the most comprehensive data on seabird demography in the UK are available from CEH's long term study on the Isle of May National Nature Reserve, located in the region. This region was the study area for our previous and ongoing projects on the effects of displacement on seabirds (McDonald et al. 2012; Searle et al. 2014), at-sea turnover (Searle et al. 2015), and demographic consequences of wind farms on seabird populations (Freeman et al. 2014; Jitlal et al. 2017). It is also of immediate policy relevance and high ecological importance, with important populations of species that are potentially vulnerable to the effects of wind farms that have recently been consented in the region. In this region we assessed the effect of three fictional ORD wind farms of varying size and distance from breeding colonies (Figure 3-2).

In the main body of the report we present results for black-legged kittiwakes (please see the accompanying User Guide and Worked Example' for a full illustration of the recommended use of the tool and interpretation of model output), for which in all simulations, the following user-specified parameters were used:

- black-legged kittiwake;
- Combinations of mapped bird density and mapped prey based on local GPS data *or* distance-decay bird density and uniform prey (see Section 4.1);
- Barrier type: perimeter;
- Probability of displacement: 0.4;
- Probability of barrier effect: 1.0 (all displacement-susceptible birds are also subject to barrier effects);
- ORD footprint border: 0.5 km;
- 50% of the total population;
- All SPA colonies in Forth-Tay region (Buchan Ness, Fowlsheugh, Forth Islands, St Abbs Head).

Outputs of all models are sensitive to the accuracy and precision of their inputs. In particular, the distribution of birds and their prey that form key inputs to the model will have large ramifications on the effect sizes of displacement and barrier effects. Where GPS tracking data are available, the fixes obtained comprise all activities at sea, including flight, foraging and resting on the sea surface (see Section 4.5.2 for further details). In the Discussion, we provide guidance on the different options of data input available to users, based on different methodologies for processing GPS data to assign fixes to different behaviours. Here, we illustrate how SeabORD's outputs vary in response to bird and prey densities generated using a subset of these different methodologies (Table 4-1):

i. Using older local GPS tracking data with flight removed (using a speed threshold) and a shorter foraging range to derive bird density and prey maps (based on the Searle et al. 2014 model inputs).
ii. Using those derived from an assumption of no available GPS tracking data (uniform prey and bird density derived by density-decay methods).
iii. Using newer local GPS tracking data including all locations (flight not removed) with a larger foraging range to derive bird density and prey maps.
iv. Using newer local GPS tracking data with flight removed (using a speed threshold) and a larger foraging range to derive bird density and prey maps.

Note that comparisons between methods iii and iv are only presented here for Forth Islands (all other colonies had very small estimates for effect sizes).

Table 4-1: Methods used to generate the four sets of bird density and prey maps used in the scenarios presented for black-legged kittiwakes in this report.

Method	Assumption	GPS tracking data	Foraging Range	Flight Behaviour	Colonies included and % population run
i. Older local GPS tracking data with flight removed	Local GPS tracking data is available capable of distinguishing flight behaviour	Forth Islands: 2010, 2012 St Abbs Head: 2011, 2012 Fowlsheugh: 2011, 2012 Buchan Ness: 2012	170km	Speed threshold used to remove flight locations (>14km/hr)	Forth Islands, At Abbs Head, Fowlsheugh & Buchan Ness (50%)
ii. Distance-decay and uniform prey	No local GPS tracking data is available	None	170km	NA	Forth Islands, At Abbs Head, Fowlsheugh & Buchan Ness (50%)
iii. Newer local GPS tracking data with flight not removed	Local GPS tracking data is available but flight cannot be distinguished (e.g., Wakefield et al. 2017 utilisation distributions)	Forth Islands: 2010, 2012, 2013, 2014 St Abbs Head: 2011, 2012 Fowlsheugh: 2011, 2012 Buchan Ness: 2012	300km	NA	Forth Islands, At Abbs Head, Fowlsheugh, Buchan Ness, Angus, Farne Islands (40%)
iv. Newer local GPS tracking data with flight removed	Local GPS tracking data is available capable of distinguishing flight behaviour	Forth Islands: 2010, 2012, 2013, 2014 St Abbs Head: 2011, 2012 Fowlsheugh: 2011, 2012 Buchan Ness: 2012	300km	Speed threshold used to remove flight locations (>10km/hr)	Forth Islands, At Abbs Head, Fowlsheugh, Buchan Ness, Angus, Farne Islands (40%)

These four methods allow the following useful comparisons to be made, although note that the magnitude and direction of the change in model outputs is entirely specific to the model region, species and fictional ORDs used within the assessments:

- Local GPS data versus distance-decay (Methods i and ii).
- Local GPS data with and without flight removed (Methods iii and iv).

For each of these four scenarios we ran ten matched pairs (baseline run + ORD run) to first provide a more extensive demonstration of effect sizes of different fictional ORDs on black-legged kittiwakes, averaging across sets of ten paired runs spread across a range of ‘moderate’ values for median prey density (user-specified input, where ‘moderate’ is determined by assessments of changes to adult body mass and chick mortality in baseline simulations with no ORDs present; Table 3-4). This serves to illustrate how ORD effect sizes vary in relation to specified regional prey levels.

We then demonstrate how to use SeabORD to generate a single estimate for each metric assessing ORD effects, by identifying the boundaries of the ‘moderate’ prey range and using stratified random sampling to complete sets of paired runs across the moderate prey range which are then averaged to produce a single estimate, plus prediction interval, for each metric.

These runs took approximately 24 hours to run a complete set of ten matched pairs (baseline + ORDs) for one prey level on a standard PC with 8GB of RAM. If the PC used has multiple processors then sets of simulations over multiple prey levels can be run at the same time, so in this instance it took 24 hours to complete all the required runs over the six sets of regional median prey densities. Users should note that there is some lead-in time to setting off these simulations, both in terms of identifying regional prey densities and processing time involved in calculating model zones and some flight paths (see Worked Example for details).

The results for other species (common guillemot, razorbill, Atlantic puffin) are shown in Appendix F.

4.1 Population-Level Metrics

The recommended use of the tool involves first establishing the boundaries for the range of median regional prey densities (user-specified input) over which ‘moderate’ effects on adult mass loss and chick mortality over the chick-rearing period are estimated, in the absence of any ORDs. This is because the estimated impact of an ORD is dependent upon the prey availability, mediated through behavioural mechanisms within the foraging model. When an adult's energy intake falls, triggering their mass to fall below a pre-determined threshold (see methods), the bird will switch behaviours so as to protect its own survival (by minimising energy deficits and, therefore, subsequent mass loss) over that of its chick (which may be left unattended and suffer mortality as a consequence). The balance between these behaviours determines the estimated mass loss in the adult and the survival of their chick, and as such has a direct impact upon the estimated demographic consequences of ORDs.

For each of the scenarios presented here (Table 4-1), we identified between five and six median regional prey densities that covered the range of moderate mass loss in adults and chick mortality (Table 4-2 to Table 4-5). Our primary focus is on birds from the Forth Islands because one regional prey value will often result in 'moderate' conditions at one colony, and 'good' or 'poor' conditions at another.

We identified five values for the median regional prey density using older GPS tracking data mapped bird densities and estimated prey maps (Method i; Table 4-2; 150-175) over which two colonies mostly experienced 'moderate' conditions (Forth Islands and St Abbs Head), and two colonies experienced 'good' conditions (Buchan Ness and Fowlsheugh).

Table 4-2: For bird densities and prey availability derived from older local GPS tracking data with flight removed, the percent mass loss (%) of adult birds during the chick-rearing period in baseline runs (no ORDs present) and their corresponding classifications into 'good' (G, dark green cells), 'moderate' (M, light green cells) and 'poor' (P) prey conditions. Colonies are: 1. Buchan Ness, 2. Fowlsheugh, 3. Forth Islands and 4. St Abbs Head.

	150		**155**		**160**		**165**		**170**		**175**	
Colony	Type	%	Type	%	Type	%	Type	%	Type	%	Type	%
1	G	0.24	G	0.18	G	0.14	G	0.11	G	0.08	G	0.06
2	G	2.64	G	1.90	G	1.50	G	1.21	G	1.00	G	0.81
3	M	10.04	M	9.92	M	9.40	M	7.66	M	5.57	G	3.90
4	M	9.98	M	9.72	M	8.25	M	6.03	G	4.41	G	3.22

Table 4-3: For bird densities derived from the density-decay method and an assumption of uniform prey, the percent mass loss (%) of adult birds during the chick-rearing period in baseline runs (no ORDs present) and their corresponding classifications into 'good' (G, dark green cells), 'moderate' (M, light green cells) and 'poor' (P) prey conditions. Colonies are: 1. Buchan Ness, 2. Fowlsheugh, 3. Forth Islands and 4. St Abbs Head.

	125		**130**		**135**		**140**		**145**		**150**	
Colony	Type	%	Type	%	Type	%	Type	%	Type	%	Type	%
1	M	10.3	M	10.2	M	10.2	M	10.1	M	10.0	M	9.8
2	M	10.4	M	10.3	M	10.2	M	10.1	M	10.0	M	9.7
3	M	10.1	M	9.8	M	9.3	M	8.6	M	7.7	M	6.7
4	M	10.2	M	10.1	M	10.0	M	9.8	M	9.3	M	8.7

Table 4-4: For bird densities and prey availability derived from newer local GPS tracking data without flight removed, the percent mass loss (%) of adult birds during the chick-rearing period in baseline runs (no ORDs present) and their corresponding classifications into 'good' (G, dark green cells), 'moderate' (M, light green cells) and 'poor' (P) prey conditions. Colonies are: 1. Buchan Ness, 2. Fowlsheugh, 3. Forth Islands and 4. St Abbs Head.

	105		**145**		**155**		**160**		**165**	
Colony	Type	%	Type	%	Type	%	Type	%	Type	%
1	M	10.2	G	1.8	G	1.1	G	0.9	G	0.7
2	M	10.5	M	8.4	M	5.6	G	4.4	G	3.4
3	M	10.5	M	9.3	M	6.8	M	5.4	G	4.1
4	M	10.5	M	6.8	G	3.7	G	2.8	G	2.2

Table 4-5: For bird densities and prey availability derived from newer local GPS tracking data with flight removed, the percent mass loss (%) of adult birds during the chick-rearing period in baseline runs (no ORDs present) and their corresponding classifications into 'good' (G, dark green cells), 'moderate' (M, light green cells) and 'poor' (P) prey conditions. Colonies are: 1. Buchan Ness, 2. Fowlsheugh, 3. Forth Islands and 4. St Abbs Head.

	89		**93**		**97**		**103**		**106**	
Colony	Type	%	Type	%	Type	%	Type	%	Type	%
1	G	2.1	G	1.4	G	1.0	G	0.7	G	0.6
2	M	9.5	M	8.5	M	7.1	G	4.8	G	3.9
3	M	9.9	M	9.4	M	8.4	M	6.1	G	5.0
4	M	9.4	M	8.0	M	6.1	G	3.6	G	2.8

For each example, we first ran sets of ten paired matching simulations (baseline + ORD) for each median prey density identified (Table 4-2 and Table ***4-3***) to illustrate a range of effect sizes for each of the metrics quantifying the impact of ORDs on seabird populations across 'moderate' and 'good' conditions for each SPA. These sets of simulations for black-legged kittiwakes are presented below for the five scenarios:

1. Mapped bird density and mapped prey availability and WFA only (older GPS tracking data with flight removed and shorter foraging range; Searle et al. 2014).
2. Mapped bird density and mapped prey availability and WFA + WFB + WFC in combination (older GPS tracking data with flight removed; Searle et al. 2014).
3. Distance-decay estimated birds density and uniform prey availability and WFA + WFB + WFC in combination.
4. Mapped bird density and mapped prey availability and WFA + WFB + WFC in combination (newer GPS tracking data without flight removed and larger foraging range).
5. Mapped bird density and mapped prey availability and WFA + WFB + WFC in combination (newer GPS tracking data with flight removed and larger foraging range).

4.1.1 Overall Population-Level Effect (P1)

We start with the population-level impacts (P1) of each ORD, defined as the percentage point additional mortality due to the ORD (or set of ORDs), derived by comparing ten pairs of baseline-ORD runs and determining the mean effect across the ten paired simulations (Table 4-6,Table 4-7, Table 4-8 and Table 4-9). Results are presented as the mean population level impact (percent additional mortality) with associated 95% prediction intervals around the mean effect (the estimated mean effect would lie within this interval in 95 out of 100 simulated populations). Mean impact (additional percent mortality) on chicks is also presented, with associated prediction interval of the mean effect across runs.

For mapped inputs for bird and prey density, there was no detectable estimated effect of any of the ORDs on birds at Buchan Ness (Table 4-6). No birds were found to interact with any of the ORDs either through displacement or barrier effects. Similarly, there were very few impacts of any ORDs on birds from Fowlsheugh (Table 4-7), and any estimated impacts on adult or chick mortality were within the bounds of quantified uncertainty in the model output (95% prediction interval included zero), meaning that there was uncertainty in whether the ORDs increased the mortality of adults and chicks at all from this colony.

However, for birds at both the Forth Islands (Table 4-8) and St Abbs Head (Table 4-9) there were detectable effects of ORDs on both adult and chick mortality over the simulated range of moderate prey availability conditions.

The estimated mean impact of WFA on adult mortality for birds from Forth Islands ranged from -0.01% to 0.32%, with 95% prediction intervals spanning from -0.18% to 0.68% (Table 4-8). The mean impact on chick mortality ranged from 0.02% to 1.85%, with 95% prediction intervals spanning -0.09% to 2.57% (Table 4-8). The largest impact on adult mortality occurred when median prey density was 170 (mean increase in mortality: 0.32%; 95% prediction interval: 0.11% - 0.52%); and the largest impact on chick mortality occurred when median prey density was 160 (mean increase in chick mortality 1.85%; 95% prediction interval: 1.14% to 2.57%). Impacts on chick mortality ranged from mean effects of 0.02% to 1.85% with 95% prediction intervals spanning -0.09% to 2.57% over the range of median prey densities used (Table 4-8).

The estimated mean impact of WFA on adult mortality for birds from St Abbs Head ranged from -0.05% to 0.35%, with 95% prediction intervals spanning from -0.22% to 0.62% (Table 4-9). The mean impact on chick mortality ranged from 0.01% to 2.07%, with 95% prediction intervals spanning -0.01% to 3.06% (Table 4-9). The largest impact on adult mortality occurred when median prey density was 165 (mean increase in mortality: 0.35%; 95% prediction interval: 0.08% to 0.62%); and the largest impact on chick mortality occurred when median prey density was 155 (mean increase in chick mortality 2.07%; 95% prediction interval: 1.08% to 3.06%). Impacts on chick mortality ranged from mean effects of 0.02% to 2.56% with 95% prediction intervals spanning -0.04% to 3.30% over the range of median prey densities used (Table 4-9).

When all three ORDs were considered in combination (WFA + WFB + WFC) birds from Buchan Ness and Fowlsheugh remained unaffected (Table 4-6 and Table 4-7). However, the estimated impacts on adult and chick mortality increased for birds from the Forth Islands and St Abbs Head (Table 4-8 and Table 4-9).

The estimated mean impact of WFA + WFB + WFC on adult mortality for birds from Forth Islands ranged from -0.01% to 2.11%, with 95% prediction intervals spanning from -0.36% to 2.88% (Table 4-8). The mean impact on chick mortality ranged from 0.46% to 11.38%, with 95% prediction intervals spanning 0.03% to 12.80% (Table 4-8). The largest impact on adult mortality occurred when median prey density was 170 (mean increase in mortality: 2.11%; 95% prediction interval: 1.34% to 2.88%); and the largest impact on chick mortality occurred when median prey density was 160 (mean increase in chick mortality 11.38%; 95% prediction interval: 9.96% to 12.80%). Impacts on chick mortality ranged from mean effects of 0.46% to 11.4% with 95% prediction intervals spanning 0.03% to 12.8% over the range of median prey densities used (Table 4-8).

The estimated mean impact of WFA + WFB + WFC on adult mortality for birds from St Abbs Head ranged from 0.07% to 0.47%, with 95% prediction intervals spanning from -0.27% to 0.78% (Table 4-9). The mean impact on chick mortality ranged from 0.02% to 2.56%, with 95% prediction intervals spanning -0.04% to 3.30% (Table 4-9). The largest impact on adult mortality occurred when median prey density was 165 (mean increase in mortality: 0.47%; 95% prediction interval: 0.17% to 0.78%); and the largest impact on chick mortality occurred when median prey density was 155 (mean increase in chick mortality 2.56%; 95% prediction interval: 1.83% to 3.30%). Impacts on chick mortality ranged from mean effects of 0.02% to 2.56% with 95% prediction intervals spanning -0.04% to 3.3% over the range of median prey densities used (Table 4-9).

Table 4-6: Population-level impact (P1) of each proposed wind farm upon black-legged kittiwake adult and chick mortality rates for all birds in the simulated population (50% of total population) at all SPA colonies in the Forth Tay region. All runs performed with 50% of the population using ten matched pairs of runs, and assume a 0.5 km border around each footprint. The impact is defined as the percent additional mortality due to the wind farm (percentage points; e.g., 0.1 is 0.1% additional mortality; and 1.5 is 1.5% additional mortality). A positive value implies an increase in additional mortality when the WF is present. Upper table 'WFA (Mapped)' shows results for bird and prey densities derived from local GPS data when only one ORD was included (WFA); middle table shows cumulative effects for results for bird and prey densities derived from local GPS data when all three ORDs were included (WFA+WFB+WFC Mapped), lower table shows cumulative effects for results for bird densities derived from distance-decay and the assumption of uniform prey when all three ORDs were included (WFA+WFB+WFC Uniform).

Buchan Ness WFA (Mapped older GPS tracking data flight removed):

PREY LEVEL	Total number of birds simulated (50% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
150 (G)	12542	0.00	0.00
155 (G)	12542	0.00	0.00
160 (G)	12542	0.00	0.00
165 (G)	12542	0.00	0.00
170 (G)	12542	0.00	0.00
175 (G)	12542	0.00	0.00

Buchan Ness WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Total number of birds simulated (50% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
150 (G)	12542	0.00	0.00
155 (G)	12542	0.00	0.00
160 (G)	12542	0.00	0.00
165 (G)	12542	0.00	0.00
170 (G)	12542	0.00	0.00

175 (G)	12542	0.00	0.00

Buchan Ness WFA+WFB+WFC (Distance-decay + Uniform):

PREY LEVEL	Total number of birds simulated (50% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
125 (M)	12542	0.01 (-0.03, 0.04)	0.00
130 (M)	12542	0.01 (-0.01, 0.03)	0.00
135 (M)	12542	0.002 (-0.02, 0.03)	0.005 (-0.02, 0.03)
140 (M)	12542	0.004 (-0.02, 0.03)	0.02 (-0.02, 0.07)
145 (M)	12542	0.006 (-0.01, 0.03)	0.06 (-0.03, 0.16)
150 (M)	12542	0.006 (-0.02, 0.04)	0.08 (-0.01, 0.16)

Table 4-7: As above for Table 4-6 for Fowlsheugh SPA

Fowlsheugh WFA (Mapped older GPS tracking data flight removed):

PREY LEVEL	Total number of birds simulated (50% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
150 (G)	9388	0.00 (-0.01, 0.01)	0.00
155 (G)	9388	0.00	0.00
160 (G)	9388	0.00	0.00
165 (G)	9388	0.00	0.00
170 (G)	9388	-0.00 (-0.01, 0.01)	0.00
175 (G)	9388	0.00 (-0.01, 0.01)	0.00

Fowlsheugh WFA + WFB + WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Total number of birds simulated (50% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
150 (G)	9388	0.00	0.00
155 (G)	9388	0.00	0.00
160 (G)	9388	0.00	0.00
165 (G)	9388	0.00 (-0.01, 0.01)	0.00
170 (G)	9388	0.00 (-0.01, 0.01)	0.00
175 (G)	9388	0.00 (-0.01, 0.02)	0.00

Fowlsheugh WFA+WFB+WFC (Distance-decay and Uniform):

PREY LEVEL	Total number of birds simulated (50% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
125 (M)	9388	0.01 (-0.01, 0.04)	0.00
130 (M)	9388	0.01 (-0.03, 0.05)	0.00
135 (M)	9388	0.02 (-0.02, 0.06)	0.004 (-0.04, 0.04)
140 (M)	9388	0.01 (-0.04, 0.05)	0.03 (-0.02, 0.07)
145 (M)	9388	0.01 (-0.04, 0.07)	0.07 (-0.02, 0.16)
150 (M)	9388	0.01 (-0.04, 0.06)	0.11 (0.00, 0.22)

Table 4-8: As above for Table 4-6 for Forth Islands SPA.

Forth Islands WFA (Mapped older GPS tracking data flight removed):

PREY LEVEL	Total number of birds simulated (50% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
150 (M)	3766	-0.01 (-0.12, 0.10)	0.11 (-0.09, 0.30)
155 (M)	3766	0.00 (-0.17, -0.17)	0.85 (0.28, 1.42)
160 (M)	3766	0.07 (-0.18, 0.32)	1.85 (1.14, 2.57)
165 (M)	3766	0.30 (-0.09, 0.68)	1.01 (0.41, 1.62)
170 (M)	3766	0.32 (0.11, 0.52)	0.22 (-0.06, 0.49)
175 (G)	3766	0.18 (0.03, 0.33)	0.02 (-0.07, 0.11)

Forth Islands WFA + WFB + WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Total number of birds simulated (50% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
150 (M)	3766	-0.01 (-0.36, 0.33)	0.46 (0.03, 0.89)
155 (M)	3766	0.06 (-0.34, 0.46)	4.34 (2.96, 5.72)
160 (M)	3766	0.40 (0.01, 0.78)	11.38 (9.96, 12.80)
165 (M)	3766	1.39 (1.00, 1.78)	9.22 (8.11, 10.3)
170 (M)	3766	2.11 (1.34, 2.88)	3.44 (2.43, 4.45)
175 (G)	3766	1.94 (1.37, 2.50)	0.75 (0.40, 1.11)

Forth Islands WFA+WFB+WFC (Distance-decay and Uniform):

PREY LEVEL	Total number of birds simulated (50% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
125 (M)	3766	0.14 (-0.16, 0.45)	1.25 (0.46, 2.03)
130 (M)	3766	0.02 (-0.34, 0.38)	2.49 (1.89, 3.08)
135 (M)	3766	0.19 (-0.17, 0.56)	2.99 (2.23, 3.74)
140 (M)	3766	0.24 (-0.05, 0.54)	3.20 (2.54, 3.86)
145 (M)	3766	0.34 (0.10, 0.58)	2.28 (1.43, 3.14)
150 (M)	3766	0.36 (0.06, 0.66)	1.67 (0.88, 2.47)

Forth Islands WFA+WFB+WFC (Mapped newer GPS tracking data flight not removed):

PREY LEVEL	Total number of birds simulated (40% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
105 (M)	3730	0.04 (-0.29, 0.36)	0.00
145 (M)	3730	0.47 (0.14, 0.80)	10.20 (8.89, 11.52)
155 (M)	3730	1.44 (0.86, 2.02)	4.63 (3.52, 5.75)
160 (M)	3730	1.60 (1.22, 1.98)	1.89 (0.81, 2.97)
165 (G)	3730	1.15 (0.86, 1.44)	0.66 (0.20, 1.12)

Forth Islands WFA+WFB+WFC (Mapped newer GPS tracking data flight removed):

PREY LEVEL	Total number of birds simulated (40% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
89 (M)	3730	0.13 (-0.28, 0.54)	5.56 (3.95, 7.16)
93 (M)	3730	0.45 (-0.02, 0.92)	8.13 (6.51, 9.75)
97 (M)	3730	0.76 (0.29, 1.22)	7.40 (6.52, 8.28)
103 (M)	3730	1.42 (0.81, 2.04)	2.80 (2.02, 3.59)
106 (G)	3730	1.11 (0.56, 1.65)	1.31 (0.74, 1.88)

Table 4-9: As above for Table 4-6 for St Abbs Head.

St Abbs Head WFA (Mapped older GPS tracking data flight removed):

PREY LEVEL	Total number of birds simulated (50% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
150 (M)	4314	0.08 (-0.22, 0.38)	0.36 (0.14, 0.58)
155 (M)	4314	0.07 (-0.14, 0.28)	2.07 (1.08, 3.06)
160 (M)	4314	0.29 (0.04, 0.54)	1.78 (1.05, 2.50)
165 (M)	4314	0.35 (0.08, 0.62)	0.35 (0.14, 0.56)
170 (G)	4314	0.11 (-0.01, 0.23)	0.04 (-0.06, 0.14)
175 (G)	4314	0.05 (-0.02, 0.12)	0.01 (-0.04, 0.06)

St Abbs Head WFA + WFB + WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Total number of birds simulated (50% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
150 (M)	4314	0.08 (-0.19, 0.35)	0.39 (0.11, 0.68)
155 (M)	4314	0.07 (-0.27, 0.40)	2.56 (1.83, 3.30)
160 (M)	4314	0.36 (0.13, 0.58)	2.35 (1.44, 3.26)
165 (G)	4314	0.33 (0.11, 0.55)	0.57 (0.25, 0.88)
170 (G)	4314	0.20 (-0.01, 0.42)	0.11 (0.01, 0.22)
175 (G)	4314	0.14 (-0.04, 0.32)	0.02 (-0.04, 0.080)

St Abbs Head WFA+WFB+WFC (Distance-decay + Uniform):

PREY LEVEL	Total number of birds simulated (50% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
125 (M)	4314	0.05 (-0.09, 0.18)	0.07 (-0.05, 0.18)
130 (M)	4314	0.04 (-0.15, 0.23)	0.10 (-0.06, 0.26)
135 (M)	4314	-0.01 (-0.19, 0.16)	0.24 (-0.11, 0.60)
140 (M)	4314	0.05 (-0.06, 0.16)	0.52 (0.07, 0.94)
145 (M)	4314	0.04 (-0.15, 0.23)	0.71 (0.22, 1.20)
150 (M)	4314	0.06 (-0.11, 0.22)	0.76 (0.34, 1.18)

4.2 Metrics on the Fate of Individual Birds

We next calculate these effects separately for the set of birds that ever directly interact with the footprint through either barrier or displacement effects (I2, “Dir”) and those birds that never do (I1, “Non”) (Table 4-10 to Table 4-13).

There was no detectable effect of any of the ORDs on birds at Buchan Ness (Table 4-6). No birds were found to interact with any of the ORDs either through displacement or barrier effects.

There were a few impacts of ORDs on directly affected birds (i2) from Fowlsheugh, however, all of the estimated impacts on adult mortality for directly affected birds (i2) were within the bounds of quantified uncertainty in the model output (95% prediction interval included zero), meaning that there was uncertainty in whether the ORDs increased the mortality of directly affected adults at all from this colony.

Birds from the Forth Islands that directly interacted with WFA (i2) experienced some increases in adult mortality; the mean effect on adult mortality ranged from 0.01% to 0.88% with 95% prediction intervals spanning -0.43% to 1.81% (Table 4-12). The largest increase in adult mortality for directly affected birds occurred when median prey density was 170, with a mean increase in adult mortality of directly affected birds of 0.88% (95% prediction interval: 0.40% to 1.37%). Non-affected birds from the Forth Islands (i1) all experienced very small positive effects (mean decrease in adult mortality ranged from -0.004% to -0.05%); however, all of these estimated effects on non-affected birds had an estimated 95% prediction interval that included zero, indicating that there was uncertainty in whether WFA had any impact upon the mortality of birds that did not directly interact with the ORD.

Birds from St Abbs Head that directly interacted with WFA (i2) experienced some increases in adult mortality; the mean effect on adult mortality ranged from 0.14% to 0.92% with 95% prediction intervals spanning -0.45% to 1.55% (Table 4-13). The largest increase in adult mortality for directly affected birds occurred when median prey density was 165, with a mean increase in adult mortality of directly affected birds of 0.92% (95% prediction interval: 0.29% to 1.55%). Non-affected birds from St Abbs Head (i1) all experienced very small positive effects (mean decrease in adult mortality ranged from -0.05% to 0.00%); however, all of these estimated effects on non-affected birds had an estimated 95% prediction interval that included zero, indicating that there was uncertainty in whether WFA had any impact upon the mortality of birds that did not directly interact with the ORD.

Birds from the Forth Islands that directly interacted with WFA + WFB + WFC (i2) experienced some increases in adult mortality; the mean effect on adult mortality ranged from 0.33% to 5.33% with 95% prediction intervals spanning -0.27% to 7.11% (Table 4-12). T he largest increase in adult mortality for directly affected birds occurred when median prey density was 170, with a mean increase in adult mortality of directly affected birds of 5.33% (95% prediction interval: 3.54% to 7.11%). Non-affected birds from the Forth Islands (i1) again all experienced very small positive effects (mean decrease in adult mortality ranged from -0.02% to -0.28%); most of these estimated effects on non-affected birds had an estimated 95% prediction interval that included zero, indicating that there was uncertainty in whether WFA had any impact upon the mortality of birds that did not directly interact with the ORD. However, when regional prey density was relatively low (150 or 155) the 95% prediction intervals indicated that there was a real decrease in adult mortality for non-affected birds (Table 4-12).

Birds from St Abbs Head that directly interacted with WFA + WFB + WFC (i2) experienced some increases in adult mortality; the mean effect on adult mortality ranged from 0.27% to 1.20% with 95% prediction intervals spanning -0.46% to 1.90% (Table 4-13). The largest increase in adult mortality for directly affected birds occurred when median prey density was 165, with a mean increase in adult mortality of directly affected birds of 1.20% (95% prediction interval: 0.50% to 1.90%). Non-affected birds from St Abbs Head (i1) all experienced very small positive effects (mean decrease in adult mortality ranged from -0.28% to 0.00%); however, all of these estimated effects on non-affected birds had an estimated 95% prediction interval that included zero, indicating that there was uncertainty in whether WFA had any impact upon the mortality of birds that did not directly interact with the ORD.

Table 4-10: Impact of each wind farm upon adult mortality rates for birds that are directly impacted by the wind farm at any point during the breeding season in terms of either displacement or barrier effects (“Dir I2”) and those that are not (“Non I1”). The number of birds in each category is presented along with the percentage of the population in brackets. Results are based on ten paired runs of 50% of the total population, and relate to black-legged kittiwakes from all SPA colonies in the Forth Tay region. Impact is defined as the mean percent additional mortality for each set of birds with associated 95% prediction intervals. Positive values represent an increase in mortality associated with the impact, negative values represent a decrease in mortality associated with the impact. Upper table ‘WFA’ shows results for bird and prey densities derived from local GPS data when only one ORD was included (WFA); middle table shows cumulative effects for results for bird and prey densities derived from local GPS data when all three ORDs were included (WFA+WFB+WFC), lower table shows cumulative effects for results for bird densities derived from distance-decay and the assumption of uniform prey when all three ORDs were included (WFA+WFB+WFC).

Buchan Ness WFA (Mapped older GPS tracking data flight removed):

PREY LEVEL	**Mean Number of birds per category**		**Impact on adult mortality (percentage points)**	
	Dir I2 (%)	**Non I1 (%)**	**Dir (I2)**	**Non (I1)**
150 (G)	0 (0)	12542 (100)	0	0
155 (G)	0 (0)	12542 (100)	0	0
160 (G)	0 (0)	12542 (100)	0	0
165 (G)	0 (0)	12542 (100)	0	0
170 (G)	0 (0)	12542 (100)	0	0
175 (G)	0 (0)	12542 (100)	0	0

Buchan Ness WFA + WFB + WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	**Mean Number of birds per category**		**Impact on adult mortality (percentage points)**	
	Dir I2 (%)	**Non I1 (%)**	**Dir (I2)**	**Non (I1)**
150 (G)	0 (0%)	12542 (100%)	0	0
155 (G)	0 (0%)	12542 (100%)	0	0
160 (G)	0 (0%)	12542 (100%)	0	0
165 (G)	0 (0%)	12542 (100%)	0	0
170 (G)	0 (0%)	12542 (100%)	0	0
175 (G)	0 (0%)	12542 (100%)	0	0

Buchan Ness WFA + WFB + WFC (Distance-decay + Uniform):

PREY LEVEL	**Mean Number of birds per category**		**Impact on adult mortality (percentage points)**	
	Dir I2 (%)	**Non I1 (%)**	**Dir (I2)**	**Non (I1)**
125 (M)	89 (<1%)	12453 (>99%)	0.36 (-1.01, 1.72)	0.003 (-0.03, 0.04)
130 (M)	89 (<1%)	12453 (>99%)	0.00	0.01 (-0.01, 0.03)
135 (M)	89 (<1%)	12453 (>99%)	-0.002 (-1.43, 1.42)	0.002 (-0.02, 0.03)
140 (M)	89 (<1%)	12453 (>99%)	-0.13 (-1.09, 0.83)	0.005 (-0.02, 0.03)
145 (M)	89 (<1%)	12453 (>99%)	0.13 (-0.82, 1.08)	0.005 (-0.02, 0.03)
150 (M)	89 (<1%)	12453 (>99%)	0.12 (-0.80, 1.05)	0.006 (-0.03, 0.04)

Table 4-11: As above for Table 4-10 for Fowlsheugh SPA.

Fowlsheugh WFA (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	**Non I1 (%)**	**Dir (I2)**	**Non (I1)**
150 (G)	13.7 (<0.1%)	9374.0 (99.9%)	0.44 (-2.83, 3.70)	0.00
155 (G)	13.7 (<0.1%)	9374.0 (99.9%)	0.00	0.00
160 (G)	13.7 (<0.1%)	9374.0 (99.9%)	0.00	0.00
165 (G)	13.7 (<0.1%)	9374.0 (99.9%)	0.00	0.00
170 (G)	13.7 (<0.1%)	9374.0 (99.9%)	-1.11 (-9.45, 7.23)	0.00
175 (G)	13.7 (<0.1%)	9374.0 (99.9%)	-0.44 (-10.63, 9.74)	0.00

Fowlsheugh WFA + WFB + WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	**Non I1 (%)**	**Dir (I2)**	**Non (I1)**
150 (G)	28.7 (<0.1%)	9359.3 (99.7%)	0.00	0.00
155 (G)	28.7 (<0.1%)	9359.3 (99.7%)	0.00	0.00
160 (G)	28.7 (<0.1%)	9359.3 (99.7%)	0.00	0.00
165 (G)	28.7 (<0.1%)	9359.3 (99.7%)	0.48 (-3.10, 4.05)	0.00
170 (G)	28.7 (<0.1%)	9359.3 (99.7%)	0.48 (-3.10, 4.05)	0.00
175 (G)	28.7 (<0.1%)	9359.3 (99.7%)	-0.12 (-4.82, 4.58)	0.00 (-0.01, 0.01)

Fowlsheugh WFA + WFB + WFC (Distance-decay and Uniform):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	**Non I1 (%)**	**Dir (I2)**	**Non (I1)**
125 (M)	805.8 (8.6%)	8582.2 (91.4%)	0.05 (-0.23, 0.34)	0.01 (-0.02, 0.04)
130 (M)	805.8 (8.6%)	8582.2 (91.4%)	0.01 (-0.24, 0.26)	0.01 (-0.03, 0.05)
135 (M)	805.8 (8.6%)	8582.2 (91.4%)	0.09 (-0.06, 0.23)	0.01 (-0.03, 0.05)
140 (M)	805.8 (8.6%)	8582.2 (91.4%)	-0.03 (-0.41, 0.36)	0.01 (-0.02, 0.04)
145 (M)	805.8 (8.6%)	8582.2 (91.4%)	0.08 (-0.30, 0.45)	0.01 (-0.03, 0.05)
150 (M)	805.8 (8.6%)	8582.2 (91.4%)	0.06 (-0.39, 0.51)	0.005 (-0.03, 0.04)

Table 4-12: As above for Table 4-10 for Forth Islands SPA.

Forth islands WFA (Mapped older GPS data with flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	**Non I1 (%)**	**Dir (I2)**	**Non (I1)**
150 (M)	1357.0 (36.0%)	2409.0 (64.0%)	0.01 (-0.30, 0.32)	-0.03 (-0.12, 0.06)
155 (M)	1357.0 (36.0%)	2409.0 (64.0%)	0.06 (-0.43, 0.55)	-0.03 (-0.13, 0.07)
160 (M)	1357.0 (36.0%)	2409.0 (64.0%)	0.28 (-0.41, 0.96)	-0.05 (-0.24, 0.13)
165 (M)	1357.0 (36.0%)	2409.0 (64.0%)	0.85 (-0.12, 1.81)	-0.02 (-0.11, 0.08)
170 (M)	1357.0 (36.0%)	2409.0 (64.0%)	0.88 (0.40, 1.37)	-0.004 (-0.08, 0.07)
175 (G)	1357.0 (36.0%)	2409.0 (64.0%)	0.51 (0.12, 0.90)	-0.004 (-0.04, 0.03)

Forth islands WFA + WFB + WFC (Mapped older GPS data with flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
150 (M)	1507.6 (40.0%)	2258.4 (60.0%)	0.33 (-0.27, 0.93)	-0.24 (-0.49, -0.002)
155 (M)	1507.6 (40.0%)	2258.4 (60.0%)	0.58 (-0.24, 1.39)	-0.28 (-0.55, -0.02)
160 (M)	1507.6 (40.0%)	2258.4 (60.0%)	1.41 (0.49, 2.33)	-0.28 (-0.56, 0.01)
165 (M)	1507.6 (40.0%)	2258.4 (60.0%)	3.74 (2.83, 4.65)	-0.18 (-0.41, 0.05)
170 (M)	1507.6 (40.0%)	2258.4 (60.0%)	5.33 (3.54, 7.11)	-0.04 (-0.14, 0.07)
175 (G)	1507.6 (40.0%)	2258.4 (60.0%)	3.81 (2.38, 4.94)	-0.02 (-0.07, 0.04)

Forth Islands WFA + WFB + WFC (Distance-decay and Uniform):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
125 (M)	1458.2 (38.7%)	2307.8 (61.3%)	0.49 (-0.20, 1.18)	-0.07 (-0.18, 0.04)
130 (M)	1458.2 (38.7%)	2307.8 (61.3%)	0.24 (-0.58, 1.07)	-0.11 (-0.31, 0.08)
135 (M)	1458.2 (38.7%)	2307.8 (61.3%)	0.68 (-0.09, 1.44)	-0.11 (-0.26, 0.03)
140 (M)	1458.2 (38.7%)	2307.8 (61.3%)	0.72 (-0.05, 1.49)	-0.05 (-0.16, 0.05)
145 (M)	1458.2 (38.7%)	2307.8 (61.3%)	0.94 (0.39, 1.49)	-0.04 (-0.11, 0.03)
150 (M)	1458.2 (38.7%)	2307.8 (61.3%)	0.96 (0.18, 1.75)	-0.03 (-0.10, 0.05)

Forth Islands WFA + WFB + WFC (Mapped newer GPS tracking flight not removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
105 (M)	1495.8 (40.1%)	2234.2 (59.9%)	0.26 (-0.44, 0.96)	-0.12 (-0.27, 0.04)
145 (M)	1495.8 (40.1%)	2234.2 (59.9%)	1.57 (0.87, 2.26)	-0.27 (-0.45, -0.08)
155 (M)	1495.8 (40.1%)	2234.2 (59.9%)	3.70 (2.35, 5.04)	-0.07 (-0.16, 0.02)
160 (M)	1495.8 (40.1%)	2234.2 (59.9%)	4.02 (2.86, 5.18)	-0.02 (-0.12, 0.09)
165 (G)	1495.8 (40.1%)	2234.2 (59.9%)	2.88 (2.05, 3.71)	0.00

Forth Islands WFA + WFB + WFC (Mapped newer GPS tracking flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
89 (M)	1507.0 (40.4%)	2223.0 (59.6%)	0.65 (-0.25, 1.55)	-0.22 (-0.59, 0.14)
93 (M)	1507.0 (40.4%)	2223.0 (59.6%)	1.43 (0.31, 2.56)	-0.21 (-0.52, 0.10)
97 (M)	1507.0 (40.4%)	2223.0 (59.6%)	2.00 (1.03, 2.98)	-0.09 (-0.27, 0.09)
103 (M)	1507.0 (40.4%)	2223.0 (59.6%)	3.46 (2.00, 4.91)	0.04 (-0.12, 0.21)
106 (G)	1507.0 (40.4%)	2223.0 (59.6%)	2.67 (1.36, 3.98)	0.05 (-0.08, 0.17)

Table 4-13: As above for Table 4-10 for St Abbs Head SPA.

St Abbs Head WFA (Mapped older GPS data with flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
150 (M)	1688.9 (39.1%)	2625.1 (60.9%)	0.28 (-0.45, 1.01)	-0.05 (-0.19, 0.09)
155 (M)	1688.9 (39.1%)	2625.1 (60.9%)	0.27 (-0.19, 0.72)	-0.03 (-0.14, 0.07)
160 (M)	1688.9 (39.1%)	2625.1 (60.9%)	0.79 (0.21, 1.36)	-0.03 (-0.17, 0.11)
165 (M)	1688.9 (39.1%)	2625.1 (60.9%)	0.92 (0.29, 1.55)	-0.02 (-0.08, 0.05)
170 (G)	1688.9 (39.1%)	2625.1 (60.9%)	0.28 (-0.00, 0.57)	-0.00 (-0.03, 0.03)
175 (G)	1688.9 (39.1%)	2625.1 (60.9%)	0.14 (-0.04, 0.31)	0.00

St Abbs Head WFA + WFB + WFC (Mapped older GPS data with flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
150 (M)	1724.0 (40%)	2590.0 (60.0%)	0.27 (-0.40, 0.94)	-0.05 (-0.19, 0.10)
155 (M)	1724.0 (40%)	2590.0 (60.0%)	0.29 (-0.46, 1.03)	-0.28 (-0.55, -0.02)
160 (M)	1724.0 (40%)	2590.0 (60.0%)	0.99 (0.51, 1.47)	-0.06 (-0.23, 0.10)
165 (M)	1724.0 (40%)	2590.0 (60.0%)	1.20 (0.50, 1.90)	-0.01 (-0.13, 0.10)
170 (G)	1724.0 (40%)	2590.0 (60.0%)	0.51 (0.02, 0.99)	0.00 (-0.06, 0.06)
175 (G)	1724.0 (40%)	2590.0 (60.0%)	0.36 (-0.08, 0.80)	0.00

St Abbs Head WFA + WFB + WFC (Distance-decay and Uniform):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
125 (M)	1557.8 (36.1%)	2756.2 (63.9%)	0.13 (-0.30, 0.55)	0.004 (-0.06, 0.07)
130 (M)	1557.8 (36.1%)	2756.2 (63.9%)	0.11 (-0.53, 0.74)	-0.004 (-0.12, 0.11)
135 (M)	1557.8 (36.1%)	2756.2 (63.9%)	-0.02 (-0.34, 0.30)	-0.01 (-0.15, 0.13)
140 (M)	1557.8 (36.1%)	2756.2 (63.9%)	0.17 (-0.09, 0.43)	-0.01 (-0.14, 0.11)
145 (M)	1557.8 (36.1%)	2756.2 (63.9%)	0.11 (-0.38, 0.60)	0.00 (-0.08, 0.08)
150 (M)	1557.8 (36.1%)	2756.2 (63.9%)	0.16 (-0.23, 0.55)	-0.004 (-0.07, 0.06)

We separate the directly impacted birds further (Table 4-14 to Table 4-17), into those which experience only displacement effects, only barrier effects, or both.

As for the previous metrics, there were no detectable effects on adult birds in any of the three categories (1) birds that experience displacement but not barrier effects “Disp”, I3; (2) birds that experience barrier but not displacement effects “Bar”, I4; and (3) birds that experience both “Both”, I5) for birds from Buchan Ness or Fowlsheugh (Table 4-14 and Table 4-15). No birds from Buchan Ness interacted with any of the ORDs, and only a few birds from Fowlsheugh interacted with WFA (up to 16 individuals) or with all three wind farms (WFA + WFB + WFC = up to 31 individuals).

On average 1,357 adult birds from Forth Islands interacted with WFA (Table 4-16). Most birds suffered either only barrier effects, or both barrier and displacement effects, with far fewer individuals suffering only from displacement. The largest increases to adult mortality occurred for birds that were affected by both barrier and displacement effects (I5) with mean increases of -0.02% to 1.13%. We can deduce that most of this effect came from barrier effects because of the birds that were only barrier affected by WFA (I4) there were also mean increases of 0.05% to 0.77% across the range of six prey densities, whilst mean impacts on birds that were only displaced were lower (-0.20% to 0.02%).

On average 1,689 adult birds from St Abbs Head interacted with WFA (Table 4-17). Most birds suffered either only barrier effects, or both barrier and displacement effects, with far fewer individuals suffering only from displacement. The largest increases to adult mortality occurred for birds that were affected by both barrier and displacement effects (I5) with mean increases of 0.12% to 1.02%. We can deduce that most of this effect came from barrier effects because of the birds that were only barrier affected by WFA (I4) there were also mean increases of 0.03% to 0.83% across the range of six prey densities, whilst mean impacts on birds that were only displaced were lower (-0.18% to 0.73%).

For birds from St Abbs Head the cumulative effects of all three ORDs (WFA + WFB + WFC) were similar to those of only WFA with 1,724 individuals interacting directly with one of the ORDs compared to 1,689 when only WFA is present (Table 4-17). Most birds suffered either only barrier effects, or both barrier and displacement effects, with far fewer individuals suffering only from displacement. The largest increases to adult mortality occurred for birds that were affected by both barrier and displacement effects (I5) with mean increases of 0.28% to 1.27%. We can deduce that most of this effect came from barrier effects because of the birds that were only barrier affected (I4) there were also mean increases of 0.04% to 1.11% across the range of six prey densities, whilst mean impacts on birds that were only displaced were lower in all cases (0.00% to 0.77%) except when median prey density was equal to 170 (2.22%).

However, for birds from Forth Islands, the cumulative effects of all three ORDs on individuals from the different categories were in general much larger than when only WFA were present (Table 4-16). When all three ORDs were present 1,508 birds were directly affected compared to only 1,357 individuals when only WFA was present. Almost all of the birds suffered either only barrier effects, or both barrier and displacement effects, with only one individual suffering only from displacement. The largest increases to adult mortality occurred for birds that were affected by both barrier and displacement effects (I5) with mean increases of 0.33% to 5.66%. We can deduce that most of this effect came from barrier effects because of the birds that were only barrier affected (I4) there were also mean increases of 0.36% to 4.57% across the range of six prey densities, whilst there were no impacts from birds that were only displaced.

Table 4-14: Impact of each wind farm upon adult and chick mortality rates for: a) birds that experience displacement but not barrier effects ("Disp", I3), b) birds that experience barrier but not displacement effects ("Bar", I4), c) birds that experience both ("Both", I5). Results are based on ten paired runs of 50% of the total population, and relate only to black-legged kittiwakes from SPA colonies in the Forth Tay region. Impact is defined as the mean percent additional mortality for each set of birds with associated 95% prediction intervals. Positive values represent an increase in mortality associated with the impact, negative values represent a decrease in mortality associated with the impact. Upper table 'WFA' shows results for bird and prey densities derived from local GPS data when only one ORD was included (WFA Mapped); middle table shows cumulative effects for results for bird and prey densities derived from local GPS data when all three ORDs were included (WFA+WFB+WFC Mapped), lower table shows cumulative effects for results for bird densities derived from distance-decay and the assumption of uniform prey when all three ORDs were included (WFA+WFB+WFC Uniform).

Buchan Ness WFA (Mapped older GPS data with flight removed):

PREY LEVEL	**Mean Number of birds per category**			**Impact on adult mortality (percentage points)**		
	Disp I3	**Bar I4**	**Both I5**	**Disp I3**	**Bar I4**	**Both I5**
150 (G)	0	0	0	0	0	0
155 (G)	0	0	0	0	0	0
160 (G)	0	0	0	0	0	0
165 (G)	0	0	0	0	0	0
170 (G)	0	0	0	0	0	0
175 (G)	0	0	0	0	0	0

Buchan Ness WFA + WFB + WFC (Mapped older GPS data with flight removed):

PREY LEVEL	**Mean Number of birds per category**			**Impact on adult mortality (percentage points)**		
	Disp I3	**Bar I4**	**Both I5**	**Disp I3**	**Bar I4**	**Both I5**
150 (G)	0	0	0	0	0	0
155 (G)	0	0	0	0	0	0
160 (G)	0	0	0	0	0	0
165 (G)	0	0	0	0	0	0
170 (G)	0	0	0	0	0	0
175 (G)	0	0	0	0	0	0

Buchan Ness WFA + WFB + WFC (Distance-decay and Uniform):

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
125 (M)	28.3	54.4	6.2	0.40 (-2.60, 3.40)	0.36 (-1.45, 2.17)	0.00
130 (M)	25.1	48.4	5.9	0.00	0.00	0.00
135 (M)	29.2	54.5	5.3	-0.37 (-3.15, 2.41)	0.20 (-1.28, 1.67)	0.00
140 (M)	28.2	54.5	6.3	-0.37 (-3.15, 2.41)	0.00	0.00
145 (M)	28.4	54.5	6.1	0.00	0.20 (-1.28, 1.67)	0.00
150 (M)	25.1	48.4	5.9	0.00	0.00	1.11 (-7.23, 9.45)

Table 4-15: As above for Table 4-14 for Fowlsheugh SPA.

Fowlsheugh WFA (Mapped older GPS data with flight removed)**:**

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
150 (G)	7.5 (<0.1%)	4.8 (<0.1%)	1.6 (<0.1%)	0	0	5.56 (-34.0, 45.1)
155 (G)	8.4 (<0.1%)	4.8 (<0.1%)	1.3 (<0.1%)	0	0	0
160 (G)	8.0 (<0.1%)	4.8 (<0.1%)	1.3 (<0.1%)	0	0	0
165 (G)	8.3 (<0.1%)	4.8 (<0.1%)	1.5 (<0.1%)	0	0	0
170 (G)	8.1 (<0.1%)	4.8 (<0.1%)	2.0 (<0.1%)	-1.67 (-14.2, 10.8)	0	0
175 (G)	8.5 (<0.1%)	4.8 (<0.1%)	1.3 (<0.1%)	-1.67 (-14.2, 10.8)	2.00 (-13.0, 17.0)	0

Fowlsheugh WFA + WFB + WFC (Mapped older GPS data with flight removed)**:**

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
150 (G)	15.5 (<0.1%)	11.9 (<0.1%)	1.6 (<0.1%)	0	0	0
155 (G)	14.8 (<0.1%)	11.9 (<0.1%)	2.2 (<0.1%)	0	0	0
160 (G)	15.1 (<0.1%)	11.9 (<0.1%)	2.4 (<0.1%)	0	0	0
165 (G)	14.8 (<0.1%)	11.9 (<0.1%)	2.2 (<0.1%)	1.00 (-6.50, 8.50)	0	0
170 (G)	15.7 (<0.1%)	11.9 (<0.1%)	1.6 (<0.1%)	1.00 (-6.50, 8.50)	0	0
175 (G)	14.9 (<0.1%)	11.9 (<0.1%)	2.7 (<0.1%)	-1.00 (-8.50, 6.50)	0.91 (-5.91, 7.73)	0

Fowlsheugh WFA + WFB + WFC (Distance-decay and Uniform):

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
125 (M)	211.9	499.1	94.8	0.09 (-0.36, 0.53)	0.02 (-0.31, 0.35)	0.12 (-0.76, 0.99)
130 (M)	217.2	510.5	93.5	-0.05 (-0.39, 0.30)	0.06 (-0.32, 0.43)	-0.07 (-1.68, 1.53)
135 (M)	210.7	499.1	96.0	0.10 (-0.38, 0.57)	0.08 (-0.17, 0.33)	0.10 (-0.63, 0.82)
140 (M)	215.5	499.1	91.2	0.002 (-0.70, 0.70)	-0.04 (-0.41, 0.33)	0.00
145 (M)	212.9	499.1	93.8	0.10 (-0.64, 0.84)	0.04 (-0.26, 0.33)	0.22 (-1.43, 1.87)
150 (M)	212.7	510.5	98.0	0.13 (-0.76, 1.02)	0.04 (-0.32, 0.40)	0.01 (-1.21, 1.23)

Table 4-16: As above for Table 4-14 for Forth Islands SPA.

Forth Islands WFA (Mapped older GPS data with flight removed)**:**

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
150 (M)	56.5 (1.5%)	859.3 (22.8%)	441.2 (11.7%)	-0.15 (-1.23, 0.94)	0.05 (-0.22, 0.32)	-0.02 (-0.81, 0.77)
155 (M)	53.5 (1.4%)	859.3 (22.8%)	444.2 (11.8%)	-0.05 (-2.06, 1.95)	0.06 (-0.42, 0.54)	0.07 (-0.70, 0.84)
160 (M)	53.8 (1.4%)	859.3 (22.8%)	443.9 (11.8%)	-0.52 (-2.52, 1.48)	0.39 (-0.30, 1.08)	0.16 (-0.84, 1.17)
165 (M)	52.5 (1.4%)	859.3 (22.8%)	445.2 (11.8%)	-0.20 (-1.67, 1.28)	0.77 (-0.16, 1.69)	1.13 (-0.65, 2.90)
170 (M)	54.7 (1.5%)	859.3 (22.8%)	443.0 (11.8%)	0.02 (-2.23, 2.26)	0.68 (-0.11, 1.48)	1.08 (-0.12, 2.29)
175 (G)	52.4 (1.4%)	859.3 (22.8%)	445.3 (11.8%)	0.00	0.51 (-0.14, 1.16)	0.54 (-0.39, 1.47)

Forth Islands WFA + WFB + WFC (Mapped older GPS data with flight removed)**:**

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
150 (M)	1.0 (<1%)	453.0 (12.0%)	1054.4 (28.0%)	0.00	0.36 (-0.69, 1.41)	0.33 (-0.47, 1.12)
155 (M)	1.0 (<1%)	453.0 (12.0%)	1054.1 (28.0%)	0.00	0.52 (-1.03, 2.06)	0.60 (-0.16, 1.35)
160 (M)	1.0 (<1%)	453.0 (12.0%)	1054.2 (28.0%)	0.00	1.33 (-0.31, 2.97)	1.45 (0.55, 2.35)
165 (M)	1.0 (<1%)	453.0 (12.0%)	1054.2 (28.0%)	0.00	3.62 (1.19, 6.05)	3.80 (2.67, 4.94)
170 (M)	1.0 (<1%)	453.0 (12.0%)	1054.2 (28.0%)	0.00	4.57 (2.33, 6.81)	5.66 (3.36, 7.97)
175 (G)	1.0 (<1%)	453.0 (12.0%)	1054.3 (28.0%)	0.00	3.09 (1.63, 4.55)	4.12 (2.57, 5.66)

Forth Islands WFA + WFB + WFC (Distance-decay and Uniform):

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
125 (M)	28.9 (<1%)	675.7 (17.9%)	753.6 (20.0%)	0.29 (-1.86, 2.43)	0.37 (-0.36, 1.11)	0.60 (-0.47, 1.67)
130 (M)	33.2 (<1%)	665.6 (17.7%)	751.8 (20.0%)	0.00	0.32 (-1.27, 1.91)	0.17 (-1.15, 1.49)
135 (M)	29.6 (<1%)	675.7 (17.9%)	752.9 (20.0%)	0.00	0.62 (-0.90, 2.15)	0.76 (0.20, 1.31)
140 (M)	29.5 (<1%)	675.7 (17.9%)	753.0 (20.0%)	0.00	0.55 (-0.26, 1.35)	0.90 (-0.18, 1.98)
145 (M)	28.8 (<1%)	675.7 (17.9%)	753.7 (20.0%)	0.00	0.99 (-0.05, 2.03)	0.93 (0.11, 1.75)
150 (M)	33.9 (<1%)	665.6 (17.7%)	751.1 (20.0%)	0.00	0.87 (-0.03, 1.78)	1.09 (-0.01, 2.19)

Forth Islands WFA + WFB + WFC (Mapped newer GPS tracking flight not removed):

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
105 (M)	0.0	361.1 (9.7%)	1134.7 (30.4%)	0.00	-0.05 (-1.62, 1.51)	0.36 (-0.37, 1.09)
145 (M)	0.0	361.1 (9.7%)	1134.7 (30.4%)	0.00	1.11 (0.27, 1.95)	1.71 (0.83, 2.59)
155 (M)	0.0	361.1 (9.7%)	1134.7 (30.4%)	0.00	3.85 (0.33, 7.38)	3.66 (2.13, 5.18)
160 (M)	0.0	361.1 (9.7%)	1134.7 (30.4%)	0.00	4.37 (2.69, 6.06)	3.91 (2.22, 5.60)
165 (G)	0.0	361.1 (9.7%)	1134.7 (30.4%)	0.00	3.37 (1.47, 5.27)	2.73 (1.46, 4.00)

Forth Islands WFA + WFB + WFC (Mapped newer GPS tracking flight removed):

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
89 (M)	1.0	409.3 (11.0%)	1097.5 (29.4%)	0.00	0.36 (-1.09, 1.81)	0.76 (-0.40, 1.91)
93 (M)	1.0	409.3 (11.0%)	1097.4 (29.4%)	0.00	1.27 (-0.65, 3.21)	1.49 (0.43, 2.56)
97 (M)	1.0	409.3 (11.0%)	1097.4 (29.4%)	0.00	1.89 (-0.77, 4.55)	2.05 (1.15, 2.94)
103 (M)	1.0	409.3 (11.0%)	1097.4 (29.4%)	0.00	2.99 (1.17, 4.82)	3.63 (2.10, 5.16)
106 (G)	1.0	409.3 (11.0%)	1097.5 (29.4%)	0.00	3.00 (0.50, 5.49)	2.55 (0.99, 4.12)

Table 4-17: As above for Table 4-14 for St Abbs Head SPA.

St Abbs Head WFA (Mapped older GPS data with flight removed)**:**

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
150 (M)	31.0 (<1%)	878.6 (20.4%)	779.3 (18.1%)	-0.18 (-4.59, 4.22)	0.22 (-0.63, 1.07)	0.37 (-0.40, 1.13)
155 (M)	29.6 (<1%)	878.6 (20.4%)	780.7 (18.1%)	-0.11 (-5.88, 5.67)	0.03 (-0.75, 0.81)	0.54 (-0.24, 1.33)
160 (M)	31.3 (<1%)	878.6 (20.4%)	779.0 (18.1%)	0.58 (-2.32, 3.48)	0.64 (0.04, 1.24)	0.96 (-0.21, 2.13)
165 (M)	27.9 (<1%)	878.6 (20.4%)	782.4 (18.1%)	0.73 (-3.04, 4.49)	0.83 (0.19, 1.47)	1.02 (0.01, 2.03)
170 (G)	28.9 (<1%)	878.6 (20.4%)	781.4 (18.1%)	0.00	0.31 (-0.04, 0.66)	0.27 (-0.23, 0.78)
175 (G)	30.8 (<1%)	878.6 (20.4%)	779.5 (18.1%)	0.28 (-1.81, 2.36)	0.15 (-0.16, 0.45)	0.12 (-0.11, 0.35)

St Abbs Head WFA + WFB + WFC (Mapped older GPS data with flight removed)**:**

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
150 (M)	18.0 (<1%)	615.1 (14.1%)	1090.9 (25.3%)	0.39 (-2.50, 3.27)	0.25 (-0.37, 0.86)	0.28 (-0.65, 1.21)
155 (M)	18.1 (<1%)	615.1 (14.1%)	1090.8 (25.3%)	0.77 (-5.00, 6.54)	0.04 (-1.30, 1.39)	0.42 (-0.47, 1.31)
160 (M)	18.2 (<1%)	615.1 (14.1%)	1090.7 (25.3%)	1.1.1.1.1.1.1 0.59 (-3.83, 5.00)	0.86 (0.15, 1.58)	1.06 (0.21, 1.91)
165 (M)	18.5 (<1%)	615.1 (14.1%)	1090.4 (25.3%)	0.00	1.11 (-0.08, 2.29)	1.27 (0.55, 2.00)
170 (G)	18.0 (<1%)	615.1 (14.1%)	1090.9 (25.3%)	2.22 (-14.5, 18.9)	0.45 (0.04, 0.87)	1.1.1.1.1.1.2 0.52 (-0.18, 1.23)
175 (G)	17.4 (<1%)	615.1 (14.1%)	1091.5 (25.3%)	0.00	0.42 (-0.18, 1.02)	0.33 (-0.14, 0.80)

St Abbs Head WFA + WFB + WFC (Distance-decay and Uniform):

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
125 (M)	140.1 (3.2%)	669.6 (15.5%)	748.1 (17.3%)	-0.01 (-1.18, 1.16)	0.10 (-0.48, 0.67)	0.17 (-0.49, 0.83)
130 (M)	141.5 (3.2%)	676.4 (15.7%)	755.9 (17.5%)	-0.02 (-1.44, 1.40)	0.15 (-0.47, 0.77)	0.09 (-0.72, 0.90)
135 (M)	139.4 (3.2%)	669.6 (15.5%)	748.8 (17.3%)	-0.12 (-1.43, 1.20)	-0.08 (-0.56, 0.40)	0.05 (-0.64, 0.75)
140 (M)	135.2 (3.2%)	669.6 (15.5%)	753.0 (17.5%)	-0.06 (-1.33, 1.20)	0.22 (-0.11, 0.56)	0.16 (-0.55, 0.87)
145 (M)	138.1 (3.2%)	669.6 (15.5%)	750.1 (17.4%)	-0.06 (-1.23, 1.12)	0.21 (-0.20, 0.62)	0.05 (-0.63, 0.74)
150 (M)	145.6 (3.4%)	676.4 (15.7%)	751.8 (17.4%)	0.009 (-0.74, 0.76)	0.07 (-0.60, 0.75)	0.27 (-0.65, 1.18)

4.3 Snapshot surveys

We next calculate the metric, P2, which quantifies the impact of the ORD upon a bird seen within a “snapshot” survey of the ORD footprint. This metric is:

$$\frac{\text{number of birds simulated to die with ORD - number of birds simulated to die without ORD}}{\text{number of birds simulated to be present in survey of ORD footprint}}$$

This metric, therefore, gives the additional number of birds that are expected to experience mortality as a result of the ORD for every individual bird sighted during the ORD footprint surveys. For example, if P2=0.5, one half of an additional bird is expected to die as a result of the ORD for every bird sighted in the survey, meaning that if 100 birds are sighted in the survey, an additional 50 birds are expected to die as a result of the ORD at the population level.

We calculated P2 for all ORDs combined (WFA + WFB + WFC) with three at-sea surveys conducted on the same day for all three footprints. The results are combined across all three surveys for each paired run (ten paired runs in the table below). Finally, the mean value for P2 is taken over all ten paired simulations to produce the final P2 metric (Table 4 18). The final mean value of P2 is specific to the level of prey availability in the model (poor, moderate, or good). This is because a ‘baseline survival’ is assumed in the calculations deriving the relationship between adult mass at the end of the breeding season and subsequent survival over the rest of the year (see Methods). This ‘baseline survival’ is dependent upon assumptions about the level of prey availability in the model region (poor, moderate, or good).

We ran the snapshot survey in the simulations with median prey density set at 170 which represented ‘good’ conditions for all colonies except the Forth Islands where it represented the upper end of ‘moderate’ prey conditions. Because P2 is expressed at the regional population level (over all colonies), we, therefore, used the P2 metric generated under assumptions of ‘good’ conditions (Table 4 18; ‘GOOD metric’). This shows that averaged over all ten sets of paired simulations and over all three ORD footprints we expect an additional 0.58 (SD 0.09) adult birds to experience mortality as a result of the three ORDs. The average number of birds observed over the three ORD footprints was 129 (Table 4 18), therefore, we expect an additional 75 adult birds to die as a result of the three ORDs, given the number of birds observed during snapshot surveys (129*0.58=74.8).

Table 4-18: Results from simulated "snapshot" surveys on three days during one set of ten matched pairs (baseline + WFA+WFB+WFC). The 'Birds sighted' is the sum of the birds sighted in the ORD footprints (all combined; WFA + WFB + WFC) averaged over the three survey days during each paired simulation. The 'P2' metric is calculated for each potential level of prey conditions (poor, moderate, or good), and the relevant level is determined from the model output by assessing the percentage mass loss of adult birds over the chick-rearing period in the baseline simulations.

	Birds sighted	POOR metric	MODERATE metric	GOOD metric
Cumulative run 1	128.7	0.81	0.78	0.53
Cumulative run 2	133.3	0.83	0.80	0.56
Cumulative run 3	131.0	0.69	0.73	0.64
Cumulative run 4	134.3	0.67	0.54	0.55
Cumulative run 5	119.3	0.81	0.74	0.74
Cumulative run 6	120.3	0.91	0.72	0.69
Cumulative run 7	134.3	0.66	0.55	0.48
Cumulative run 8	135.3	0.68	0.58	0.47
Cumulative run 9	129.0	0.77	0.74	0.51
Cumulative run 10	127.0	0.78	0.87	0.62
Mean number of birds sighted over all 10 paired runs	129.3			
Mean value metric P2		0.76	0.70	0.58
SD value of metric P2		0.08	0.11	0.09

4.4 Summary of Example Results from Multiple Paired Runs

In terms of the overall impact of the ORDs on black-legged kittiwakes we focus on the main population level metric for adult birds, P1 (Table 4-6 to Table 4-9 and Figure 4-1 to Figure 4-3) derived from methods using local GPS data with flight removed (Method i; Table 4-1). A summary of results for chick mortality is presented in Figure 4-1. Across the range of simulated median prey densities, there are no detectable effects for birds from either Buchan Ness or Fowlsheugh when either only WFA is present, or when all three ORDs are present (Table 4-6 to Table 4-9). However, for birds from St Abbs Head, and particularly birds from Forth Islands, there are detectable effects on adult and chick mortality across the range of median prey densities simulated, and these effects increase particularly for birds from Forth Islands when all three ORDs are considered together (Figure 4-1 and Figure 4-2).

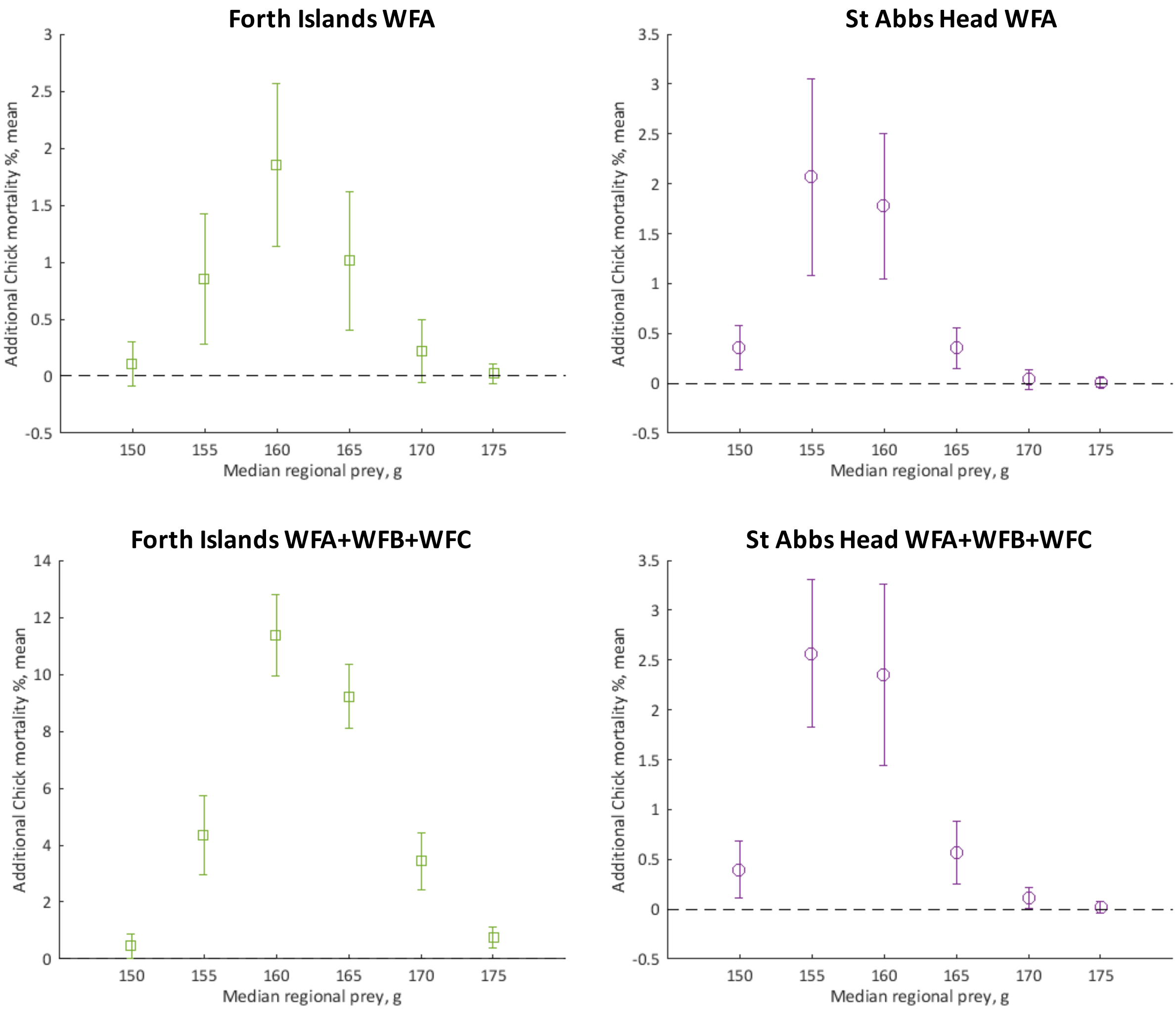


Figure 4-1: Estimated mean change in chick mortality with associated 95% prediction intervals for black-legged kittiwakes at Forth Islands (left panels) and St Abbs Head (right panels) over the range of six median prey densities simulated using mapped bird densities and prey availability from local GPS tracking data with flight removed (method i). Upper panel shows results for runs with one ORD (WFA); lower panel shows cumulative effects from all three ORDS (WFA+WFB+WFC).

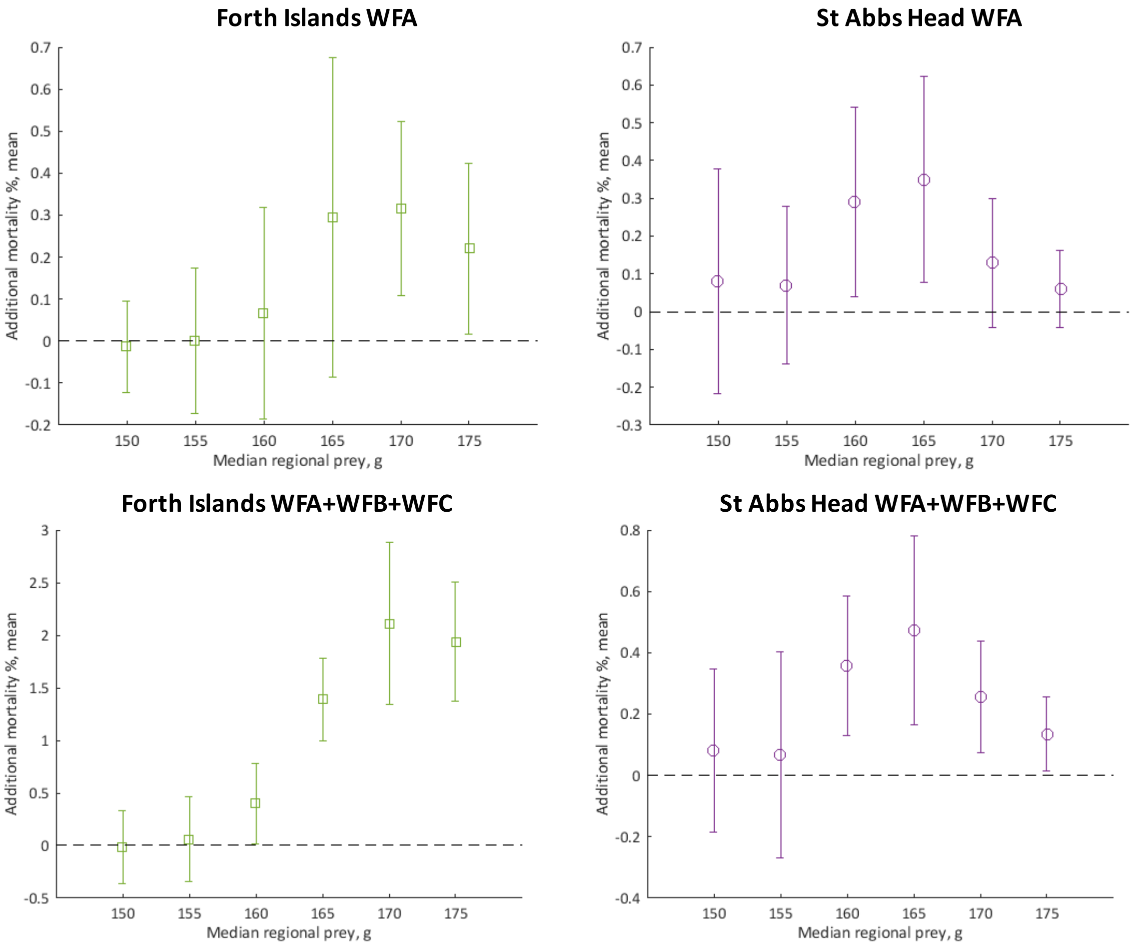


Figure 4-2: Estimated mean change in adult mortality with associated 95% prediction intervals for black-legged kittiwakes at Forth Islands (left panels) and St Abbs Head (right panels) across the range of six median prey densities simulated using mapped bird densities and prey availability from local GPS tracking data with flight removed (method i). Upper panel shows results for runs with one ORD (WFA); lower panel shows cumulative effects from all three ORDS (WFA+WFB+WFC).

4.5 Comparison of Predicted Model Output Using Different Bird and Prey Inputs

4.5.1 Bird and Prey Inputs Derived from Local Tracking Data Versus Distance-Decay Methods

We compared predicted effect sizes for the cumulative impact of all three fictional ORDs (WFA+WFB+WFC) under two sets of model inputs for bird density and prey availability across ‘moderate’ conditions for black-legged kittiwakes on the Forth Islands in the Forth-Tay region:

i. Local GPS tracking data with flight removed was used to estimate bird density and prey availability (Method i, Table 4-1).

ii. Distance-decay was used to estimate bird density and prey availability was assumed to be uniform over the study region (Method ii, Table 4-1).

When the cumulative impact of all three ORDs (WFA + WFB + WFC) was assessed under assumptions of no available GPS data (uniform prey and distance-decay bird densities), the population level effects on birds from Buchan Ness and Fowlsheugh did not differ substantially from those estimated using GPS-derived inputs. Few birds were affected (Buchan Ness: GPS<1%, distance-decay<1%; Fowlsheugh: GPS<1%, distance-decay<9%) and the majority of estimated impacts on affected birds were within the bounds of model uncertainty meaning that the ORDs may not have affected additional mortality at all (Table 4-6 and Table 4-7).

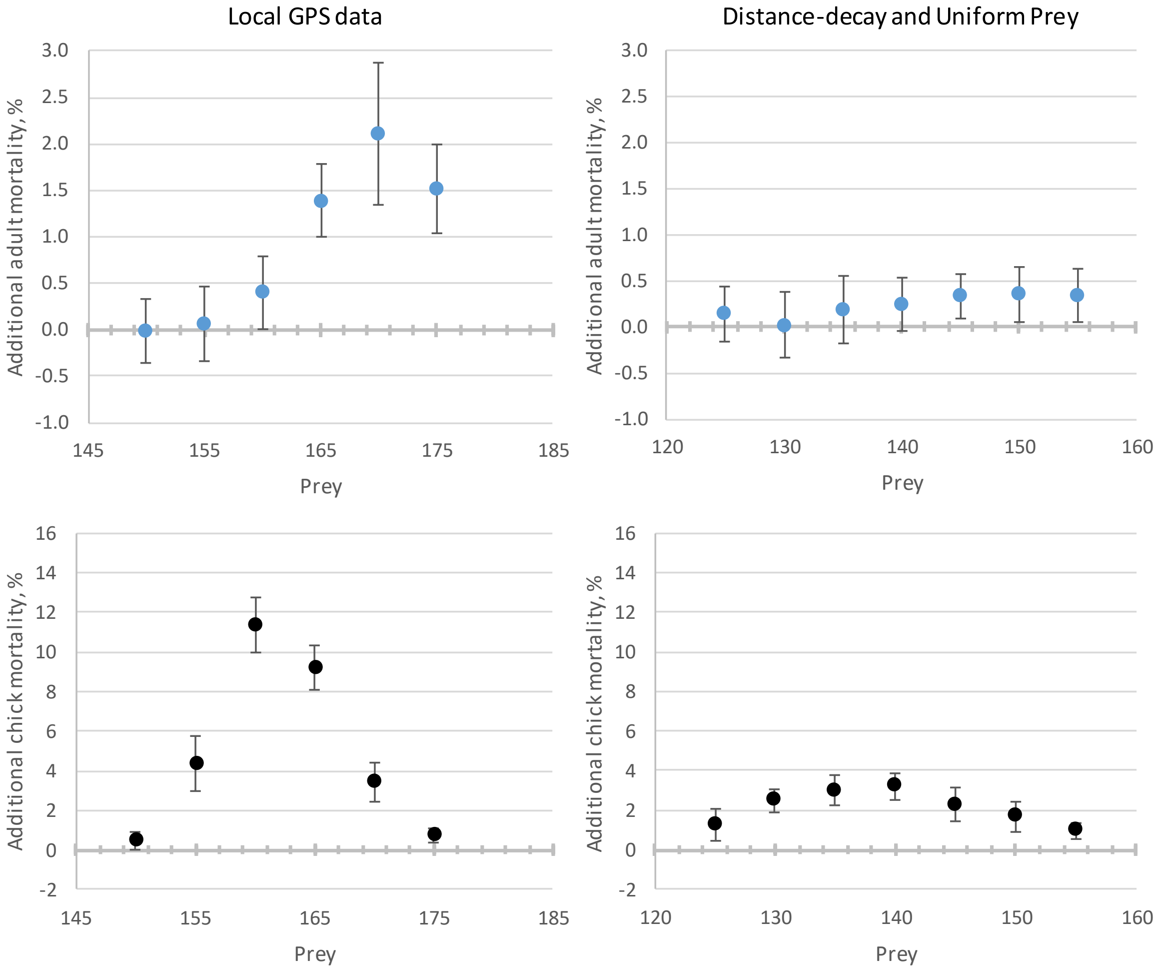
However, for birds from the Forth Islands, the population level effect of all three ORDs on adult and chick mortality was lower under assumptions of distance-decay and uniform prey when compared to the effect estimated using local GPS data (maximum mean additional adult mortality = 2.11% for GPS data versus 0.36% for distance-decay; Figure 4-3, Table 4-8). A similar number of adult birds were affected by the ORDs in both scenarios; under distance-decay an average of 1458.2 adult birds (38.7% of the populations) were directly affected by the ORDs, compared to 1507.6 adult birds (40.0% of the population) under GPS data (Table 4-12). However, the impact on additional adult mortality of these directly affected birds (birds that were displaced or experienced barrier effects at least once during the season) was considerably lower under distance-decay (maximum mean over all paired runs = 0.96%; Table 4-12) compared to GPS data (maximum mean over all paired runs = 5.33%; Table 4-12).

On average, fewer birds from the Forth Islands were displaced (I3 + I5; Table 4-16) under distance-decay (782.5 birds) compared to GPS data (1055.4 birds), and fewer birds experienced barrier effects (I4 + I5) under distance-decay (1429.3 birds) compared to GPS data (1507.4 birds). Noticeably fewer birds under distance-decay (approximately 753 adults) were displaced and experienced barrier effects (I5) compared to under GPS data (approximately 1054 birds), meaning that under distance-decay, when a bird was displaced it had a lower tendency to also experience barrier effects as a result of its new foraging location in the ORD buffer zones.

In addition, under distance decay, although birds tended to make more foraging trips per day (model output, not shown here), the additional average distance flown due to barrier effects was lower (over all colonies approximately 4.06 km; Forth Islands only approximately 25.1 km) compared to that under GPS data (over all colonies approximately 7.75 km; Forth Islands only approximately 38.9 km). This meant birds that experienced barrier effects (I4) or barrier effects and displacement (I5) suffered lower additional adult mortality under distance-decay (maximum mean over all paired runs I4=0.99%; maximum mean I5=1.09%; Table 4-16) compared to under GPS data (maximum mean I4=4.57%; maximum mean over all paired runs I5=5.66%; Table 4-16). This, combined with the very negligible impact on displaced only birds (I3) in both scenarios (Table 4-16) means that the overall ORD impact on both additional adult and chick mortality is lower under distance-decay than under GPS data (Figure 4-3).

In summary, under distance-decay, the bird density distribution will be much smoother, and may result in bird densities tending to be higher closer to the colony. In addition, displaced birds will always tend to be more frequently displaced into the colony-side area of the ORD buffer zone (thereby incurring no additional barrier effects) under distance-decay because new locations are selected in proportion to bird density, which necessarily declines with distance. In contrast, local GPS data will tend to create hotspots of bird density (and prey availability) that could, as in this instance, potentially result in more and stronger interactions with ORDs, depending on where ORDs are situated in relation to both the colony and these hotspots. However, the relative strength of impacts between distance-decay and GPS data will depend strongly upon the relative juxtapositions of bird and prey hotspots, ORD footprints and colony locations.

Figure 4-3: Comparison of estimated effect sizes for additional adult mortality (upper panels) and additional chick mortality (lower panels) for black-legged kittiwakes on the Forth Islands in the Forth-Tay region under assumptions of moderate conditions. Comparison in model output is made for bird density and prey availability inputs derived from local GPS tracking data with flight removed (left panels) versus bird density derived from distance-decay methods and an assumption of uniform prey (right panels). Note that regional median prey levels ('Prey', x-axis) are different under the two scenarios because although under uniform prey (distance-decay) birds will encounter the median value across all foraging locations, under patchy prey (GPS derived prey) the actual prey that birds encounter will vary considerably due to hotspots in prey availability and bird densities. The precise nature of the patchiness in bird and prey densities encountered under GPS derived inputs will, in most cases, therefore, require the median regional prey density to be adjusted from that used in distance-decay to result in the same baseline conditions for adult mass loss and productivity.



4.5.2 Model Output Using GPS Data with Flight Included Versus with Flight Removed

Whilst local GPS tracking data represents the ‘gold standard’ for estimating bird densities (and prey availability where no empirical data on prey is available), the method in which these data are analysed to derive estimated foraging densities is likely to have a profound impact on effect sizes (see Section 3.11.1). This is particularly the case in terms of whether the statistical analysis has included, or removed flight locations from GPS tracking data prior to estimating bird densities, but is also important in terms of whether non-flight fixes have been partitioned into foraging and resting, whose distributions may differ. This leads to a hierarchy in the relative defensibility of inputs required by SeabORD, whereby the nature of the empirical data and the statistical methodologies and assumptions employed will inevitably affect the estimates of ORD impacts from the SeabORD model – see later in this Section and Discussion for full details (Table 5-1).

Critically, SeabORD assumes that the bird density maps used within the model represent foraging locations, rather than utilisation distributions in which no separation of behavioural states has been made. During the breeding season when birds act as centrally placed foragers, it will generally be the case that including flight locations in GPS data used to estimate bird densities will result in a higher estimated density of birds close to the colony than when flight is removed because of the effect of birds commuting back and forth between foraging locations and the colony. An effect of this sort is demonstrated below (Figure 4-4) where we estimate the difference in predicted black-legged kittiwake density in the Forth-Tay region using Method iii (local GPS data with flight included) and Method iv (local GPS data with flight removed using a speed threshold). This demonstrates, that for this set of GPS locations, there is a tendency for greater bird densities estimated close to the colony (Isle of May) when flight is not removed (Figure 4-4: negative values shown in green and blues); and a tendency for greater bird densities at further distances from the colony (Isle of May) when flight is removed (Figure 4-4: positive values show in lighter yellows). However, there is considerable spatial variation in the difference in the relative estimated densities using these two methods, which will mean that deciphering the effect of this methodological decision on model output is by no means as straightforward as a tendency towards increasing or decreasing ORD effects. This is particularly true because the change in model output will depend heavily upon the relative juxtapositions of the colony, ORD, foraging range of the species, and any bird or prey hotspots predicted using the different methodologies, and the relative importance of barrier effects and displacement. Further examples could be considered in future work where foraging and resting are partitioned, because the distribution of these two activities may vary.

We explored the effects of including or removing flight from local GPS tracking data for black-legged kittiwakes from the Isle of May in the Forth Tay region and three fictional ORDs (WFA+WFB+WFC) by comparing SeabORD output across multiple ‘moderate’ runs using bird and prey maps derived from local GPS data with flight included (Method iii) to output from runs where maps were derived from the same local GPS data but with flight removed using a speed threshold (Method iv).

For each method we initially conducted five sets of ten paired runs covering the range of ‘moderate’ conditions (Table 4-12: four ‘moderate’ and one ‘good’). These results demonstrate that although slightly more birds were directly affected by the ORDs under Method iv (Table 4-12, flight removed from GPS: 40.4%) than under Method iii (Table 4-12, flight included in GPS: 40.1%), the subsequent predicted additional mortality over the moderate range was on average slightly lower when flight was removed from GPS data (Table 4-12: flight included = maximum mean impact of 4.02% versus flight removed = maximum mean impact = 3.46%).

Of directly impacted birds, slightly more birds were affected by barrier (I4) or barrier and displacement (I5) effects using Method iv (flight removed); with an average of 1507 birds in those categories under Method iv (flight removed) compared to an average of 1496 birds in those categories under method iii when flight was not removed (Table ***4-16***; I4 and I5). This means when flight was removed from GPS there was a shift towards more birds affected by barrier effects (Table ***4-16***, I4 + I5).

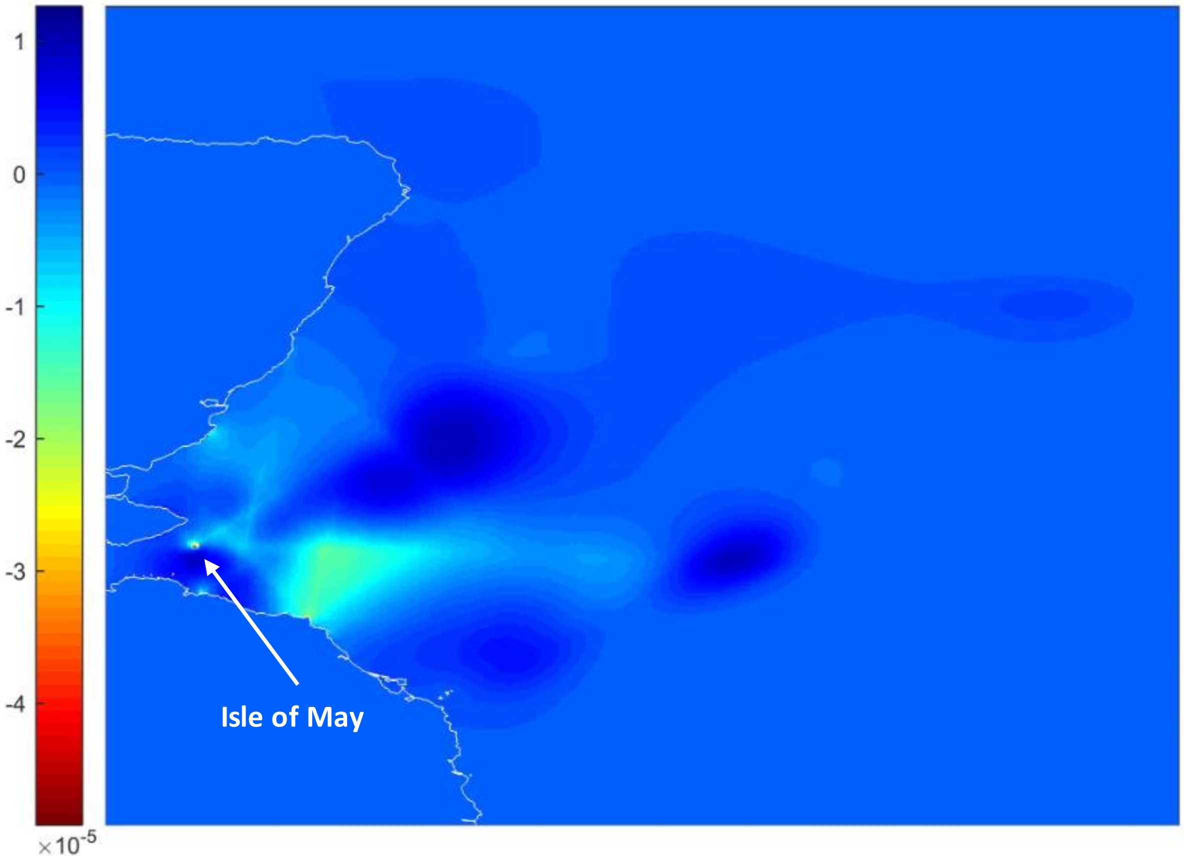


Figure 4-4: Difference in predicted bird density between when bird density is estimated from GPS tracking data with flight included versus when it is estimated from tracking data with flight removed using a speed threshold. Example pertains to local GPS tracking data for black-legged kittiwakes in the Forth-Tay region foraging from the Isle of May. Positive areas (darker blue) indicate higher predicted bird density when flight was removed from GPS data; negative areas (green-yellow-red) indicate higher predicted bird density when flight was included in GPS data.

with fewer birds being affected by displacement (Table ***4-16***, I3 + I5). The overall impact of this on additional mortality is a result of the complex behavioural and energetic decisions within the SeabORD model. We can see from the model output for each scenario that the overall additional adult mortality for barrier affected birds (I4) and barrier and displacement affected birds (I5) was slightly lower when flight was removed (Table ***4-16***; maximum mean effect of 4.37% [I4] and 3.91% [I5]) compared to when flight was included (Table ***4-16***; maximum mean effect of 3.00% [I4] and 3.63% [I5]). This is because birds travelling to foraging locations further away from the colony will tend to complete fewer foraging trips per day because of behavioural routines within the model where individuals attempt to minimise time away from the nest whilst meeting energetic demands. Therefore, in this case study, the average additional distance travelled due to barrier effects was lower when flight was removed from GPS data because birds were on average completing fewer trips (maximum additional distance flown: mean = 49.9km, SD = 4.7km) compared to birds in the scenario where flight was not removed from GPS data (maximum additional distance flown: mean = 71.3km, SD = 5.1km).

In summary, these results show a slight overall decrease in population level additional adult and chick mortality when method iv (flight removed) is compared to Method iii (flight included) in this specific scenario across the range of moderate prey conditions (Figure 4-5). This translates into a slightly lower overall single estimate for additional mortality under Method iv (flight removed) compared to Method iii (flight included) in this instance (Figure 4-5). However, we stress these findings are entirely particular to the specific case study presented here, and cannot be extrapolated to other situations (species, GPS data, ORDs etc), or indeed be expected to hold true for this species in the modelled region due to the specific juxtaposition of the simulated ORDs and the colony.

We would usually expect including flying and resting at sea locations in GPS tracking data to decrease the mean distance of estimated bird locations from the colony. This in turn, would usually be expected to lead to a reduction in the total proportion of birds that lie within or beyond an ORD footprint (depending on the juxtaposition of the colony and the ORD). A decrease in the total proportion of birds that lie within or beyond an ORD footprint would be expected to lead to a decrease in the estimated impact of the ORD on additional mortality.

However, importantly, this change to additional mortality results from a set of complex interactions between the relative extent of displacement and barrier effects upon the population and the behavioural and energetic consequences of these effects upon individuals, pairs and their offspring. It is, therefore, not inevitable that removing flight and resting at sea locations from GPS data would certainly lead to an increase in barrier and displacement affected birds – this is because the spatial distributions of flying and resting at sea locations in the GPS may also differ from the foraging locations in terms of features other than simply the mean distance to colony, and because when mean distance travelled by birds within SeabORD increases (which will generally be the effect of removing flying locations) then this will impact upon the number of trips that birds decide to undertake within the model, and this will in turn alter the simulated effect of the ORD.

Similarly, an increase in the proportion of bird locations beyond the ORD footprint would increase barrier effects, but could simultaneously decrease displacement effects (if fewer bird locations are predicted within the ORD footprint itself). The relative importance of displacement and barrier effects within SeabORD is affected by multiple behavioural and energetic decisions, so it is difficult to say precisely how the impact of an ORD upon additional mortality would change in such a situation; there is no reason to expect it to always increase (or decrease).

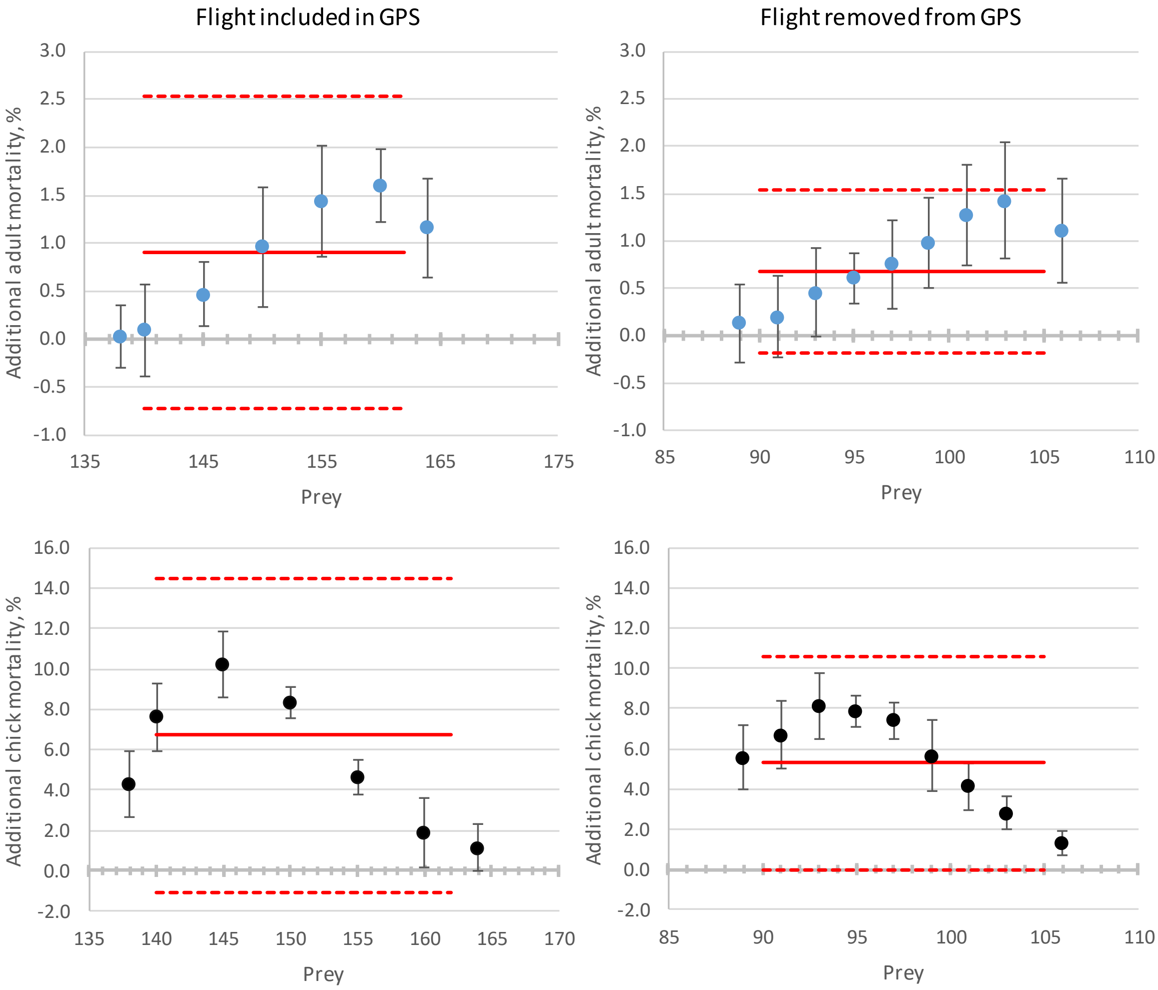


Figure 4-5: Summary of the predicted additional adult (top panels) and chick (lower panels) mortality in black-legged kittiwakes on the Isle of May as a consequence of three ORDs. Results are shown for model inputs for bird and prey densities created using method iii (left panels; flight included in GPS data) and method iv (right panels; flight removed from GPS data using speed threshold). On each panel a single mean estimate under moderate conditions for additional adult and chick mortality and associated 95% prediction intervals are shown by the solid and dashed red lines (see Section 0 for methodology). The horizontal length of the red line indicates the median regional prey range over which stratified random sampling was performed using this method. Dots and error bars represent the mean effect and associated 95% prediction intervals at a given median regional prey availability averaged over ten matched pairs. Note that on each panel the rightmost dot and error bars represents ORD impacts under ‘good’ conditions.

4.6 Averaging Over Prey Values to Estimate a Single Effect Size for an ORD

The results in Section 4.1 serve to illustrate how effect sizes of ORDs vary in relation to specified median regional prey values used in simulations. The approach to uncertainty quantification that we outlined in Section 3.11 involves using multiple model runs to capture the uncertainty associated with knowing the level of prey under a capture scenario (e.g. “moderate” prey).

Here, we present results from applying this method to black-legged kittiwakes in the Forth-Tay region for the combined effects of three fictional ORDs, focusing on birds from the Forth Islands using bird density and prey maps derived from older GPS tracking data (Method i; WFA+WFB+WFC; for full details see Worked Example).

Initial single baseline runs using 10% of the total black-legged kittiwake population in the region identified the lower and upper boundaries for moderate conditions as median regional prey values lying within 157-172. Stratified random sampling was then used to select ten regional prey levels lying within this range (Table 5-1).

Table 4-19: Median regional prey values selected using stratified random sampling across the range of prey values identified to correspond to ‘moderate’ conditions for black-legged kittiwakes in the Forth-Tay region.

Run	Median Regional Prey, g per unit volume
1	158
2	159
3	161
4	162
5	164
6	165
7	166
8	168
9	169
10	171

These ten regional prey values produced, on average, approximately 7.8% mass loss in adults over the chick-rearing period for black-legged kittiwakes at the Forth Islands (Table 4-20).

Table 4-20: Summary of average percent mass loss in adult black-legged kittiwakes breeding in the Forth-Tay region for the range of ten regional prey values selected in this example.

Colony	Season Type	% adult mass loss
Buchan Ness	Good	0.11
Fowlsheugh	Good	1.24
Forth Islands	Moderate	7.76
St Abbs Head	Moderate	6.48

The mean effect on additional adult mortality (P1) for all three ORDs in combination was 1.25% with a 95% prediction interval of -0.38% to 2.88%. The mean effect on additional nest mortality was 8.6% with a 95% prediction interval of 0.03% to 17.23%. These results for a single metric for P1 for both adults and chicks are in line with those produced over the range of multiple paired 'moderate' runs using the same inputs and parameters documented in Section 4.1 (Table 4-8). For detailed results for all metrics (P3-P6) please refer to the accompanying Worked Example.

4.7 Additional Model Output

SeabORD produces additional model output that may be used to 'sense-check' results and provide additional insight into the effects and processes contributing to ORD impacts (Table 4-21).

Output for 'adult survival at the end of the breeding season' in baseline (no ORD) and ORD runs (with ORD) is provided, note that this differs from that used in final model metric which relates to adult survival over the entire year. This serves as a useful sense check that prey levels are not set too low so as to cause mortality of adults during chick-rearing, a phenomenon very rarely seen in most UK seabird species.

Mean and standard deviations for initial and final adult body mass (over chick-rearing) both without (no ORD) and with (with ORD) ORDs are provided to add more detail in terms of the physiological effects of ORDs, and to allow users to see the average mass loss in adults over chick-rearing which forms a key component of the subsequent conversion into adult survival over the entire year.

Differences between average total distances flown and average total number of foraging trips with and without ORDs over the course of the chick-rearing period are outputted (mean and SD) to provide users with more insight into the extent of barrier effects and how additional flight costs affect optimisation decisions for the number of foraging trips performed by birds.

Finally, output for the mean and standard deviation for the number of chicks not surviving the breeding season with (with ORD) and without (no ORD) ORDs is shown to provide users with the model outputs used to determine chick survival and additional mortality calculations in final model metrics.

Table 4-21: Additional model output from SeabORD.

Colony	Buchan Ness	Fowlsheugh	Forth Islands	St Abbs
Number of adult birds in group	12542	9388	3766	4314
Adult survival at end of breeding season, %, mean (no ORD)	100.0	100.0	100.0	100.0
Adult survival at end of breeding season, %, stdev (no ORD)	0.00	0.00	0.00	0000
Initial adult body mass, g, mean (no ORD)	372.65	371.80	372.91	372.14
Initial adult body mass, g, stdev (no ORD)	0.00	0.00	0.00	0.00
Final adult body mass, g, mean (no ORD)	372.24	367.20	343.96	348.02
Final adult body mass, g, stdev (no ORD)	0.09	0.75	5.97	6.17
Adult survival at end of breeding season, %, mean (with ORD)	100.00	100.00	100.00	100.00
Adult survival at end of breeding season, %, stdev (with ORD)	0.00	0.00	0.00	0.00
Initial adult body mass, g, mean (with ORD)	372.66	371.80	372.91	372.14
Initial adult body mass, g, stdev (with ORD)	0.00	0.00	0.00	0.00
Final adult body mass, g, mean (with ORD)	372.24	367.14	342.12	347.46
Final adult body mass, g, stdev (with ORD)	0.09	0.84	4.85	6.17
Difference between total distances flown with and without ORD, km, mean	3.55	8.78	1.28	9.03
Difference between total distances flown with and without ORD, km, stdev	11.24	27.81	32.47	10.48
Difference between total number of trips flown with and without ORD, mean	-0.005	-0.011	-1.556	-0.180
Difference between total number of trips flown with and without ORD, stdev	0.015	0.033	0.610	0.143
Chicks not surviving the season, mean (no ORD)	304	245	472	319
Chicks not surviving the season, stdev (no ORD)	0.00	0.00	391.8	230.6
Chicks not surviving the season, mean (with ORD)	304	245	635	344
Chicks not surviving the season, stdev (with ORD)	0.00	0.00	436.6	258.1

5 Discussion

5.1 Estimating Displacement and Barrier Effects

The analytical tool developed in this project offers a user-friendly framework for assessing the impacts of ORDs on breeding seabirds, estimating the fate of individual birds, and translating these effects into policy-targeted metrics at the population level to improve precision of assessment and facilitate the transition of Scotland's energy sector to a more environmentally sustainable suite of platforms.

A key development has been the integration of methodology from a number of previous Scottish Government projects to produce a method capable of estimating the change in population level mortality of adults and chicks in relation to the number of birds observed in at-sea surveys of ORD footprints – the typical pre-construction method for providing empirical data to inform predictions of ORD impacts on birds breeding at nearby SPAs.

In all, the improvements to the previous simulation model (Searle et al. 2014), including better biological plausibility and estimation of bird habitat use, combined with the new policy-relevant metrics for assessing ORD effects on mortality and productivity, represent a comprehensive framework that will reduce current levels of uncertainty in assessing the effects of ORDs in UK coastal waters.

As with previous research, however, there remain important caveats associated with the estimated effects arising from this work. Notably there is still a lack of empirical data on how affected birds will navigate around ORDs to gain access to foraging grounds and in how birds displaced from ORD footprints will select new foraging locations. There is also, as yet, no empirical data available to assess how affected birds may habituate to ORDs over time, thereby potentially reducing the impacts of developments as birds become accustomed to their presence. Finally, bird distributions, such as those considered here using local GPS data offer state-of-the-art estimation of breeding bird habitat use based on all available tracking data, but bird habitat use varies considerably over space and time. Therefore, the precise interaction of ORDs with the foraging and flight patterns of breeding birds (and their prey) in any one season will inevitably vary from model predictions, and remains a key driver of ORD impacts on SPA populations.

5.2 Metrics for Assessing ORD Effects and Relevance to Policy

A key development in this work has been the development of a range of metrics for assessing impacts of ORDs in breeding seabirds. These metrics are all estimated using the same underlying model, and the tool provides outputs for each of them. However, they have different interpretations and can be used for different purposes.

5.2.1 Population-Level Metrics

These metrics quantify the impact of the ORD upon the mortality rate (of either adults or chicks) across the colony population in question. We output two population-level metrics:

P1: the original metric, used in Searle et al. (2014), in which we look at the change in mortality rate (ORD minus baseline) as a percentage of the total population size.

P2: a novel metric, introduced here, in which we look at the change in mortality rate (ORD minus baseline) as a ratio to the number of birds seen within a "snapshot" survey of the ORD footprint.

P1 combines information on the *frequency* and *demographic consequences* of displacement/barrier effects within a simple metric. In contrast, metric P2 separates out the *demographic consequences* of displacement/barrier effects from the *frequency* with which these effects occur.

The former metric (P1) is of direct policy relevance, and formed the key output from the assessment within Searle et al. (2014). The latter metric (P2) provides a mechanism for relating the effects of the ORD upon mortality to data on the number of birds within the footprint at a particular "snapshot" in time (as are typically collected using boat-based or aerial spatial surveys within pre-construction ORD footprint assessments).

5.2.2 Individual-Level Metrics

These metrics quantify the impact of the ORD upon the mortality rate (for either adults or chicks) of individual birds. The metrics use the same formula as for P1, but apply this formula to subsets of the population rather than to the entire population. These subsets are determined based on the frequency with which individuals interact with the ORD, and the nature of these interactions (barrier or displacement effects). These individual-level metrics allow us to understand the nature of the displacement and barrier effects. However, we do not consider them to be of direct policy relevance, because in practice it will not be possible to know the number of birds that belong to any of the groupings considered here. The population-level metrics (P1 and P2) may be of greater policy relevance, because they allow the effects of ORDs to be derived from data (either, in P1 from GPS data, or, in P2 from a combination of GPS data and survey data).

5.2.3 Development of a Single Estimate for ORD Effects

We have developed a method that uses stratified random sampling to estimate effect sizes across the range of ‘moderate’ conditions experienced by birds in the baseline (no ORDs present) and generate an average for each model metric (P1-P6). This method has the advantage that it incorporates uncertainty in model outputs deriving from uncertainty in prey levels, to produce both an overall mean estimate for each metric and a corresponding 95% prediction interval that includes prey uncertainty. The method involves the following steps, available within SeabORD, detailed in the accompanying ‘Worked Example’:

- Identify the median regional prey levels corresponding to the upper and lower boundaries of ‘moderate’ conditions based on adult mass change and chick/nest survival in baseline runs for each species and colony of interest (Table 3-4).
- Simulate *n* prey levels randomly from within this range (using stratified random sampling, with *n* strata of equal width and one prey value simulated per stratum) and run a set of *n* paired simulations (one paired simulation for each of the *n* prey levels) to produce *n* estimates for each metric with associated standard deviations and 95% prediction intervals.
- Combine the *n* estimates for each metric to produce a final, single mean estimate with associated 95% prediction intervals incorporating model uncertainty and uncertainty derived from prey levels.

We recommend the use of as high a percentage of the population and as many replicate pairs as computationally feasible when calculating the single, averaged value for each metric (i1-i6). In particular, it is important to assess the relative width of the peak impact over the range of moderate conditions, so as to ensure enough stratified random runs are conducted over the moderate prey range to capture this peak impact.

5.3 Developments Since Searle et al. 2014

The underlying model has refined and improved that used in Searle et al. (2014) in a number of ways addressing both ‘usability’ and biological realism. The developments that are of most relevance to usability are:

i. The running time for the model has been made much faster through improvements to the underlying model structure, efficiency of code, and through the conversion of the model from R to Matlab. This makes the model much more feasible to use in practice, and makes it possible to run larger numbers of scenarios. In general, although this will vary by species and location, it takes approximately 24 hours to complete a set of ten matched pairs (baseline + ORDs) over one median prey density using 50% of the bird population.

ii. The previous model (Searle et al. 2014) only included one option for specifying the route that birds will take to avoid an ORD (“barrier effects”) – that option assumed that birds did not begin to avoid the ORD until they reach the edge of the footprint, and that they returned to their original route only once they intersected with the straight line path they would have taken if the ORD were not in place. The model now includes a second option, in which birds modify their route in advance (assuming a level of spatial memory) to choose the shortest possible route between the colony and foraging location that avoids the ORD footprint. It is not clear, due to a lack of empirical evidence, which of these options is more plausible, but by including both options in the model we allow the user to explore the impact of varying these assumptions.

iii. A link to at-sea survey data is included.

iv. Impacts of ORDs may now be assessed for different categories of birds in terms of how often they interact with an ORD.

v. The model runs at a finer spatial resolution improving the accuracy of determining which birds are affected by ORDs.

vi. The model has been developed into a tool with a user-friendly interface and a range of user-specified options to define bird behaviour and assess ORD effects.

vii. The model provides 95% prediction intervals capturing uncertainty in the overall level of prey, together with stochastic inherent uncertainty between populations of birds.

viii. The model provides a method for generating a single estimate of an ORD impact across the range of ‘moderate’ conditions, capturing the uncertainty involved in estimating the overall level of prey that is actually associated with such conditions.

Importantly, the biological realism within the underlying simulation model has been improved as follows:

i. Re-parameterisation of adult mass loss in line with published studies to provide a more robust estimate of adult mass loss in relation to food acquisition and energy expenditure.

ii. Re-parameterisation of chick growth in line with published studies to provide a more robust estimate for changes in chick mass in relation to provisioning by parents.

iii. Re-parameterisation of the effect of intra-specific competition on intake rate to have fewer unknown parameters and to be insensitive to the proportion of the population included in simulations, allowing model runs to be performed with only a proportion of the population.

iv. Provision of 95% prediction intervals on metrics for assessing ORD impacts.

v. Matching of paired baseline and ORD runs to reduce stochasticity affecting ORD impacts.

vi. Simplification of mechanisms to reduce the number of unknown parameters within the model.

vii. An optimisation routine whereby birds select the optimal number of trips to make each day based on optimal foraging decisions.

5.4 Input bird and Prey Distribution Data

Within the modelling it is necessary to separately specify both a density of birds, and a map of prey, as inputs to the model, because these determine separate things: (a) the spatial distribution of (foraging) locations that birds visit, and (b) the levels of prey they encounter when they reach these locations. Often, there will be no relevant empirical data on prey distributions for the species in the geographical areas of interest, so it is necessary to make some assumptions about the spatial distribution of prey. Within this project, following the approach taken in the earlier project (Searle et al., 2014), we consider two assumptions regarding the spatial distribution of prey:

1. that the spatial distribution of prey is proportional to the spatial distribution of bird (foraging) locations, after the latter is adjusted to remove the effects of accessibility (distance from source colony) and competition (distance from next nearest colony);
2. that prey is uniformly distributed in space.

The analysis of Searle et al. (2014) used local GPS data collected only at the colonies of interest, with flight locations removed from the dataset using a speed threshold. Ideally, local GPS data should be used to provide the spatial input (bird density maps) for the analytical tool; however, these data, are not always available. Therefore, within the analytical tool we have included an option that allow the user to estimate ORD effects in areas lacking in GPS tracking data. Under this option (‘distance-decay’) users can tell the model to assume that bird density decays exponentially with distance from colony. This is a simplistic approach, and so should only be used in situations where local GPS data are unavailable. However, some defence for the use of this approach where a species-specific estimate of rate of distance decay is available lies in the fact that predictions from models containing only “distance to colony” effects fit observed GPS data well (Wakefield et al., 2017).

The example presented in this report focusing on black-legged kittiwakes demonstrates that using the distance-decay bird density method with uniform prey can reduce the estimated impact of ORDs in comparison to the effects estimated using local GPS tracking data. The exact nature of the change in the barrier effect between distance-decay inputs and GPS-derived inputs will depend very much upon the juxtaposition of the ORDs and the ‘hotspots’ of bird densities in the GPS-derived inputs.

Overall, we consider the use of local GPS-derived inputs for bird and prey density within SeabORD to provide more reliable estimates of the impacts of ORDs, and users should bear in mind that the ‘real’ impact of an ORD (assuming GPS-derived inputs capture the true bird foraging and prey distributions) may be considerably larger than that estimated under assumptions of distance-decay and uniform prey.

However, the method in which tracking data are analysed to derive estimated foraging densities may have an impact on effect sizes. This is particularly the case in terms of whether the statistical analysis has included, or removed flight locations from GPS tracking data prior to estimating bird densities, but is also important in terms of whether non-flight fixes have been partitioned into foraging and resting, whose distributions may differ. Table ***5-1*** provides a summary of potential data and methodologies for estimating bird density for use in SeabORD, ranked by their relative defensibility.

However, importantly, any change to additional mortality estimated by SeabORD resulting from different methodologies used to derive bird foraging locations from GPS tracking data arises from a set of complex interactions between the relative extent of displacement and barrier effects upon the population and the behavioural and energetic consequences of these effects upon individuals, pairs and their offspring. The relative importance of displacement and barrier effects within SeabORD is affected by multiple behavioural and energetic decisions, so it is difficult to say precisely how the impact of an ORD upon additional mortality would change in such a situation; there is no reason to expect it to always increase (or decrease).

Table 5-1: Summary of potential data and methodologies for estimating bird density for use in the SeabORD model, ranked by their relative defensibility (one being the highest level of defensibility).

Method	**Data requirements**	**Relative defensibility**
Modelling of local GPS data with behaviour inferred from direct observation on at least a subset of individuals (*sensu* Browning et al. 2017)	Local GPS tracking data coupled with activity loggers on at least a subset of the same individuals (e.g., TDRs, accelerometers)	1
Modelling of local GPS data with behaviour inferred using a statistical model (e.g., Hidden Markov Models)	Local GPS tracking data of high frequency	2
Modelling of local GPS data with behaviour determined using threshold methods (Searle et al. 2014)	Local GPS tracking data of high frequency and published threshold value (e.g., speed threshold for identifying flight; tortuosity threshold for identifying foraging)	3
Modelling of local GPS data where it is not possible to infer behaviour	Local GPS tracking data of lower frequency, or local ARGOS data	4
Regional habitat association modelling of GPS data to produce predicted utilisation distributions at colonies without GPS data (*sensu* Wakefield et al. 2017)	Regional GPS data and habitat covariates	5
Simple distance-decay methods where decay parameter is estimated from regional level GPS data	Regional GPS data	6
Simple distance-decay methods where decay parameter is taken from species level published foraging ranges	Published value for foraging range	7

5.5 Value of GPS Tracking Data Versus at-Sea Data

To understand the population consequences of ORDs on SPA seabird populations, it is necessary to address three questions:

i. Is there a measurable effect of ORDs on individual birds i.e., is there evidence for collisions, displacement, barrier effects or other factors?
ii. Are the affected individuals likely to be in any way now or in the future (through for example recruitment) linked to the SPA?
iii. Are there consequences of these effects on SPA populations?

The most appropriate method to address these questions in the context of displacement and barrier effects is to use GPS tracking data because it is possible to directly quantify the connectivity to SPAs and population level consequences of these developments, especially if tracking can be achieved by deploying GPS loggers on the same individuals before and after construction, which substantially enhances power. In contrast, at-sea bird surveys can only provide indirect evidence for displacement or barrier effects, based on the assumption that differences in densities of birds are due to the development and not to other factors such as prey availability. In addition, at-sea surveys do not provide any information on either the origin of individuals or the population level consequences of these effects at relevant SPAs.

Several studies have attempted to measure displacement of seabirds by ORDs using GPS tracking studies. Thaxter et al (2015) used tracking data to quantifying overlap between at-sea distributions of lesser black-backed gulls and offshore wind farms. Garthe et al. (2016) report on a preliminary tracking study of breeding gannets from Helgoland that demonstrated the potential to quantify the extent to which individuals avoid operational ORDs within the foraging range from the colony. As with barrier effects, GPS tracking of individuals of known breeding status and provenance, whose demography and physiologically can be monitored at colonies, opens up the potential to quantify the population level consequences of these developments. In contrast, at-sea bird surveys can only provide indirect evidence for displacement (or indeed barrier effects), and do not provide any information on either the provenance of individuals or the population level consequences of these effects at relevant SPAs. Substantially enhanced power to detect effects of displacement and barrier effects on demographic rates using tracking can be achieved by deploying GPS loggers on the same individuals before and after construction.

Quantifying displacement is more challenging than barrier effects because the tracks of displaced individual birds may not reveal that they had an initial preference for the ORD footprint and were displaced from it. This is because birds may fly directly to the displacement location such that their GPS track would not exhibit any detectable deviation away from the ORD location. Therefore, the GPS tracks of displaced individuals are hard to distinguish from those of individuals that instead had an initial preference for the location to which other birds were displaced. An approach that has frequently been employed when analysing bird survey or tracking data is to assume that changes in the spatial distribution of individuals between the pre-construction and post-construction periods have been caused by the marine renewables development (NIRAS/DHI 2018). However, changes in the spatial distribution of species may have occurred due to other factors such as change in spatial distribution of prey. There are two alternative approaches that attempt to address this problem:

i. Before-After-Control-Impact (BACI) analysis, using another colony as a control; (note that, in practice, it may be challenging to find an appropriate control colony);

ii. Gradient analysis at a single colony, where the change in bird distribution pre- and post-construction is quantified in relation to distance to the ORD, thereby attributing changes in bird spatial distribution to the ORD effect by way of its proximity to different distance bands of foraging locations.

5.6 Limitations of Tool

The tool only captures the chick-rearing period of the breeding season. There is potential for important displacement and barrier effects at other times of the year (e.g., incubation, pre-breeding, and non-breeding seasons), but assessing these effects would require a different set of behavioural rules appropriate to each phase, in particular the extent and frequency with which adults return from foraging to a central place, as well as offspring demands. However, in theory it is possible to adapt the model to be appropriate during periods other than chick-rearing, and to combine effects to obtain a year-round assessment, though it is important to note that empirical data on foraging energetics to parameterise the model are scarcer outwith the chick-rearing period.

The tool defines colonies as point locations and does not have a specific option to instead input colonies as linear features (e.g., where breeding birds are distributed along a stretch of the mainland coast). However, linear features can be readily included by converting linear colonies into a series of segments of equal length and assigning each segment a point location, as is commonly done with count data in the Seabird Monitoring Programme.

There is considerable interest in understanding the population-level consequences of offshore tidal devices. As with wind farms, birds can be displaced from sites or they may act as a barrier to movement of birds, particularly the diving of birds whilst foraging. The model is not currently set-up for behavioural changes with diving depth. However, the model could be adapted so that the geometric changes to flight paths and foraging locations that individuals are displaced to in a wind farm context are translated into equivalent changes with depth for diving species faced with a submerged tidal array. One key consideration is that tidal developments are typically much smaller than offshore wind farms and, therefore, the appropriate grid cell size for the model would need to be smaller to properly capture effects. This is feasible within SeabORD,;however, it has associated consequences for computing time and the spatial and temporal resolution of empirical data needed to parameterise the model. A key consideration would be the spatial scale at which behavioural responses occur. If barrier effects during diving occur at small scales with depth, then increased swimming distances would arise that in principal would be similar in structure to barrier flight effects around wind farms. Alternatively, individuals that wish to utilise foraging locations beyond the tidal turbine may simply fly over the site and then undertake typical diving behaviour at the chosen foraging location, thereby not incurring additional costs. Clearly, empirical data on the behavioural responses of birds around tidal turbines are required to inform how a displacement model should be parameterised.

We have not included foraging site fidelity as a mechanism in this model, although its impact on assessments was considered in a previous project (At-sea turnover of breeding birds, Searle et al. 2015). Data to accurately specify the particular intensity and spatial scale of foraging site fidelity in these species is lacking, and will clearly be very contingent upon local conditions. Similarly, the influence of site fidelity on assessments of ORDs will depend heavily upon the accuracy of bird density distributions derived from GPS tracking data or other sources. As such, we felt insufficient data were available to properly parameterise the model to account for foraging site fidelity effectively. If simulated birds showed site fidelity to specific foraging locations the impact of ORDs would disproportionately affect those birds whose favoured foraging sites were in the vicinity of the ORD. However without more extensive data on the prevalence (i.e., is there individual variation?), frequency (how long does fidelity to a chosen location persist and under what conditions?) and spatial scale of site fidelity it is difficult to say how these behaviours would affect population level assessments of ORDs.

Within this project it has not been possible to comprehensively assess the sensitivity of model output to parameter specification, or to perform in depth optimisation procedures for unknown parameters, such as those involved in the specification of the functional response. Whilst we have conducted quality control and model testing within the scope of this project, it has been impossible to comprehensively test the tool across all possible scenarios and locations due to time and resource constraints. Therefore it is possible that users may encounter model errors when using SeabORD (that we have been unable to anticipate or fully guard against), which will have no accompanying user friendly error message. This is most likely to occur during the route-finding stage in areas with a complex coastline or when using 'distance-decay' methods. Errors are more likely in these scenarios because the routine could randomly select a cell that cannot be reached using the flight rules (e.g. because the bird would have to cross land, which is not allowed in the model)." However, all model output, as part of the calibration process to identify 'moderate' conditions with no ORDs present, has been compared against empirical data for adult mass change, chick growth, and productivity, and output from the functional response parameterisation has been validated against empirical data for time-activity budgets specifying time spent foraging, time spent flying, and the number of trips made per day.

5.7 Further Research Requirements

There are a number of future research requirements that are necessary to improve the precision of effects of displacement and barrier effects from ORDs on seabirds (see also Appendix A). There is a lack of understanding on the rates of displacement and barrier effects, which appear to be context dependent, varying within species among locations. Mechanisms that may underpin this variation, such as local geography, prevailing wind directions, prey distribution and responses of different prey types to ORDs, is also poorly understood. Furthermore, there is a limited understanding of behavioural responses of individual seabirds to wind farms, including the flight paths they take in attaining destinations beyond the wind farm, and the decisions they make on alternative foraging locations for displaced birds, particularly for central-place foraging birds. In addition, it is unclear to what extent individuals habituate to wind farms over time.

A key knowledge gap is the drivers of spatio-temporal variation in bird distribution, which is known to vary considerably over space and time, potentially associated with variation in prey distribution. As well as population-level habitat preferences, individual variation in foraging site fidelity has also not been incorporated in assessments but could be important. Another key priority is to develop bird distribution maps that approximate true foraging locations. Deriving foraging locations from raw GPS fixes is challenging, but will become more tractable as more studies undertake dual GPS/activity logger deployments, enabling algorithms for detecting behaviours from GPS data available in a wider suite of species (Browning et al. 2017). Another key priority for future research is to quantify relationships between adult body mass and survival in new case study species and populations, since they may vary from current published estimates on Shetland kittiwakes and Norwegian puffins. Finally, there is a need to incorporate potential effects on other age classes (notably immatures) and periods of the year outside chick-rearing to obtain a comprehensive assessment of potential population-level effects of displacement and barrier effects of ORDs on seabirds.

5.8 Uncertainty

We used prediction intervals to quantify the uncertainty that arises from the intrinsic stochastic variability within the model and from the uncertainty regarding the overall level of prey. Within the context of this project it has not been feasible to quantify other sources of uncertainty: the uncertainty associated with parameter estimation, the structural uncertainty associated with the model, and the uncertainty associated with the spatial distributions of both birds and prey. Since many of these additional sources of uncertainty are likely to be substantial the prediction intervals that we present should be treated with caution, and regarded as *lower* bounds on the actual level of uncertainty.

When running SeabORD the user needs to specify:

a) the percentage of the population to be used when generating simulations;
b) the run of simulation runs, R.

The percentage should ideally be set to 100%, and the value of R should be set to be as large as is computationally feasible. In practice, however, the model is computationally intensive to run, so it may be necessary to run the model with percentages of less than 100% and with relatively small values of R. Both of these things will tend to have the following impacts:

1) they will tend to increase the width of the resulting prediction intervals – i.e. to increase uncertainty;

2) they will tend to reduce the defensibility and reliability of both estimates and prediction intervals.

We recommend that the model is never run with very small values of R, or with very small percentages of the population.

5.8.1 Choice of Snapshots

The number of snapshots should be chosen so as to reflect the number of surveys on which an assessment of the number of birds utilising a particular area of sea would usually be based.

The value of P2 is likely to vary considerably between simulation runs, especially if the number of snapshots is small. This is correct, and reflects the fact that snapshot surveys are likely to be capturing a relatively small proportion of the overall population. This does mean, however, that:

1) it is particularly important to use sufficiently large values for the percentage of population being simulated, and for the number of simulation runs (the value of R), if the values of P2 are to be used;
2) it is *not* appropriate to keep repeatedly re-running the model until a particular outcome for P2 is achieved – to do so is a mis-use of the model. This is true for all of the metrics, but the high variability in P2 between simulation runs means it is particularly relevant here.

6 Acknowledgements

This project was funded by Scottish Government's Contract Research Fund. We wish to thank Marine Scotland Science and the Project Steering Group for support and guidance throughout this project. This work draws on data supplied by RSPB, but the use of RSPB data does not imply RSPB endorsement of this tool.

7 References for Main Report

Birt-Friesen, V.L., Montevecchi, W.A., Cairns, D.K. & Macko, S.A. (1989) Activity-specific metabolic rates of free-living northern gannets and other seabirds. Ecology 70: 357-36

Cook, A.S.C.P., Parsons, M., Mitchell, I & Robinson, R.A. (2011) Reconciling policy with ecological requirements in biodiversity monitoring. Marine Ecology-Progress Series 434:267-277

Daunt, F., Benvenuti, S., Harris, M.P., Dall'Antonia, L., Elston, D.A. & Wanless, S. (2002) Foraging strategies of the black-legged kittiwake *Rissa trydactyla* at a North sea colony: evidence for a maximum foraging range. Marine Ecology Progress Series 245: 239-247

Daunt, F. & Wanless, S. (2008) Determining marine Special Protection Areas (mSPAs) for breeding seabirds. Report to RSPB.

Daunt, F., Bogdanova, M.I., Newell, M., Harris, M.P. & Wanless, S. (2011) Literature review of foraging distribution, foraging range and feeding behaviour of common guillemot, razorbill, Atlantic puffin, black-legged kittiwake and northern fulmar in the Forth/Tay region. Report to FTOWDG

Daunt, F., Wanless, S., Greenstreet, S.P.R., Jensen, H., Hamer, K.C. & Harris, M.P. (2008) The impact of the sandeel fishery closure in the northwestern North Sea on seabird food consumption, distribution and productivity. Canadian Journal of Fisheries and Aquatic Sciences 65: 362-381

Daunt, F., Wanless, S., Peters, G., Benvenuti, S., Sharples, J., Gremillet, D. & Scott, B. (2006). Impacts of oceanography on the foraging dynamics of seabirds in the North Sea. In: Top predators in marine ecosystems: their role in monitoring and management. (eds I.L. Boyd, S. Wanless & K. Camphuysen). Cambridge University Press, Cambridge, pp177-190.

Dolman, P.M. (1995) The intensity of interference varies with resource density: evidence from a field study with snow buntings, *Plectrophenox nivalis*. Oecologia 102:511-514

Drewitt, A.L. & Langston, R.H.W. (2006) Assessing the impacts of wind farms on birds. Ibis, 148, 29-42

Ens, B.J. & Goss-Custard, J.D. (1984) Interference amongst oystercatchers *Haematopus ostralegus* feeding on mussels *Mytilus edulis* on the Exe estuary. Journal of Animal Ecology 53: 217-231.

Enstipp, M.R., Daunt, F., Wanless, S., Humphreys, E., Hamer, K.C., Benvenuti, S. & Gremillet, D. (2006) Foraging energetics of North Sea birds confronted with fluctuating prey availability. In: Top predators in marine ecosystems: their role in monitoring and management. (Eds I.L. Boyd, S. Wanless & K. Camphuysen). Cambridge University Press, Cambridge, pp191-210

Enstipp, M.R., Grémillet, D. & Jones, D.R. (2007) Investigating the functional link between prey abundance and seabird predatory performance. Marine Ecology Progress Series 331: 267-279

Erikstad, K.E. Sandvik, H., Fauchald, P. & Tveraa, T. (2009) Short- and long-term consequences of reproductive decisions: an experimental study in the puffin 90: 3197-3208

Fauchald, P. (2009). Spatial interaction between seabirds and prey: Review and synthesis. Marine Ecology Progress Series, 391, 139–151.

Fox, A., Christensen, T.K., Desholm, M., Kahlert, J., & Petersen, I.K. (2006). Chapter 7 Birds: avoidance responses and displacement. In: Danish offshore wind: key environmental issues The Environment Group: The Danish Energy Agency, The Danish Nature Agency, DONG Energy and Vattenfall. pp. 94-111.

Frederiksen, M. Mavor, R.A. & Wanless, S. (2007) Seabirds as environmental indicators: the advantages of combining data sets. Marine Ecology Progress Series 352: 205–211

Freeman, S., Searle, K. Bogdanova, M., Wanless, S. & Daunt, F. (2014) Population dynamics of Forth & Tay breeding seabirds: review of available models and modelling of key breeding populations. Ref MSQ-0006. Final report to Marine Scotland Science

Gabrielsen, G.W., 1996. Energy expenditure of breeding Common Murres. Occas. Pap. Can. Wildl. Serv. 91, 49–58.

Garthe, S., Markones, N. and Corman, A-M. 2016. Possible impacts of offshore wind farms on seabirds: a pilot study in northern gannets in the southern North Sea. Journal of Ornithology DOI 10.1007/s10336-016-1402-y

Gaston, A.J. & Hipfner, J.M. (2006) Body mass changes in Brunnich's guillemots Uria lomvia with age and breeding stage. Journal of Avian Biology 37: 101-109

Goss-Custard, J.D., Caldow, R.W.G., Clarke, R.T., Durell, S.E.A. le V. dit & Sutherland, W.J. (1995) Deriving population parameters from individual variations in foraging behaviour. I. Empirical game theory distribution model of oystercatchers *Haematopus ostralegus* feeding on mussels *Mytilus edulis*. Journal of Animal Ecology 64:265-276

Golovkin, A.N., 1963. On eating up of fish by Uria species and Rissa tridactyla during their nesting period in the Barents Sea. Zool. Zh. 42, 408–416.

Grecian, W.J., Inger, R., Attrill, M.J., Bearhop, S., Godley, B.J., Witt, M.J. & Votier, S.C. (2010) Potential impacts of wave-powered marine renewable energy installations on marine birds. Ibis 152: 683-97

Grémillet, D., Wright, G., Lauder, A., Carss, D.N. & Wanless, S. (2003) Modelling the daily food requirements of wintering great cormorants: a bioenergetics tool for wildlife management. J. Appl. Ecol. 40:266-277

Hamer,K.C., Humphreys, E. M. Garthe, S. Hennicke, J., Peters, G., Grémillet, D., Phillips, R.A., Harris, M.P. & Wanless, S. (2007). Annual variation in diets, feeding locations and foraging behaviour of gannets in the North Sea: flexibility, consistency and constraint. Marine Ecology Progress Series 338: 295-305.

Harris, M.P. (1979) Measurements and weights of British puffins. Bird Study 26: 179-186

Harris, M.P. & Wanless, S. (1988) Measurement and seasonal changes in weight of guillemots Uria aalge at a breeding colony. Ringing and Migration 9: 32-36

Harris, M.P. & Wanless, S. (2011) The Puffin. T & A.D. Poyser.

Harris, M.P., Bogdanova, M.I., Daunt, F. & Wanless, S. (2012) Feeding areas of Atlantic Puffins *Fratercula arctica* assessed by GPS technology. Ringing & Migration 27:43-49

Harris, M.P., Newell, M., Daunt, F., Speakman, J.R. & Wanless, S. (2008) Snake pipefish Entelurus aequoreus are poor food for seabirds. Ibis, 150, 413-415

Hassell, M.R. & Varley, G.C. (1969) New inductive population model for insect parasites and its bearing on biological control. Nature 223. 1133-1136.

Hilton, G.M., Furness, R.W. & Houston, D.C. (2000) A comparative study of digestion in North Atlantic seabirds. Journal of Avian Biology, 31: 36-46.

Humphreys. E.M. (2002) Energetics of spatial exploitation of the North Sea by kittiwakes breeding on the Isle of May, Scotland. Unpublished PhD thesis, University of Stirling

Jitlal, M., Burthe, S., Freeman, S. & Daunt, F. (2017) Testing and validating metrics of change produced by Population Viability Analysis (PVA). Report to Marine Scotland.

Langton, R., Davies, I.M. & Scott, B.E. (2011) Seabird conservation and tidal stream and wave power generation: information needs for predicting and managing potential impacts. Marine Policy 35: 623-30

Langton, R., Davies, I.M. & Scott, B.E. (2014) A simulation model coupling the behaviour and energetics of a breeding central place forager to assess the impact of environmental changes. Ecological Modelling, 273: 31-43.

Larsen, J.K. & Guillemette, M. (2007) Effects of wind turbines on flight behaviour of wintering common eiders: implications for habitat use and collision risk. Journal of Applied Ecology 44: 516-522.

Leonhard, S.B., Pedersen, J., Grøn, P.N., Skov, H., Jansen, J., Topping, C. & Petersen, I.K. (2013) Chapter 5 Wind farms affect common scoter and red-throated diver behaviour. In: Danish offshore wind: key environmental issues – a follow-up. The Environment Group: The Danish Energy Agency, The Danish Nature Agency, DONG Energy and Vattenfall. pp 70-93

Leopold, M.F., Dijkman, E.M., Teal, L. & the OWEZ-Team (2011) Local birds in and around the Offshore Wind Farm Egmond an Zee (Owez) (T-0 & T-1, 2002-2010). IMARES Wageningen UR, Texel.

Lewis, S., Sherratt, T.N., Hamer, K.C., Harris, M.P. & Wanless, S. (2003) Contrasting diet quality in northern gannets *Morus bassanus* at two colonies. Ardea 91: 167-176.

McDonald, C., Searle, K., Wanless, S & Daunt, F. (2012) Effects of displacement from marine renewable development on seabirds breeding at SPAs. Marine Scotland Research Projects Workpackage E

Masden, E.A., Haydon, D.T., Fox, A.D. & Furness, R.W. (2010) Barriers to movement: Modelling energetic costs of avoiding marine wind farms amongst breeding seabirds. Marine Pollution Bulletin 60: 1085-1091

Middlemas, S.J., Barton, T.R., Armstrong, J.D., Thompson, P.M. (2006) Functional and aggregative responses of harbour seals to changes in salmonid abundance. Philosophical Transactions of the Royal Society B 273:193-198.

Montevecchi, W., Ricklefs, R., Kirkham, I., & Gabaldon, D. (1984). Growth energetics of nestling Northern gannets (*Sula bassanus*). The Auk, 101: 334–341.

Nelson, B. (2013) Early warnings of climate change on ecosystems: hormonally-mediated life-history decisions in seabirds. Unpublished PhD thesis, University of Glasgow

Newell, M., Harris, M., Wanless, S., Burthe, S., Bogdanova, M., Gunn, C. & Daunt, F. (2016). The Isle of May long-term study (IMLOTS) seabird annual breeding success 1982-2016. NERC Environmental Information Data Centre. https://doi.org/10.5285/02c98a4f-8e20-4c48-8167-1cd5044c4afe

NIRAS/DHI (2018) Bird avoidance behaviour and collision impact monitoring at offshore wind farms. Report to ORJIP

Oro, D. & Furness, R.W. (2002) Influences of food availability and predation on survival of kittiwakes. Ecology 83: 2516-2528

Pennycuick, C.J. (1997) Actual and ‘optimum’ flight speeds: field data reassessed. Journal of Experimental Biology 200: 2355–2361.

Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M. & Fox, A.D. (2006) Final results of bird studies at the offshore wind farms at Nysted and Horns Rev, Denmark. Report commissioned by DONG Energy and Vattenfall A/S. National Environmental Research Institute. 166pp.

Petersen, I.K., MacKenzie, M.L., Rexstad, E., Wisz, M.S. & Fox, A.D. (2011) Comparing pre- and post-construction distributions of long-tailed ducks Clangula hyemalis in and around the Nysted offshore wind farm, Denmark : a quasi-designed experiment accounting for imperfect detection, local surface features and autocorrelation. CREEM Technical Report 2011-1.

Robinson RA. BirdFacts: profiles of birds occurring in Britain & Ireland (BTO Research Report 407). 2005.

Scottish Government (2011) Habitats Regulations Appraisal of Draft Plan for Offshore Wind Energy in Scottish Territorial Waters: Appropriate Assessment Information Review (2011). http://www.scotland.gov.uk/Publications/2011/03/04165857/15

Searle, K., Butler, A., Mobbs, D., Bogdanova, M., Wanless, S., Bolton, M. & Daunt, F. (2015) At-Sea turnover of breeding seabirds. Scottish Marine and Freshwater Science 6 (10).

Searle, K., Mobbs, D., Butler, A., Bogdanova, M., Freeman, S., Wanless, S. & Daunt, F. (2014) Population consequences of displacement from proposed offshore wind energy developments for seabirds breeding at Scottish SPAs (CR/2012/03). Report to Scottish Government.

Thaxter, C.B., Daunt, F., Grémillet, D., Harris, M.P., Benvenuti, S., Watanuki, Y., Hamer, K.C. & Wanless, S. (2013) Modelling the effects of prey size and distribution on prey capture rates of two sympatric marine predators. PLoS One 8: e79915

Thaxter, C.B., Lascelles, B., Sugar, K., Cook, A.S.C.P., Roos, S., Bolton, M., Langston, R.H.W. & Burton, N.H.K. (2012) Seabird foraging ranges as a preliminary tool for identifying candidate Marine Protected Areas. Biological Conservation, 156, 53-61.

Thaxter, C.B., Wanless, S., Daunt, F., Harris, M.P., Benvenuti, S., Watanuki, Y., Grémillet, D & Hamer, K.C. (2010) Influence of wing loading on trade-off between pursuit-diving and flight in common guillemots and razorbills. Journal of Experimental Biology 213: 1018-1025

Thaxter, CB, VH Ross-Smith, W Bouten, NA Clark, GJ Conway, MM Rehfisch, NHK Burton. 2015. Seabird–wind farm interactions during the breeding season vary within and between years: A case study of lesser black-backed gull Larus fuscus in the UK. Biological Conservation 186:347-358.

Vanermen, N., Stienen, E.W.M., Onkelinx, T., Courtens, W., Van de walle, M. & Verstraete, H. (2011) Seabirds and offshore wind farms: power and impact analysis. In: Offshore wind farms in the Belgian part of the North Sea: selected findings from the baseline and targeted monitoring (Eds: Degraer, S., Brabant, R. & Rumes, B). Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Pp93-129

Vanermen, N., Stienen, E.W.M., Onkelinx, T., Courtens, W., Van de walle, M., Verschelde, P. & Verstraete, H. (2012) Seabirds and offshore wind farms monitoring results 2011. Research Institute for Nature and Forest, Brussels. INBO.R.2012.25.

Wakefield, E.D., Owen, E., Baer, J., Daunt, F., Dodd, L.S., Green, J.A., Guildford, T., Mavor, R., Miller, P.I., Newell, M., Newton, S.F., Robertson, G., Shoji, A., Soanes, L.M., Votier, S., Wanless, S. and Bolton, M. (2017) Breeding density, fine scale telemetry and large-scale modelling reveal the regional distribution of a four-species seabird assemblage. Ecological Applications

Wanless, S., Bacon, P.J., Harris, M.P., Webb, A.D., Greenstreet, S.P.R. & Webb, A. (1997) Modelling environmental and energetic effects on feeding performance and distribution of shags (*Phalacrocorax aristotelis*): integrating telemetry, geographical information systems, and modelling techniques. ICES Journal of Marine Science 54: 524-544

Wanless, S., Daunt, F., Camphuysen, C.J., Humphreys, E. & Hamer, K.C. (2005) Setting the scene: seabird foraging behaviour, diet and breeding success. In: Interactions between the marine environment, predators, and prey: implications for sustainable sandeel fisheries (IMPRESS) (Ed. C.J. Camphuysen). Final Report to EU, Texel, Netherlands. pp 88-134.

Wanless, S., Frederiksen, M., Harris, M.P. & Freeman, S.F. (2006) Survival of Gannets *Morus bassanus* in Britain and Ireland, 1959–2002. Bird Study 53: 79–85.

WWT Consulting (2012) Demographic data, population model and outputs. Wildfowl and Wetlands trust, Gloucestershire. Strategic Ornithological Support Services (SOSS) Report SOSS-04 Gannet Population Viability Analysis.

8 Appendix A - Literature Review

Barrier Effects and Displacement of Seabirds by Offshore Wind Farms

Bob Furness, Mark Trinder and Francis Daunt
December 2016

1. Introduction

Seabirds may be affected by Offshore Wind Farms (OWFs) in several ways:

1. Collision mortality as a result of birds flying into rotating turbine blades (Desholm and Kahlert 2005; JNCC et al. 2014);
2. Displacement/disturbance effects as a consequence of seabirds avoiding OWFs or being disturbed from the area by associated activities such as helicopter or ship traffic, resulting in habitat loss (Garthe and Hüppop 2004; Furness et al. 2013; JNCC 2015);
3. Barrier effects when seabirds avoid OWFs so have to fly greater distances (including changing flight trajectory to fly under or over as well as around OWFs) so incur increased energetic costs or time costs or both (Masden et al. 2009, 2010);
4. Enhanced opportunities for roosting on structures that may reduce thermostatic costs compared to roosting on the sea surface (Humphreys et al. 2007; Garthe et al. 2012a);
5. Enhanced opportunities for foraging, either as a consequence of OWFs providing roosting platforms allowing coastal seabirds to extend range into the offshore environment (Dierschke et al. 2016), or as a consequence of OWFs causing local increases in prey abundance or density that can be exploited by seabirds (Stenberg et al. 2015; Coates et al. 2016).

This literature review only considers literature relating to identification of, and quantification of Displacement/disturbance and Barrier effects (Items 2 and 3 above), and the ways in which these impacts are assessed in relation to OWF planning applications.

We used search engines and databases designed to identify a wide suite of literature types, including Web of Knowledge, Google Scholar and ResearchGate. We include in the review publically available contract reports underpinning Environmental Statements, guidance documents prepared by Statutory Nature Conservation Bodies (SNCBs), and publically available submissions from stakeholders such as RSPB, as well as published papers in scientific journals. Searches for material to inform this review were carried out between 5 and 10 October 2016.

Predictions as to the likely sensitivity of different marine bird species to impacts from OWFs, and hence the vulnerability of their populations to these impacts, have been made on the basis of seabird ecology and behaviour and population status (Garthe and Hüppop 2004; Furness et al. 2013). Garthe and Hüppop (2004) produced a single score based on the assessed sensitivity to both collisions and displacement/disturbance. A significant step forward in the Furness et al. (2013) analysis compared to that of Garthe and Hüppop (2004) was the separation of impacts into two distinct categories and scores; collision impact and displacement/disturbance impact, which recognises that the seabird species that score highly on one of these scales do not necessarily score highly on the other. An extension of this ranking/scoring approach is to use these scores in a GIS to present the spatial distribution of vulnerability, allowing detection of hot spots, and conversely to identify regions where the numbers of vulnerable seabirds are low so that OWF development might be targeted into areas where impacts on marine birds would be low (Bradbury et al. 2014). These approaches benefit from iterative adjustment to seabird sensitivity scores as new empirical evidence becomes available from studies of the strength of barrier effects and displacement effects. For example, Wade et al. (2016) point out increasing empirical evidence that gannets show strong avoidance of offshore wind farms yet show little avoidance of, or disturbance by, ships. They also assume that over time the use of a scoring system will become obsolete as quantitative data become available from post-construction monitoring studies of collision rates, barrier effects and the magnitude of displacement from operating OWFs. Reducing uncertainty is a key requirement for future development (Masden et al. 2015). Further, research has focussed on the impacts of single stressors, yet it is important to gain an understanding of the cumulative effects of multiple stressors on seabirds (e.g. renewable developments, climate change, fisheries, pollution, predation, disease) for effective conservation management (Burthe et al. 2014). A particularly difficult problem is how to assess population-level impacts of barrier effects, disturbance and displacement (MacArthur Green 2013).

2. Barrier Effects

Empirical evidence from tracking the flight lines of birds (visually or more often by radar with species identity determined visually) in relation to operational OWFs shows that many marine bird species change flight trajectory to avoid coming close to turbines. These empirical studies show strong (avoidance) adjustment of flight trajectory in relation to operating OWFs by gannets, divers and sea ducks, less pronounced responses in auks, and little response in gulls, skuas and terns (Kahlert et al. 2004a,b; Desholm and Kahlert 2005; Petersen et al. 2006; Larsen and Guillemette 2007; Krijgsveld et al. 2008, 2010, 2011; Leopold and Camphuysen 2008, Leopold et al. 2010, 2011, 2013; Vanermen et al. 2011, 2013a,b,c, 2015a,b, 2016; Humphreys et al. 2015). Thus barrier effects seem unlikely for gulls (including kittiwakes), skuas and terns, and most likely to be seen in gannets, divers and sea ducks. Empirical evidence is that operating OWFs represent a semi-permeable barrier to marine birds; and that for birds entering OWFs, there is a strong tendency for them to fly between rows of turbines (i.e. showing meso-avoidance) (Desholm and Kahlert 2005).

GPS tracking data also has considerable potential to record the flight lines of birds in relation to OWFs. This approach is in its infancy, since the vast majority of GPS tracking has been undertaken at breeding colonies where no wind farm is present (but see Garthe et al. 2016). The advantage of this approach is that it opens up the potential to address questions relating not only to the spatial distribution of birds in the vicinity of a development, but also connectivity to SPAs and population level consequences of these developments – the ultimate goal of any assessment of impact.

The current ORJIP (Offshore Renewables Joint Industry Programme) ornithology project at Thanet OWF is using a combination of radar tracking and thermal camera filming of tracked birds that come close to turbines to measure behavioural responses of marine birds. A sample of tracked birds is identified to species by visual observation and some birds are also being tracked by laser rangefinder (NIRAS, DHI and DOF 2016). These studies should improve understanding of macro-avoidance, meso-avoidance and micro-avoidance behaviour (May 2015) and are being carried out primarily in relation to collision risk assessment. However, the data will also inform on barrier effects, since by definition a high macro-avoidance response indicates that the species is subject to a barrier effect. However, a key limitation of the ORJIP study is that it is not analysing flight lines directly for evidence of changing trajectories and, thus, barrier effects. Rather, it is assuming that relative proportions of birds inside and outside amount to an estimate of macro-avoidance. This assumption is problematic because differences in densities of birds may be due to other factors, such as prey availability. However, a large data set of flight lines has been collected for potential use in future analyses.

Evidence of a barrier effect can also be obtained indirectly by high definition digital aerial photography of the spatial distribution of marine birds at operating OWFs. In this case, barrier effect can be considered equivalent to the ‘macro-avoidance’ behaviour of marine birds, defined as proportions that avoid entering the wind farm. Analysis of the photographs allows estimation of marine bird macro-avoidance behaviour (Maclean et al. 2006; APEM 2014).

JNCC et al. (2014) suggest that an evidence-based macro-avoidance rate of 0.64 is appropriate for gannet. This suggests that at least two-thirds of gannets experience a barrier effect when approaching an OWF (since meso-avoidance responses closer to the edge of an OWF most likely represent a further component of a barrier effect). APEM (2014) carried out an aerial survey of waters around Greater Gabbard OWF to assess the suitability of aerial survey data to quantify wind farm macro-avoidance by gannets. They demonstrated that such an approach is feasible, although estimation of macro-avoidance (and hence barrier effect) requires a number of assumptions regarding the expected distribution of gannets in relation to locations of turbines and requires adequate numbers of birds to be present for statistical assessment of avoidance. However, this approach appears to have merit and could be developed further.

There is still considerable uncertainty as to the strength of barrier effect in relation to wind turbine size and placement. Where rows of turbines are further apart, seabirds may fly between rows rather than flying around the perimeter of the wind farm (Desholm and Kahlert 2005). However, greater spacing increases overall footprint so could increase barrier effects, or may increase barrier effects for species showing a strong response but decrease barrier effects for species showing a low response (Masden et al. 2012). This aspect of barrier effects remains to be investigated in a quantitative comparative way.

A significant limitation of the empirical evidence for barrier effects at operational OWFs is that almost all of the data relate to marine birds studied during the nonbreeding season and during daylight hours in ‘good’ weather conditions. Since most study sites where seabirds have been observed at OWFs are distant from breeding colonies, even birds observed during the breeding season are most likely to be nonbreeders (immature birds or birds taking a year off from breeding or birds that failed early and left the breeding area). Behavioural responses of breeding birds to OWFs may differ from those of nonbreeding birds. It has been suggested that the imperative to provision chicks may result in breeding adults during the chick-rearing period showing lower macro-avoidance than shown by nonbreeding birds (Green et al. 2016; RSPB 2016). There is no empirical evidence that such a difference exists in any seabird species, but life history theory suggests that it may. If it does, OWFs may represent less of a barrier effect to breeding seabirds than to nonbreeding seabirds, with a consequent potential for greater collision risk for breeding birds if they choose to fly through OWFs in order to minimise time taken to bring food to their chicks.

Modelling indicates that barrier effects are likely to have a negligible cost for migrating seabirds because the extra distance flown to go around rather than through an OWF represents a trivial distance in the context of a migration journey (Masden et al. 2009, 2012; Speakman et al. 2009; Topping and Petersen 2011). For breeding seabirds, the situation might be different. If subjected to barrier effects because they have to fly around OWFs located between their breeding site and feeding area, the extra time and energy expenditure required could affect aspects of breeding success such as chick growth and survival, could affect adult body condition and hence adult survival prospects, or could affect both of these components of fitness.

Scenario modelling of these effects using realistic (but uncertain) parameterisation has been developed in a number of ways with increasing complexity/sophistication (Speakman et al. 2009; Masden et al. 2010, 2012; McDonald et al. 2012; Langton et al. 2014; Searle et al. 2014). Speakman et al. (2009) used a simple bioenergetics model to assess the cost of extra flight time caused by a barrier effect for migrating seabirds and concluded that the impact would be trivial, but pointed out that repeated barrier effects where an OWF was between nest site and foraging area of breeding seabirds might represent a significant additional energy demand. That scenario was not explored in detail so the magnitude of the extra cost, and any consequences for fitness were not evaluated. Masden et al. (2010) used a similar bioenergetics approach but compared additional costs for breeding seabird species with differing ecologies. They concluded that barrier effects would be most likely to be evident in breeding seabirds with high wing loading (such as auks and cormorants) or with frequent foraging trips (such as terns). Species for which barrier effects were least likely to impact fitness were identified as species with low wing loading and few foraging trips per day (e.g. fulmar and gannet). They also concluded that barrier effects would be additional costs to effects caused by low prey abundance or adverse weather conditions, but would be likely to be much less than the influences of those. McDonald et al. (2012) developed a proof of concept model of common guillemot breeding energetics parameterised for the Isle of May, which combined energy costs of barrier effects and displacement (foraging habitat loss) for breeding adults and their chicks. They concluded that barrier effects and displacement could significantly impact both breeding success and body condition, but identified a number of constraints in the modelling caused by lack of adequate empirical data even for this much studied colony.

Searle et al. (2014) developed this modelling approach further by expanding the number of species, exploring the influence of the spatial distribution of prey and, crucially, estimating the effects on breeding success and survival of barrier effects (and displacement – see next section) for the first time. The model of the Bass Rock gannet population in relation to displacement and barrier effects caused by proposed offshore wind farms (Neart na Gaoithe, Inch Cape, Firth of Forth Seagreen Alpha and Bravo) concluded that cumulative impacts of displacement and barrier effect on gannet population dynamics at the Bass Rock would be negligible, and too small to detect from monitoring of breeding numbers, breeding success, or adult survival. Similarly, impacts on common guillemots and razorbills were estimated to be small under a range of tested scenarios. Impacts on Atlantic puffins were higher, and were influenced strongly by assumptions made regarding spatial distribution of their prey. Stronger impacts on kittiwakes were estimated from tested scenarios (Searle et al. 2014), although empirical evidence suggests that kittiwakes show little barrier effect and little displacement (Krijgsveld et al. 2008, 2010, 2011; Leopold and Camphuysen 2008, Leopold et al. 2010, 2011, 2013; Vanermen et al. 2011, 2013a,b,c, 2015a,b, 2016; Dierschke et al. 2016) so the appropriateness of those tested scenarios is uncertain.

3. Displacement

3.1 Empirical Evidence for Displacement of Marine Birds by Offshore Wind Farms

Avoidance/displacement of marine birds by OWFs has been studied by post-construction monitoring at many sites, including the following for which reports are publically available:

- Robin Rigg: Canning et al. 2013;
- North Hoyle: PMSS 2006, 2007; May 2008;
- Kentish Flats: Gill et al. 2008; Rexstad and Buckland 2012; Percival 2011, 2014; Banks et al. 2011;
- London Array: McCormack et al. 2013;
- Thanet: Ecology Consulting 2012; Percival 2013;
- Gunfleet Sands: Barker 2011;
- Scroby Sands: Perrow et al. 2006;
- Sheringham Shoal: Perrow et al. 2015;
- Bligh Bank: Vanermen et al. 2011, 2013a, 2015a, 2016;
- Thorntonbank: Vanermen et al. 2011, 2013a,b,c, 2015a, 2016;
- Princess Amalia: Leopold et al. 2010, 2011, 2013; Camphuysen 2011;

- Egmond aan Zee: Krijgsveld et al. 2008, 2010, 2011; Leopold and Camphuysen 2008; Leopold et al. 2011, 2013; Camphuysen 2011;
- BARD Offshore 1: Braasch et al. 2015;
- alpha ventus: Sonntag et al. 2011; Garthe et al. 2012b; Aumüller et al. 2013; Mendel et al. 2014, 2015; Hill et al. 2014; Welcker and Nehls 2016;
- Horns Rev 2: Skov et al. 2012; Petersen et al. 2014;
- Horns Rev 1: Petersen 2005, Petersen et al. 2004, 2006, 2014; Christensen et al. 2004; Christensen and Hounisen 2004; Fox et al. 2006; Petersen and Fox 2007; Blew et al. 2008; Skov et al. 2008; Leonhard et al. 2012, 2013;
- Tunø Knob: Guillemette et al. 1998, 1999; Tulp et al. 1999;
- Nysted: Kahlert et al. 2004a, 2004b; Petersen 2004; Fox et al. 2006; Petersen et al. 2006, 2008, 2011; Blew et al. 2008;
- Lillgrund: Nilsson and Green 2011;
- Utgrunden: Pettersson 2002, 2005.

In addition, avoidance/displacement has been studied at some sites for which reports have not yet been made public. These include Lincs Offshore Wind Farm (Webb et al. 2014, 2015, 2016), and possibly also Burbo Bank Offshore Wind Farm. Reports from those two sites were not available in the public domain at the time of writing of this review.

Comparing across studies is made more difficult than it might be due to differences in methodology used in different studies and differences in the way data are reported and interpreted. Nevertheless, Dierschke et al. (2016) reviewed evidence in scientific papers, and the grey literature relating to post-construction monitoring at these 20 OWFs in European waters (ten in the southern North Sea, four in the German Bight/Danish North Sea, four in the Baltic, two in the Irish Sea), to extract and classify data for displacement or attraction of 33 different marine bird species. There was considerable variation in responses reported from different OWFs. Nevertheless, there were clear patterns among different species. Dierschke et al. (2016) concluded that divers showed the strongest avoidance/displacement response, with none entering most OWFs and only incidental records in those where they were recorded; avoidance/displacement was close to 100%. This is consistent with divers also showing the strongest disturbance response to ships (Schwemmer et al. 2011). Gannets also showed consistent and strong avoidance/displacement with only a small minority of gannets entering OWFs, and there was some evidence that great crested grebes and fulmars also show strong avoidance/displacement but those species occurred at few OWFs or were present in low numbers making assessment difficult. Long-tailed ducks, common scoters, Manx shearwaters, razorbills, common guillemots, little gulls and Sandwich terns showed less consistent avoidance/displacement. Common eiders, kittiwakes, common terns and Arctic terns showed avoidance and attraction approximately equally over all studies, and in some cases the response varied from year to year in relation to food abundance within the OWF. Red-breasted mergansers, common gulls, black-headed gulls, great black-backed gulls, herring gulls, and lesser black-backed gulls showed overall a weak attraction to OWFs but with no effect at many sites. Great cormorants and shags showed strong attraction to OWFs, both species using OWF structures to extend their foraging range offshore.

An alternative approach to measure displacement of seabirds by OWFs may be the use of GPS tracking studies. Thaxter et al (2015) used tracking data to quantifying overlap between at-sea distributions of lesser black-backed gulls and offshore wind farms. Garthe et al. (2016) report on a preliminary tracking study of breeding gannets from Helgoland which demonstrates the potential to quantify the extent to which individuals avoid operational OWFs within the foraging range from the colony. As with barrier effects, GPS tracking of individuals of known breeding status and provenance, whose demography and physiology can be monitored at colonies, opens up the potential to quantify the population level consequences of these developments. In contrast, at-sea bird surveys can provide indirect evidence for displacement (or indeed barrier effects), but do not provide any information on either the provenance of individuals or the population level consequences of these effects at relevant SPAs. Substantially enhanced power to detect effects of displacement and barrier effects on demographic rates using tracking can be achieved by deploying GPS loggers on the same individuals before and after construction, in order to assess behavioural changes in individual birds and so accounting for the high individual variability in foraging among individuals typically observed in seabird colonies.

Quantifying displacement is more challenging than barrier effects because the tracks of displaced individual birds may not reveal that they had an initial preference for the wind farm footprint and were displaced from it, because they may fly directly to the displacement location such that their GPS track would not exhibit any detectable deviation away from the wind farm location. Thus, the GPS tracks of displaced individuals are hard to distinguish from those of individuals that instead had an initial preference for the location to which other birds were displaced. An approach that has frequently been employed when analysing bird survey or tracking data is to assume that changes in the spatial distribution of individuals between the pre-construction and post-construction periods have been caused by the marine renewables development. However, as previously explained in the context of limitations of the ORJIP project, changes in the spatial distribution of species may have occurred due to other factors (e.g. change in spatial distribution of prey). There are two alternative approaches that attempt to address this problem:

- BACI analysis, using another colony as a control;
- Gradient analysis at a single colony, where the change in distribution pre- and post-construction is quantified in relation to distance to the wind farm, thereby attributing changes in spatial distribution to the wind farm effect by way of its proximity to different distance bands of foraging locations.

BACI analyses are appealing in theory, but challenging in practice in the marine environment because control colonies that otherwise need to be very similar/identical to the 'impact colony' are rarely available. A gradient approach is more likely to be a useful approach to quantifying displacement in many circumstances.

3.2 Empirical Evidence for Survival Consequences of Displacement

There are no measurements of survival consequences of displacement of seabirds from OWF sites, nor are there measurements of changes in survival caused by displacement of seabirds from other marine developments or from disturbance of seabirds by shipping etc. Designing a scientific study to measure downstream mortality caused by such displacement would be extremely difficult. Annual mortality of adult seabirds tends to be low. For example, annual mortality of adult common guillemot is 6.1% (Horswill and Robinson 2015). However, a BACI or gradient analysis of seabird movements (i.e. comparing changes in foraging distribution of individuals in relation to distance from the focal development site), coupled with analyses of demographic consequences on changes in energetic budgets (ideally empirical, if not through energetic models such as devised by Searle et al. 2014), may offer a potential way forward, especially if based on GPS tracking data and in particular where movements, energetics and demography can be directly linked at the individual level.

Annual mortality of juveniles is generally much higher than that of adults; in the case of common guillemot it is estimated at 44% compared to 6.1% in adults. This would suggest that displacement impact could be much greater for juveniles than for adults because juveniles are more likely to be close to the threshold for mortality effects during much of the winter whereas adults appear better able to buffer themselves against adverse conditions unless these conditions are particularly extreme. Theory would suggest that mortality from displacement would be more likely the closer the population was to environmental carrying capacity. Since carrying capacity is likely to be determined by food abundance, displacement may result in mortality more often in years when food supply is unusually low (Kaiser et al. 2006). Reiertsen et al. (2014) showed that prey density in nonbreeding areas used by kittiwakes affects adult survival rate. Reynolds et al. (2011) showed that common guillemot survival rate varied among years in a synchronous manner across colonies, implying that it was driven by variation in foraging opportunities during winter in the shared wintering area of birds from multiple colonies. Szostek and Becker (2015) showed that survival and local recruitment of common terns was driven by carry-over effects from their wintering area. Those results suggest that mortality is likely to vary among years in relation to winter food supply, and, therefore, that any effect of displacement from OWFs will vary among years as a consequence of these variations. There may be no mortality consequence from displacement in years when environmental conditions are good and the population is below carrying capacity, but significant mortality may occur (especially to juveniles) if conditions are bad and the population is at or above carrying capacity. At present, such interactions can only be inferred, and they would be extremely challenging to measure.

3.3 Assessing Impacts of Displacement in OWF Planning Applications

Natural England and Joint Nature Conservation Committee produced an Interim Advice Note (NE and JNCC 2012) on how Developers and their Consultants should present information to inform assessment of the potential magnitude and consequences of displacement of marine birds from OWFs. They recommended the use of a matrix presenting numbers of birds that may be affected based on the mean maximum numbers of the species recorded in the area during pre-construction (‘baseline’) surveys. The matrix columns present a range of values (from 0 to 100%) of the percentage displaced from the OWF area plus a suitable surrounding buffer zone, while the rows present a range of values (from 0 to 100%) for the percentage of the displaced birds that would die as a consequence of displacement. The emphasis in this approach is to present the full range of possible combinations of displacement and mortality. There is no available evidence to indicate the appropriate value for the mortality resulting from displacement. This could be zero if, as may often be the case, birds are displaced from a relatively small area of foraging habitat that is not of particularly high quality. However, mortality may occur if the population is already at carrying capacity and if the habitat lost is of high quality, particularly for species that have a limited amount of suitable foraging habitat (such as sea ducks requiring shallow water and suitable mollusc prey stocks). This matrix approach has been used for assessment of potential impacts of OWF developments on seabirds considered more vulnerable to displacement, such as auks (e.g. APEM 2012, MacArthur Green 2015; Forewind 2013). The approach was to be updated following a Workshop in May 2015 (JNCC 2015). As a postscript added to this review in March 2017, we now note that further guidance was published in January 2017, to provide clarity on best practice use of the existing Matrix Approach (http://jncc.defra.gov.uk/page 4274). However, that further guidance remains 'interim' and does not incorporate outputs from the Workshop in May 2015.

The workshop recommended that the concept of 'displacement as habitat loss' should be explored as a longer-term potential assessment approach. The workshop also recommended that individual-based modelling would be a good longer-term goal, in cases where the matrix approach indicated that effects could potentially be present. The rationale for this approach is that it enables the population consequences of offshore wind farms on SPA seabird populations through barrier effects, displacement or any other effect (e.g. collisions) to be estimated, enabling the three key questions in assessments to be addressed i.e. a) is there a measurable effect of wind farms on individual birds i.e. is there evidence for collisions, displacement, barrier effects or other factors?; b) Are the affected individuals from SPAs (whether colony or marine)?; c) Are there consequences of these effects on SPA populations?

Poot et al. (2011) provided the first assessment of cumulative impacts on seabird populations of OWFs in Dutch territorial waters. They assumed that barrier effects were negligible, and reported that there was no significant displacement effect and, therefore, cumulative impacts were only from collision mortality. However, assessing data from the same OWFs, Lindeboom et al. (2011) reported that 'several bird species seem to avoid the park while others are indifferent or are even attracted' which appears to contradict Poot et al. (2011). IMARES (2015a,b) and Leopold et al. (2014, 2015) provided a spatial mapping framework for assessment of impacts of OWFs on seabird populations in southern North Sea waters. The assessment combined cumulative impacts of collision and displacement for all OWFs in the southern North Sea expected to be operational in 2023, and used Potential Biological Removal (PBR) models of maximum sustainable harvest as a reference against which to assess the impact on seabird populations. The impact of displacement was estimated using what the authors termed 'the extended Bradbury-method' (this was based on Bradbury et al. 2014). Leopold et al. (2014) suggested it would be highly precautionary to infer that 10% of displaced birds would die as a consequence of displacement. Leopold et al. (2014): '*extended their* [i.e. Bradbury et al. 2014's] *methods by introducing a scaling factor, which allows us to estimate absolute mortalities, per seabird species and per individual wind farm based on quantitative information on densities of seabirds*'. From their analysis, Leopold et al. (2014, 2015) estimated that the cumulative impact of all anticipated OWFs in the southern North Sea in 2023 in terms of combined collision and displacement mortality fell well below PBR thresholds for all seabird species.

Busch et al. (2013) considered cumulative habitat loss as a consequence of displacement of seabirds from OWFs on the scale of the whole North Sea in the context of the Marine Strategy Framework Directive. They used the ESAS (European Seabirds at Sea) database and a GIS approach to define areas of the North Sea in terms of habitat quality based on seabird density (high density implying high quality habitat). Overlay of seabird abundance data with the OWF scenario allows calculation of the potential loss of habitat for each species. Regarding the species showing the strongest avoidance/displacement by existing and currently proposed OWFs, they inferred that divers would incur the greatest loss of habitat (about 5.4% cumulatively over the North Sea as a whole), while gannets would lose 1.8% of their foraging habitat. Busch and Garthe (2016) advocated the use of a combination of a matrix table displaying the full range of potential displacement and mortality levels (as advocated by Natural England and JNCC) together with seasonal potential biological removal (PBR) assessments to provide a tool '*that increases confidence in the conclusions of impact assessments. If unrealistic displacement levels and/or mortality rates are required to equal or approach seasonal PBRs, this gives an indication of the likeliness of adverse impacts on the assessed population*'. Busch and Garthe (2016) demonstrated the approach by assessing the displacement impacts of an offshore wind farm cluster in the German North Sea on the local common guillemot population. However, it is now considered by Natural England and other SNCBs that PBR is an inadequate tool and that assessments should use PVA models, because PBR does not estimate the effect of additional mortality on population size, and because PBR assesses the level of impact above which impacts are likely to be unsustainable, and does not provide a sound assessment of levels of impact that are sustainable (Natural England 2016). This viewpoint has also recently been presented in the published literature by RSPB (Green et al. 2016).

4. Conclusions

To understand the consequences of barrier effects and displacement from wind farms on seabird populations involves two key steps a) quantifying the proportion of birds that are barrier affected and displaced; b) quantifying the energetic and, in turn, demographic consequences on barrier affected birds, and impacts of such changes on population size. These are challenging questions that require well designed studies.

To date, studies addressing the first question have shown marked variation in barrier effects and displacement among species. This work suggests that they may be important processes in some circumstances but less important in others. However, there is huge variation within species among studies. This suggests that extent of these effects is highly context dependent. One potential cause of this variation is intrinsic differences between central place foraging breeding individuals and independent wintering individuals. However, another reason for the strong variation among studies is that they have important limitations in terms of robust quantification of barrier effects and displacement. Of particular concern is where studies have made fundamental but untested assumptions in quantifying the proportion of birds affected, of which the most important is arguably that differences in the density of birds inside and outside wind farms can be attributed entirely to barrier effects/displacement, when alternative explanations may be equally plausible.

Another key limitation is that studies have not been structured to robustly address the second question outlined above. To test the demographic consequences of affected birds is a considerable challenge, but approaches do exist, detailed in this review, that make direct estimations of these links and are less reliant on assumptions that are open to challenge.

5. References

APEM. 2012. East Anglia ONE Offshore Windfarm: Environmental Statement Volume 2 Chapter 12 Ornithology, Marine and Coastal Appendices. Document Reference 7.3.7b Appendix 12.1. Ornithology Baseline Technical Report. APEM Ref 411490.

APEM. 2014. Assessing northern gannet avoidance of offshore windfarms. East Anglia Offshore Wind Ltd.

Aumüller, R., Boos, K., Freienstein, S., Hill, K. and Hill, R. 2013. Weichen Zugvögel Windenergieanlagen auf See aus? Eine Methode zur Untersuchung und Analyse von Reaktionen tagsüber ziehender Vogelarten auf Offshore-Windparks. Vogelwarte 51, 3–13.

Banks, A.N., Graham, A., Jowett,M., Lawrie, K., Rowell, H., Holroyd, S., Ball, M.,Wilkins,M., Anderson, E. and Williams, A.E. 2011. Red-throated divers & offshore wind farms in the Outer Thames: historic data review. APEM Scientific Report 411134, Stockport.

Barker, R., 2011. Gunfleet Sands Gunfleet Sands 2 OffshoreWind Farms. Year 1 Post-construction Ornithological Monitoring. NIRAS Consulting Ltd, Cambridge.

Blew, J., Hoffmann, M., Nehls, G. and Hennig, V. 2008. Investigations of the bird collision risk and the responses of harbour porpoises in the offshore wind farms Horns Rev, North Sea, and Nysted, Baltic Sea, in Denmark. Part I: Birds. Univ. Hamburg, Hamburg und BioConsult SH, Husum.

Braasch, A., Michalik, A. and Todeskino, D. 2015. Assessing Impacts of offshore Wind Farms on two highly Pelagic Seabird Species. In: Köppel, J., Schuster, E. (Eds.), Book of Abstracts: Conference on Wind Energy and Wildlife Impacts: 95. Berlin Institute of Technology, Berlin.

Bradbury, G., Trinder, M., Furness, R.W., Banks, A.N., Caldow, R.W.G. and Hume, D. 2014. Mapping seabird sensitivity to offshore wind farms. PLoS ONE 9(9), e106366.

Burthe, S., Wanless, S., Newell, M.A., Butler, A. and Daunt, F. 2014. Assessing the vulnerability of the marine bird community in the western North Sea to climate change and other anthropogenic impacts. Marine Ecology Progress Series 507, 277–295.

Busch, M., Kannen, A., Garthe, S. and Jessopp, M. 2013. Consequences of a cumulative perspective on marine environmental impacts: Offshore wind farming and seabirds at North Sea scale in context of the EU Marine Strategy Framework Directive. Ocean & Coastal Management 71, 213-224.

Busch, M. and Garthe, S. 2016. Approaching population thresholds in presence of uncertainty: Assessing displacement of seabirds from offshore wind farms. Environmental Impact Assessment Review 56, 31-42.

Camphuysen, C.J. 2011. Lesser Black-Backed Gulls Nesting at Texel. Foraging Distribution, Diet, Survival, Recruitment and Breeding Biology of Birds Carrying Advanced GPS Loggers. NIOZ Report 2011–05, Texel.

Canning, S., Lye, G. and Kerr, D. 2013. Analysis of Marine Ecology Monitoring Plan Data from the Robin Rigg Offshore Wind Farm, Scotland (Operational Year 3) – Birds. Natural Power Consultants, Dalry.

Christensen, T.K. and Hounisen, J.P. 2004. Investigations of Migratory Birds during Operation of Horns Rev offshore Wind Farm: Preliminary Note of Analysis of Data from Spring 2004. National environmental Research Institute, Kalø.

Christensen, T.K., Hounisen, J.P., Clausager, I. and Petersen, I.K. 2004. Visual and radar observations of birds in relation to collision risk at the Horns Rev offshore wind farm Annual status report 2003 National environmental Research Institute, Kalø.

Coates, D.A., Kapasakali, D.A., Vincx, M. and Vanaverbeke, J. 2016. Short-term effects of fishery exclusion in offshore wind farms on macrofaunal communities in the Belgian part of the North Sea. Fisheries Research 179, 131-138.

Desholm, M. and Kahlert, J. 2005. Avian collision risk at an offshore wind farm. Biology Letters 1, 296–298.

Dierschke, V., Furness, R.W. and Garthe, S. 2016. Seabirds and offshore wind farms in European waters: Avoidance and attraction. Biological Conservation 202, 59-68.

Ecology Consulting. 2012. Thanet Offshore Wind Farm Ornithological Monitoring 2010–2011. Vattenfall & Royal Haskoning.

Forewind. 2013. Dogger Bank Creyke Beck Environmental Statement Chapter 11 Marine and Coastal Ornithology. Document Reference F-OFC-CH-011 Issue 3.

Fox, A., Christensen, T.K., Desholm, M., Kahlert, J. and Petersen, I.K. 2006. Chapter 7 Birds: avoidance responses and displacement. pp. 94-111. In: Danish Offshore Wind: Key Environmental Issues. DONG Energy, Vattenfall, Danish Energy Authority and Danish Forest and Nature Agency, Copenhagen, ISBN 87-7844-625-2.

Furness, R.W., Wade, H. and Masden, E.A. 2013. Assessing vulnerability of seabird populations to offshore wind farms. Journal of Environmental Management 119, 56–66.

Garthe, S and Hüppop, O. 2004. Scaling possible adverse effects of marine wind farms on seabirds: developing and applying a vulnerability index. Journal of Applied Ecology 41, 724-734.

Garthe, S., Ludynia, K., Hüppop, O., Kubetzki, U., Meraz, J.F. and Furness, R.W. 2012a. Energy budgets reveal equal benefits of varied migration strategies in northern gannets. Marine Biology 159, 1907-1915.

Garthe, S., Mendel, B., Kotzerka, J., Schwemmer, H. and Sonntag, N. 2012b. Possible impacts of wind farms on seabirds: the case study Alpha Ventus. FTZ, University of Kiel. http://rave2012.iwes.fraunhofer.de/img/pdfs/Session2/2.6_Garthe.pdf

Garthe, S., Markones, N. and Corman, A-M. 2016. Possible impacts of offshore wind farms on seabirds: a pilot study in northern gannets in the southern North Sea. Journal of Ornithology DOI 10.1007/s10336-016-1402-y

Gill, J.P., Sales, D., Pinder, S., Salazar, R., Ford, J. and Harding, I. 2008. Kentish Flats Wind Farm fifth ornithological monitoring report. Report to Kentish Flats Ltd. Environmentally Sustainable Systems, Edinburgh.

Green, R.E., Langston, R.H.W., McCluskie, A., Sutherland, R. and Wilson, J.D. 2016. Lack of sound science in assessing wind-farm impacts on seabirds. Journal of Applied Ecology doi: 10.1111/1365-2664.12731

Guillemette, M., Larsen, J.K. and Clausager, I. 1998. Impact assessment of an off-shore wind park on sea ducks. NERI Technical Report No 227. National Environmental Research Institute.

Guillemette, M., Larsen, J.K. and Clausager, I. 1999. Assessing the impact of the Tunø Knob wind park on sea ducks: the influence of food resources. NERI Technical Report No 263. National Environmental Research Institute.

Hill, R., Hill, K., Aumüller, R., Boos, K. and Freienstein, S. 2014. Testfeldforschung zum Vogelzug am Offshore-Pilotpark alpha ventus” und “Auswertung der kontinuierlich auf FINO1 erhobenen Daten zum Vogelzug der Jahre 2008 bis 2012. StUKplus-Enbericht (FKZ 0327689A/Avitec1 und Avitec2). Avitec Research, Osterholz-Scharmbeck.

Horswill, C. and Robinson, R.A. 2015. Review of seabird demographic rates and density dependence. JNCC Report No. 552. JNCC, Peterborough.

Horswill, C., O'Brien, S.H. & Robinson, R.A. 2016. Density-dependence and marine bird populations: are wind farm assessments precautionary? Journal of Applied Ecology 54: 1406-1414

Humphreys, E.M., Wanless, S. and Bryant, D.M. 2007. Elevated metabolic costs while resting on water in a surface feeder: the black-legged kittiwake Rissa tridactyla. Ibis 149, 106-111.

Humphreys, E.M., Cook, A.S.C.P. and Burton, N.H.K. 2015. Collision, displacement and barrier effect concept note. BTO Research Report 669. British Trust for Ornithology, Thetford.

IMARES. 2015a. Framework for assessing ecological and cumulative effects of offshore wind farms Part A: Methods. Netherlands Ministry of Economic Affairs and Ministry of Infrastructure and the Environment.

IMARES. 2015b. Framework for assessing ecological and cumulative effects of offshore wind farms Part B: Description and assessment of the cumulative effects of implementing the Roadmap for Offshore Wind Power. Netherlands Ministry of Economic Affairs and Ministry of Infrastructure and the Environment.

JNCC, NE, NIEA, NRW and SNH. 2014. Joint Response from the Statutory Nature Conservation Bodies to the Marine Scotland Science Avoidance Rate Review. JNCC.

JNCC. 2015. Seabird displacement impacts from offshore wind farms. MROG Workshop 6-7th May 2015. Workshop report. JNCC Report 568. ISBN 0963 8091 http://jncc.defra.gov.uk/page-7063 (including 14 online supplementary/background papers)

Kahlert, J., Desholm, M. and Clausager, I. 2004a. Investigations of Migratory Birds during Operation of Nysted offshore Wind Farm at Rødsand: Preliminary Analysis of Data from spring 2004. NERI note.

Kahlert, J., Petersen, I.K., Fox, A.D., Desholm, M. and Clausager, I. 2004b. Investigations of birds during construction and operation of Nysted offshore wind farm at Rødsand. Annual status report 2003. NERI Report.

Kaiser, M.J., Galanidi, M., Showler, D.A., Elliott, A.J., Caldow, R.W.G., Rees, E.I.S., Stillman, R.A. and Sutherland, W.J. 2006. Distribution and behaviour of common scoter *Melanitta nigra* relative to prey resources and environmental parameters. Ibis 148, 110–128.

Krijgsveld, K.L., Fijn, R.C., Heunks, C., van Horssen, P.W., Poot, M.J.M. and Dirksen, S. 2008. Effect studies Offshore Wind Farm Egmond aan Zee. Progress report on fluxes and behaviour of flying birds. Noordzeewind report nr OWEZ R 231 T1 20080304. Bureau Waardenburg, Culemborg.

Krijgsveld, K.L., Fijn, R.C., Heunks, C., van Horssen, P.W., de Fouw, J., Collier, M., Poot, M.J.M., Beuker, D. and Dirksen, S. 2010. Effect studies Offshore Wind Farm Egmond aan Zee. Progress report on fluxes and behaviour of flying birds covering 2007 & 2008. Noordzeewind report OWEZ R 231 T1 20100810. Bureau Waardenburg, Culemborg.

Krijgsveld, K.L., Fijn, R.C., Japink, M., van Horssen, P.W., Heunks, C., Collier, M.P., Poot, M.J.M., Beuker, D. and Dirksen, S. 2011. Effect studies Offshore Wind Farm Egmond aan Zee. Final report on fluxes, flight altitudes and behaviour of flying birds. Noordzeewind RapportOWEZ R 231 T1 20111110. BureauWaardenburg, Culemborg.

Langton, R., Davies, I.M. and Scott, B.E. 2014. A simulation model coupling the behaviour and energetics of a breeding central place forager to assess the impact of environmental changes. Ecological Modelling, 273, 31-43.

Larsen, J.K. and Guillemette, M. 2007. Effects of wind turbines on flight behaviour of wintering common eiders: implications for habitat use and collision risk. Journal of Applied Ecology 44, 516–522.

Leonhard, S.B., Skov, H., Jansen, J., Closter, R.M.,Mohlenberg, F., Pedersen, J.,Madsen, K.M., Grøn, P.N., Christensen, J. and Nygård, K. 2012. Food resources for Common Scoter. Horns Reef Monitoring 2009-2010. Orbicon, DHI, Wageningen IMARES, report to DONG Energy.

Leonhard, S.B., Pedersen, J., Grøn, P.N., Skov, H., Jansen, J., Topping, C. and Petersen, I.K. 2013. Chapter 5 Wind farms affect common scoter and red-throated diver behaviour. Pp. 70-93 In Danish Offshore Wind: Key Environmental Issues – a Follow-up. The Environment Group: The Danish Energy Agency, The Danish Nature Agency, DONG Energy and Vattenfall.

Leopold, M.F. and Camphuysen, C.J. 2008. Local birds in and around the offshore Wind Park Egmond aan Zee (OWEZ) (T1). Noordzeewind Rapport OWEZ R 221 T1 20080201. IMARES, Wageningen.

Leopold, M.F., Camphuysen, C.J., Verdaat, H., Dijkman, E.M., Meesters, H.W.G., Aarts, G.M., Poot, M. and Fijn, R. 2010. Local birds in and around the OffshoreWind Park Egmond aan Zee (OWEZ) (T0 & T1). Noordzeewind Rapport OWEZ R 221 T1 20100329. IMARES, Wageningen.

Leopold, M.F., Dijkman, E.M., Teal, L. and the OWEZ-Team. 2011. Local birds in and around the Offshore Wind Farm Egmond aan Zee (OWEZ) (T0 & T1, 2002–2010). Noordzeewind Rapport OWEZ R 221 T1 20111220. IMARES, Wageningen.

Leopold, M.F., van Bemmelen, R.S.A. and Zuur, A.F. 2013. Responses of local birds to the offshore wind farms PAWP and OWEZ off the Dutch mainland coast. Report C151/12. IMARES, Wageningen.

Leopold, M.F., Boonman, M., Collier,M.P., Davaasuren, N., Fijn, R.C., Gyimesi, A., de Jong, J., Jongbloed, R.H., Poerink, B.J., Kleyheeg-Hartman, J.C., Krijgsveld, K.L., Lagerveld, S., Lensink, R., Poot, M.J.M., van der Wal, J.T. and Scholl, M. 2014. A first approach to deal with cumulative effects on birds and bats of offshore windfarms and other human activities in the Southern North Sea. IMARES report no. C166/14, IMARES, Wageningen.

Leopold, M.F., Collier, M.P., Gyimesi, A., Jongbloed, R.H., Poot, M.J.M., van der Wal, J.T. and Scholl, M. 2015. Iteration cycle: Dealing with peaks in counts of birds following active fishing vessels when assessing cumulative effects of offshore wind farms and other human activities in the Southern North Sea. Additional note to IMARES report no. C166/14, Wageningen.

Lindeboom, H. J., H. J. Kouwenhoven, M. J. N. Bergman, S. Bouma, S. Brasseur, R. Daan, R. C. Fijn, D. de Haan, S. Dirksen, R, van Hal, R. Hille Ris Lambers, R. ter Hofstede, K. L. Krijgsveld, M. Leopold and M. Scheidat 2011. Short-term ecological effects of an offshore wind farm in the Dutch coastal zone; a compilation. Environmental Research Letters 6, 035101.

MacArthur Green. 2013. Extent of displacement, and mortality implications of displacement of seabirds by offshore wind farms. Dogger Bank Creyke Beck Environmental Statement Chapter 11 Appendix B.

https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010021/EN010021-000503-6.11.2%20Chapter%2011%20Appendix%20B%20Seabird%20Displacement%20and%20mortality%20implications%20-%20Application%20Submission_F-OFC-CH-011.pdf

MacArthur Green. 2015. East Anglia THREE Offshore Windfarm: Chapter 13 Offshore Ornithology. Environmental Statement, Vol. 1 Doc. Ref. 6.1.13

Maclean, I.M.D., Skov, H., Rehfisch, M.M. and Piper, W. 2006. Use of aerial surveys to detect bird displacement by offshore windfarms. BTO Research Report No. 446 to COWRIE. BTO, Thetford.

Masden, E.A., Haydon, D.T., Fox, A.D., Furness, R.W., Bullman, R. and Desholm, M. 2009. Barriers to movement: impacts of wind farms on migrating birds. ICES Journal of Marine Science 66, 746-753.

Masden, E.A., Haydon, D.T., Fox, A.D. and Furness, R.W. 2010. Barriers to movement: Modelling energetic costs of avoiding marine wind farms among breeding seabirds. Marine Pollution Bulletin 60, 1085–1091.

Masden, E.A., Reeve, R., Desholm, M., Fox, A.D., Furness, R.W. and Haydon, D.T. 2012. Assessing the impact of marine wind farms on birds through movement modelling. Journal of the Royal Society Interface 9, 2120-2130.

Masden, E.A., McCluskie, A., Owen, E. and Langston, R.H.W. 2015. Renewable energy developments in an uncertain world: The case of offshore wind and birds in the UK. Marine Policy 51, 169–172.

May, J. 2008. North Hoyle Offshore Wind Farm. Final annual FEPA monitoring report (2006–7) & Five year monitoring programme summary. NWP Offshore Ltd.

May, R.F. 2015. A unifying framework for the underlying mechanisms of avian avoidance of wind turbines. Biological Conservation 190, 179-187.

McCormack, D., McGovern, S., Goodship, N., Barlow, E., Rowlands, N. and Rehfisch, M. 2013. Aerial Bird Surveys in the Outer Thames Estuary SPA. APEM Scientific Report for Natural England, Stockport.

McDonald, C., Searle, K., Wanless, S. and Daunt, F. 2012. Effects of displacement from marine renewable development on seabirds breeding at SPAs: a proof of concept model of common guillemots breeding on the Isle of May. CEH, Edinburgh.

Mendel, B., Kotzerka, J., Sommerfeld, J., Schwemmer, H., Sonntag, N. and Garthe, S. 2014. Effects of the alpha ventus offshore test site on distribution patterns, behaviour and flight heights of seabirds. Federal Maritime and Hydrographic Agency & Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (Hrsg.): Ecological Research at the Offshore Windfarm alpha ventus. Springer, Wiesbaden, pp. 95–110.

Mendel, B., Sonntag, N., Sommerfeld, J., Kotzerka, J., Müller, S., Schwemmer, H., Schwemmer, P. and Garthe, S. 2015. Untersuchungen zu möglichem Habitatverlust und möglichen Verhaltensänderungen beiSeevögeln im Offshore-Windenergie-Testfeld (TESTBIRD). Schlussbericht zum Projekt Ökologische Begleitforschung am Offshore- Testfeldvorhaben alpha ventus zur Evaluierung des Standarduntersuchungskonzeptes des BSH (StUKplus). FTZ Westküste, Büsum.

Natural England 2016. https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010053/EN010053-001014-Natural%20England%20(5).pdf

NE and JNCC. 2012. Joint Natural England and JNCC Interim Advice Note: Presenting information to inform assessment of the potential magnitude and consequences of displacement of seabirds in relation of Offshore Windfarm Developments. NE/JNCC.

Nilsson, L. and Green, M. 2011. Birds in southern Öresund in relation to the wind farm at Lillgrund. Final report of the monitoring program 2001-2011. Lunds Universitet, Lund

NIRAS, DHI and DOF. 2016. Offshore Renewables Joint Industry Programme - Bird avoidance behaviour and collision impact monitoring at offshore wind farms. Interim Report 4 July 2016 to Vattenfall and Carbon Trust.

Percival, S. 2011. Kentish Flats OffshoreWind Farm Extension. Environmental Statement. Section 9: Offshore Ornithology. IPC Document Ref 4.2.9. Ecology Consulting on behalf of Vattenfall Wind Power.

Percival, S. 2013. Thanet Offshore Wind Farm Ornithological Monitoring 2012–2013. Vattenfall & Royal Haskoning.

Percival, S. 2014. Kentish Flats Offshore Wind Farm: Diver Surveys 2011–12 and 2012–13. Ecology Consulting, Durham.

Perrow, M.R., Skeate, E.R. and Tomlinson, M.L. 2006. Scroby Sands Ornithological Monitoring. Assessing the potential impact of the proposed wind farm upon Little Tern *Sterna albifrons*: the post construction phase beginning in 2005. Ecological Consultancy, Norwich.

Perrow, M.R., Harwood, A.J.P., Berridge, R. and Skeate, E.R. 2015. Avoidance of an Offshore Wind Farm by a Breeding Seabird has Implications for the Offshore Renewables Industry. In: Köppel, J., Schuster, E. (Eds.), Book of Abstracts: Conference on Wind Energy and Wildlife Impacts: 52. Berlin Institute of Technology, Berlin.

Petersen, I.K. 2004. Investigation of Birds during the Operational Phase of the Nysted offshore Wind Farm Preliminary notes on the issue of potential habitat loss. NERI note.

Petersen, I.K. 2005. Bird Numbers and Distribution in the Horns Rev Offshore Wind Farm Area. Annual Status Report 2004. National Environmental Research Institute, Kalø.

Petersen, I.K. and Fox, A.D. 2007. Changes in Bird Habitat Utilisation around the Horns Rev 1 offshore Wind Farm, with Particular Emphasis on Common Scoter. NERI Report, Århus.

Petersen, I.K., Clausager, I. and Christensen, T.K. 2004. Bird numbers and distribution in the Horns Rev offshorewind farm area. Annual status report 2003. National Environmental Research Institute, Kalø.

Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M. and Fox, A.D. 2006. Final Results of Bird Studies at the Offshore Wind Farms at Nysted and Horns Rev, Denmark. National Environmental Research Institute, Kalø.

Petersen, I.K., Fox, A.D. and Kahlert, J. 2008. Waterbird Distribution in and around the Nysted offshore Wind Farm, 2007. National Environmental Research Institute, Kalø.

Petersen, I.K., MacKenzie, M., Rexstad, E.,Wisz, M.S. and Fox, A.D. 2011. Comparing pre- and post-construction distributions of long-tailed ducks Clangula hyemalis in and around the Nysted offshore wind farm, Denmark: a quasi-designed experiment accounting for imperfect detection, local surface features and autocorrelation. CREEM Technical Report 2011–1.

Petersen, I.K., Nielsen, R.D. and Mackenzie, M.L. 2014. Post-Construction Evaluation of Bird Abundances and Distributions in the Horns Rev 2 offshore Wind Farm Area, 2011 and 2012. Aarhus University, Aarhus.

Pettersson, J. 2002. Bird observation in southern Kalmar Sound. Spring and autumn 2001. Report to Vindkompaniet AB and Enron Wind Sverige.

Pettersson, J. 2005. The Impact of Offshore Wind Farms on Bird Life in Southern Kalmar Sound, Sweden. Report to Swedish Energy Agency. Lunds Universitet, Lund.

PMSS. 2006. North Hoyle Offshore Wind Farm. Annual FEPA monitoring report (2004-5). NWP Offshore Ltd.

PMSS. 2007. North Hoyle Offshore Wind Farm. Annual FEPA monitoring report (2005-6). NWP Offshore Ltd.

Poot, M.J.M., van Horssen, P.W., Collier, M.P., Lensink, R. and Dirksen, S. 2011. Effect studies Offshore Wind Egmond aan Zee: cumulative effects on seabirds: A modelling approach to estimate effects on population levels in seabirds. Report No 11-026. Bureau Waardenburg, Culemborg.

Reiertsen, T.K., Erikstad, K.E., Anker-Nilssen, T., Barrett, R.T., Boulinier, T., Frederiksen, M., Gonzalez-Solis, J., Gremillet, D., Johns, D., Moe, B., Ponchon, A., Skern-Mauritzen, M., Sandvik, H. and Yoccoz, N.G. 2014. Prey density in non-breeding areas affects adult survival of black-legged kittiwakes Rissa tridactyla. Marine Ecology Progress Series 509, 289-302.

Rexstad, E. and Buckland, S. 2012. Displacement Analysis Boat Surveys Kentish Flats. SOSS Report 1A.

Reynolds, T.J., Harris, M.P., King, R., Swann, R.L., Jardine, D.C., Frederiksen, M. and Wanless, S. 2011. Among-colony synchrony in the survival of common guillemots Uria aalge reflects shared wintering areas. Ibis 153, 818-831.

RSPB. 2016. Written Representations for the Royal Society for the Protection of Birds Submitted for Deadline II 27 July 2016 Planning Act 2008 (as amended) in the matter of: Application by East Anglia THREE Limited for an Order Granting Development Consent for the East Anglia THREE Offshore Wind Farm. PINS Reference EN010056, Registration Identification 10032142.

Schwemmer, P., Mendel, B., Sonntag, N., Dierschke, V. and Garthe, S. 2011. Effects of ship traffic on seabirds in offshore waters: implications for marine conservation and spatial planning. Ecological Applications 21, 1851–1860.

Searle, K., Mobbs, D., Butler, A., Bogdanova, M., Freeman, S., Wanless, S. and Daunt, F. 2014. Population consequences of displacement from proposed offshore wind energy developments for seabirds breeding at Scottish SPAs (CR/2012/03). CEH Report to Marine Scotland Science.

Searle, K, A Butler, D Mobbs, M Bogdanova, S Wanless, M Bolton and F Daunt (2015) At-Sea Turnover of Breeding Seabirds: Final Report to Marine Scotland Science. Scottish Marine and Freshwater Science Vol 6 No 10. Edinburgh: Scottish Government, 60pp.

Skov, H., Piper, W. and Leonhard, S.B. 2008. Horns Rev II Offshore Wind Farm Monitoring of Resting Waterbirds. Baseline Studies 2007–08. Orbicon, Viby und DHI, Hørsholm.

Skov, H., Leonhard, S.B., Heinänen, S., Zydelis, R., Jensen, N.E., Durinck, J., Johansen, T.W., Jensen, B.P., Hansen, B.L., Piper, W. and Grøn, P.N. 2012. Horns Rev 2 Monitoring 2010- 2012. Migrating Birds. Orbicon, DHI, Marine Observers und Biola, Report commissioned by DONG Energy.

Sonntag, N., Güpner, F., Mendel, B. and Garthe, S. 2011. Untersuchungen zu möglichem Habitatverlust und möglichen Verhaltensänderungen bei Seevögeln im Offshore-Windenergie-Testfeld (TESTBIRD). Zwischenbericht StUKplus, FTZWestküste, Büsum.

Speakman, J., Gray, H. and Furness, L. 2009. University of Aberdeen report on effects of offshore wind farms on the energy demands of seabirds. Report to DECC; URN 09D/800.

Stenberg, C., Stottrup, J.G., van Deurs, M., Berg, C.W., Dinesen, G.E., Mosegaard, H., Grome, T.M. and Leonhard, S.B. 2015. Long-term effects of an offshore wind farm in the North Sea on fish communities. Marine Ecology Progress Series 528, 257-265.

Szostek, K.L. and Becker, P.H. 2015. Survival and local recruitment are driven by environmental carry-over effects from the wintering area in a migratory seabird. Oecologia 178, 643-657.

Thaxter, C.B., Ross-Smith, V.H., Bouten, W., Clark, N.A., Conway, G.J., Rehfisch, M.M. and Burton, N.H.K. 2016. Seabird–wind farm interactions during the breeding season vary within and between years: A case study of lesser black-backed gull *Larus fuscus* in the UK. Biological Conservation 186, 347–358.

Topping, C. and Petersen, I.K. 2011. Report on a red-throated diver agent-based model to assess the cumulative impact from offshore wind farms. Report commissioned by the Environmental Group. Aarhus University, DCE - Danish Centre for Environment and Energy. 44 pp.

Tulp, I., Schekkerman, H., Larsen, J.K., van der Winden, J., van de Haterd, R.J.W., van Horssen, P., Dirksen, S. and Spaans, A.L. 1999. Nocturnal flight activity of sea ducks near the windfarm Tunø Knob in the Kattegat. BureauWaardenburg report nr. 99.64.

Vanermen, N., Stienen, E.W.M., Onkelinx, T., Courtens, W. and Van de Walle, M. 2011. Seabirds & offshore wind farms: Power and impact analyses 2010. In: Degraer, S., Brabant, R., Rumes, B. (Eds.), Offshore wind farms in the Belgian part of the North Sea: Selected findings from the baseline and targeted monitoring: 93–129. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine ecosystem management unit.

Vanermen, N., Stienen, E.W.M., Courtens, W., Onkelinx, T., Van de Walle, M. and Verstraete, H. 2013a. Bird monitoring at offshore wind farms in the Belgian part of the North Sea. Assessing seabird displacement effects. Instituut voor Natuur- en Bosonderzoek, Brussel.

Vanermen, N., Brabant, R., Stienen, E., Courtens, W., Onkelinx, T., Van de Walle, M., Verstraete, H. and Vigin, L. 2013b. Bird monitoring at the Belgian offshore wind farms: results after five years of impact assessment. In: Degraer, S., Degraer, S., Brabant, R., Rumes, B. (Eds.), Environmental Impacts of offshore Wind Farms in the Belgian Part of the North Sea: Learning from the Past to

Optimise Future Monitoring Programmes: 49–61. Royal Belgian Institute of Natural Sciences, Brussels.

Vanermen, N., Stienen, E., Courtens, W., Van de Walle, M. and Verstraete, H. 2013c. Attraction of seabirds. In: Degraer, S., Brabant, R., Rumes, B. (Eds.), Environmental Impacts of offshore Wind Farms in the Belgian Part of the North Sea: Learning from the Past to Optimise Future Monitoring Programmes: 163–165. Royal Belgian Institute of Natural Sciences, Brussels.

Vanermen, N., Onkelinx, T., Courtens, W., Van de Walle, M., Verstraete, H. and Stienen, E.W.M. 2015a. Seabird avoidance and attraction at an offshore wind farm in the Belgian part of the North Sea. Hydrobiologia 756, 51–61.

Vanermen, N., Onkelinx, T., Verschelde, P., Courtens, W., Van de Walle, M., Verstraete, H. and Stienen, E.W.M. 2015b. Assessing seabird displacement at offshore wind farms: power ranges of a monitoring and data handling protocol. Hydrobiologia 756, 155–167.

Vanermen, N., Courtens, W., Van de Walle, M., Verstraete, H. and Stienen, E.W.M. 2016. Seabird Monitoring at offshore Wind Farms in the Belgian Part of the North Sea: Updated Results for the Bligh Bank & First Results for the Thorntonbank. Instituut voor Natuuren Bosonderzoek, Brussel.

Wade, H.M., Masden, E.A., Jackson, A.C. and Furness, R.W. 2016. Incorporating data uncertainty when estimating potential vulnerability of Scottish seabirds to marine renewable energy developments. Marine Policy 70, 108-113.

Webb, A., Mackenzie, M., Kidney, D., and Donovan, C. 2014. *Lincs Wind Farm First annual post-construction aerial ornithological monitoring report*. Report by HiDef Aerial Surveying Limited to Centrica Renewable Energy Limited, HP00045-703.

Webb, A., Mackenzie, M., Caneco, B., and Donovan, C. 2015. *Lincs Wind Farm Second annual post-construction aerial ornithological monitoring report.* Report by HiDef Aerial Surveying Limited to Centrica Renewable Energy Limited, CREL REF: LN-E-EV-013-0006-400013-006.

Webb, A., Mackenzie, M., Caneco, B., and Donovan, C. 2016. *Lincs Wind Farm Third annual post-construction aerial ornithological monitoring report.* Report by HiDef Aerial Surveying Limited to Centrica Renewable Energy Limited.

Welcker, J. and Nehls, G. 2016. Displacement of seabirds by an offshore wind farm in the North Sea. Marine Ecology Progress Series 554, 173–182.

9 Appendix B - Species-Specific Parameters used in Foraging Simulation Model within the Analytical Tool

Table B.1: Parameter values used SeabORD for each species.

Name	Description	Units	kittiwake	guillemot	razorbill		puffin
BM_adult_mn	Initial adult body mass mean	g	372.69	920.34	582.9		392.8
BM_adult_sd	Initial adult body mass standard deviation	g	33.62	57.44	26		21.95
BM_adult_mortf	Critical mass below which adult is assumed dead	proportion of mean mass	0.6	0.6	0.6		0.6
BM_adult_abdn	Critical mass below which adult abandons chick	proportion of mean mass	0.8	0.8	0.8		0.8
BM_chick_mn	Initial chick body mass mean	g	36	75.8	64.9		42.2
BM_chick_sd	Initial chick body mass standard deviation	g	2.2	1	6.3		3.7
BM_Chick_mortf	Critical mass below which chick is dead	proportion of initial mass	0.6	0.6	0.6		0.6
daylength	Number of hours per time step	hours	36	24	24		24
seasonlength	Number of time steps per season		30	21	21		40
unattend_max_hrs	Critical time threshold for unattendance at nest above which a chick is assumed to die through exposure or predation	hours	18	18	18		0
adult_DEE_mn	Adult daily energy expenditure mean	kJ	802	1489.1	1231.89		871.5
adult_DEE_sd	Adult daily energy expenditure standard deviation	kJ	196	169.9	95.3		80
chick_DER	Chick energy requirement (Harris & Wanless 1985)	kJ per day	525.71	221.71	195.67		325
IR_max	Maximum prey intake rate	g per minute	4.369	2.95	3.066		3.293
IR_half_a	Intake rate parameter	g	900	700	600		1000
IR_half_b	Intake rate parameter		0.02	0.02	0.02		0.02
flight_msec	Average speed in flight	metres per second	13.1	19.1	16		17.6
assim_eff	Assimilation efficiency, Hilton et al 2000b		0.74	0.78	0.79		0.78
energy_prey	Energy gained from prey (Harris et al 2008)	kJ per gram	6.1	6.1	6.1		6.1
energy_nest	Energy cost of nesting at colony	kJ per day	427.75	1168.91	932.17		665.41
energy_flight	Energy cost of flight	kJ per day	1400.74	7361.72	3581.34		3113.85
energy_searest	Energy cost of resting at sea	kJ per day	400.57	810.28	646.15		461.24
energy_forage	Energy cost of foraging	kJ per day	1400.74	1894.9	1421.45		974.97
energy_warming	Energy cost of warming food	kJ per day	34.15	65.07	47.317		35.84
chick_mass_a	maximum chick mass gain per day	g	11	9	7		6
adult_mass_KG	Energy density of the bird's tissue (kJ g−1)	Kj g-1	38	38	38		38
beta	Survival metrics parameter		0.038	1.03	NA		1.03

9.1 Parameterisation of the Type II Functional Response Curve

Empirical data on the relationship between prey availability and intake rate is not available for these species of seabirds. Therefore, we parameterised the functional response using data from time-activity budgets of individuals from each species for the average number of foraging trips per day and the average amount of time spent foraging (Table B.2). We implemented the widely used Michaelis-Menton form of the Type II functional response formula to simulate intake rates over a range of values for the model parameter (*IR_HALF_a*; *IR_MAX* was derived from empirical data) across variation in prey levels. The value of *IR_HALF_a* was then set so as to match the desired number of trips and time spent foraging to reach the individual's DER based on the summaries of empirical data (Table B.2: 'parameters set to achieve:'), with the minimum and maximum time spent foraging constrained to be within those observed from empirical data (Table B.3). No interference competition was assumed when calibrating the functional response.

The value for parameter *IR_HALF_b* (which controls the effect of conspecifics at the same foraging location on intake rate) was set to 0.02 for all species based on expert judgement.

Table B.2: Empirical data from time-activity budgets used to parameterise the functional response curve for each species, relating the intake rate of individuals to prey availability at the chosen foraging location.

	Black-legged kittiwake	**Common guillemot**	**Razorbill**	**Atlantic puffin**
Mean number of trips per 24 hours	1.9	2.0	2.35	3.3
Foraging hours per day (24 hours)	3.2	5.6	5.1	6.5
Flying hours per day (24 hours)	4.9	0.8	1.9	2.4
Parameters set to achieve:	3 foraging trips over model time step (36 hours) lasting in total 4.8 hours	2 foraging trips over model time step (24 hours) lasting in total 5.0 hours	2 foraging trips over model time step (24 hours) lasting in total 5.0 hours	3 foraging trips over model time step (24 hours) lasting in total 6.0 hours

Table B.3: Required time to spend foraging for a given number of trips (*n*=1 to 5) to reach Daily Energy Requirements (DER) for each species under the three classifications of prey availability (poor, moderate and good). These values are the result of calibrating the parameters of the functional response to achieve the best match in foraging time versus number of trips as that seen in empirical data for each species. 'NA' means it is not possible for the individual to meet its DER under the specified number of trips (i.e., the time taken to acquire DER is longer than the model time step).

		Number of trips				
		1	**2**	**3**	**4**	**5**
Prey level	**Species**	**Time to acquire DER (hrs)**				
G	KW	5.8	4.2	3.9	3.8	3.8
M	KW	11.1	5.3	4.8	4.6	4.5
P	KW	NA	10.7	8.2	7.5	7.2
G	GU	7.1	4.9	4.6	4.4	4.3
M	GU	11.5	5.9	5.4	5.1	5.0
P	GU	NA	7.5	6.6	6.2	6.0
G	RZ	5.7	4.0	3.7	3.6	3.5
M	RZ	10.5	4.9	4.5	4.3	4.2
P	RZ	NA	6.8	5.8	5.5	5.3
G	PU	7.3	5.3	4.9	4.7	4.7
M	PU	13.7	6.9	6.3	6.0	5.8
P	PU	NA	10.5	8.8	8.2	7.9

10 Appendix C - Detailed Parameterisation of the Intra-Specific Competition Effect

The effect of intra-specific competition upon *IR_HALF_a* is assumed to be equal to

$$y = ax^{IR\_HALF\_b}$$

where $y =$ value of *IR_HALF_a* with competition, $a =$ value of *IR_HALF*_a without competition, and $x =$ total number of birds within grid cell (summed across all colonies).

We reparameterise this to be

$$y = IR\_HALF\_a\left(\frac{x}{pq}\right)^{IR\_HALF\_b}$$

where $p =$ proportion of the total population that is included within simulation run and q = typical number of birds per grid cell for this species. Parameter q is calculated as the summed population across all colonies divided by the total foraging area used by each species.

It can be seen that this is a reparameterisation of the original model, with the original model parameters obtainable by setting:

$$a = (pq)^{IR\_HALF\_b} * IR\_HALF\_a$$

The advantage of the reparameterisation is that:

I. the original parameters are dependent upon the proportion of the total population included within the simulation model, but the reparameterised parameters are not. This improves the biological interpretation of IR_HALF_b (so that it now relates to our best estimate of the “true” number of birds within the grid cell), and means that the value of IR_HALF_b will now be independent of the proportion of the population being simulated.

II. the new parameters (IR_HALF_a, IR_HALF_b) should be less correlated than the original parameters, (a, IR_HALF_b), and this should make the calibration process that relates the functional response to the time spent foraging and the number of trips more straightforward (Appendix A).

11 Appendix D - Source List for Model Parameters Derived from Empirical Eata and Published Studies

Guillemot

Birkhead, T.R. & Nettleship, D.N. (1984) Alloparental care in the common murre (Uria aalge). Canadian Journal of Zoology 62: 2121-2124

Birkhead, T.R. & Nettleship, D.N. (1987) Ecological relationships between common murres Uria aalge and thick-billed murres Uria lomvia at the Gannet Islands, Labrador. Canadian Journal of Zoology 65: 1621-1629

Croll, D. A. & McLaren, E. (1993) Diving metabolism and thermoregulation in common and thick-billed murres. J. Comp. Physiol. B, 163, 160–6.

Daunt, F. & Wanless, S. (2008) Determining marine Special Protection Areas (mSPAs) for breeding seabirds. Report to RSPB.

Enstipp, M.R., Daunt, F., Wanless, S., Humphreys, E., Hamer, K.C., Benvenuti, S., & Gremillet, D. (2006). Foraging energetics of North Sea birds confronted with fluctuating prey availability. In: Top predators in marine ecosystems: their role in monitoring and management. (eds I.L. Boyd, S. Wanless & K. Camphuysen). Cambridge University Press, Cambridge, pp191-210.

Grémillet, D., Wright, G., Lauder, A., Carss, D.N. & Wanless, S. (2003) Modelling the daily food requirements of wintering great cormorants: a bioenergetics tool for wildlife management. J. Appl. Ecol. 40:266-277

Harris, M.P., Newell, M., Daunt, F., Speakman, J.R. & Wanless, S. (2008) Snake pipefish *Entelurus aequoreus* are poor food for seabirds. Ibis 150:413-415

Harris MP, Wanless S (1985) Fish fed to young guillemots, Uria aalge, and used in display on the Isle of May, Scotland. J Zool 207: 441-458

Hilton GM, Ruxton GD, Furness RW, Houston DC (2000a) Optimal digestion strategies in seabirds: a modeling approach. Evol. Ecol. Res 2: 207-230.

Hilton GM, Furness RW, Housten DC (2000b) A comparative study of digestion in North Atlantic seabirds. J Avian Biol 31: 36-46

Pennycuick CJ (1989). Bird flight performance: a practical calculation manual. Oxford Univ. Press, Oxford.

Thaxter, C.B., Daunt, F., Grémillet, D., Harris, M.P., Benvenuti, S., Watanuki, Y., Hamer, K.C. & Wanless, S. (2013). Modelling the effects of prey size and distribution on prey capture rates of two sympatric marine predators. PLoS ONE 8(11): e79915. https://doi.org/10.1371/journal.pone.0079915

Thaxter, C.B., Wanless, S., Daunt, F., Harris, M.P., Benvenuti, S., Watanuki, Y., Grémillet, D & Hamer, K.C. (2010) Influence of wing loading on trade-off between pursuit-diving and flight in common guillemots and razorbills. Journal of Experimental Biology 213: 1018-1025

Wanless, S., Daunt, F., Camphuysen, C.J., Humphreys, E., Scott, B. & Wanless, S. (2005) Setting the scene: seabird foraging behaviour, diet and breeding success. In: Final report to EU, 'Interactions between the Marine environment, PREdators, and prey: implications for Sustainable Sandeel fisheries (IMPRESS; Q5RS-2000-30864)' (Ed. Camphuysen. C.J.) pp88-134

Razorbill

Birt-Friesen WL, Montevecchi WA, Cairns DK, Macko SA (1989) Activity-specific metabolic rates of free-living northern gannets and other seabirds. Ecology 70: 357-36

Croll, D. A. & McLaren, E. (1993) Diving metabolism and thermoregulation in common and thick-billed murres. J. Comp. Physiol. B, 163, 160–6.

Enstipp, M.R., Daunt, F., Wanless, S., Humphreys, E., Hamer, K.C., Benvenuti, S., & Gremillet, D. (2006). Foraging energetics of North Sea birds confronted with fluctuating prey availability. In: Top predators in marine ecosystems: their role in monitoring and management. (eds I.L. Boyd, S. Wanless & K. Camphuysen). Cambridge University Press, Cambridge, pp191-210.

Grémillet, D., Wright, G., Lauder, A., Carss, D.N. & Wanless, S. (2003) Modelling the daily food requirements of wintering great cormorants: a bioenergetics tool for wildlife management. J. Appl. Ecol. 40:266-277

Harris, M.P., Newell, M., Daunt, F., Speakman, J.R. & Wanless, S. (2008) Snake pipefish *Entelurus aequoreus* are poor food for seabirds. Ibis 150:413-415

Hilton GM, Ruxton GD, Furness RW, Houston DC (2000a) Optimal digestion strategies in seabirds: a modeling approach. Evol. Ecol. Res 2: 207-230.

Hilton GM, Furness RW, Housten DC (2000b) A comparative study of digestion in North Atlantic seabirds. J Avian Biol 31: 36-46

Hipfner, J.M. (2000) The effect of egg size on post-hatching development in the Razorbill: an experimental study. Journal of Avian Biology 31: 112-118

Pennycuick CJ (1989). Bird flight performance: a practical calculation manual. Oxford Univ. Press, Oxford.

Thaxter, C.B., Daunt, F., Grémillet, D., Harris, M.P., Benvenuti, S., Watanuki, Y., Hamer, K.C. & Wanless, S. (2013). Modelling the effects of prey size and distribution on prey capture rates of two sympatric marine predators. PLoS ONE 8(11): e79915. https://doi.org/10.1371/journal.pone.0079915

Thaxter, C.B., Wanless, S., Daunt, F., Harris, M.P., Benvenuti, S., Watanuki, Y., Grémillet, D & Hamer, K.C. (2010) Influence of wing loading on trade-off between pursuit-diving and flight in common guillemots and razorbills. Journal of Experimental Biology 213: 1018-1025

Kittiwake

Baird, P.N. (1994) Black-legged kittiwake *Rissa tridactyla*. The Birds of North America No. 92.

Daunt, F., Wanless, S., Peters, G., Benvenuti, S., Sharples, J., Gremillet, D. & Scott, B. (2006). Impacts of oceanography on the foraging dynamics of seabirds in the North Sea. In: Top predators in marine ecosystems: their role in monitoring and management. (eds I.L. Boyd, S. Wanless & K. Camphuysen). Cambridge University Press, Cambridge, pp177-190.

Enstipp, M.R., Daunt, F., Wanless, S., Humphreys, E., Hamer, K.C., Benvenuti, S., & Gremillet, D. (2006). Foraging energetics of North Sea birds confronted with fluctuating prey availability. In: Top predators in marine ecosystems: their role in monitoring and management. (eds I.L. Boyd, S. Wanless & K. Camphuysen). Cambridge University Press, Cambridge, pp191-210.

Grémillet, D., Wright, G., Lauder, A., Carss, D.N. & Wanless, S. (2003) Modelling the daily food requirements of wintering great cormorants: a bioenergetics tool for wildlife management. J. Appl. Ecol. 40:266-277

Harris, M.P., Newell, M., Daunt, F., Speakman, J.R. & Wanless, S. (2008) Snake pipefish *Entelurus aequoreus* are poor food for seabirds. Ibis 150:413-415

Hilton GM, Furness RW, Housten DC (2000b) A comparative study of digestion in North Atlantic seabirds. J Avian Biol 31: 36-46

Humphreys. E.M. (2002) Energetics of spatial exploitation of the North Sea by kittiwakes breeding on the Isle of May, Scotland. Unpublished PhD thesis, University of Stirling

Nelson, B. (2013) Early warnings of climate change on ecosystems: hormonally-mediated life-history decisions in seabirds. Unpublished PhD thesis, University of Glasgow.

Pennycuick CJ (1989). Bird flight performance: a practical calculation manual. Oxford Univ. Press, Oxford.

Visser GH (2002). Chick growth and development in seabirds. In: Biology of Marine Birds, pp 439-465. Schreiber EA and Burger J (Eds.); CRC Press, Boca Raton, 722 pp.

Puffin

Birkhead, T.R. & Nettleship, D.N. (1984) Alloparental care in the common murre (Uria aalge). Canadian Journal of Zoology 62: 2121-2124

Birt-Friesen WL, Montevecchi WA, Cairns DK, Macko SA (1989) Activity-specific metabolic rates of free-living northern gannets and other seabirds. Ecology 70: 357-36

Croll, D. A. & McLaren, E. (1993) Diving metabolism and thermoregulation in common and thick-billed murres. J. Comp. Physiol. B, 163, 160–6.

Daunt, F. & Wanless, S. (2008) Determining marine Special Protection Areas (mSPAs) for breeding seabirds. Report to RSPB.

Enstipp, M.R., Daunt, F., Wanless, S., Humphreys, E., Hamer, K.C., Benvenuti, S., & Gremillet, D. (2006). Foraging energetics of North Sea birds confronted with fluctuating prey availability. In: Top predators in marine ecosystems: their role in monitoring and management. (eds I.L. Boyd, S. Wanless & K. Camphuysen). Cambridge University Press, Cambridge, pp191-210.

Grémillet, D., Wright, G., Lauder, A., Carss, D.N. & Wanless, S. (2003) Modelling the daily food requirements of wintering great cormorants: a bioenergetics tool for wildlife management. J. Appl. Ecol. 40:266-277

Harris, M.P., Newell, M., Daunt, F., Speakman, J.R. & Wanless, S. (2008) Snake pipefish *Entelurus aequoreus* are poor food for seabirds. Ibis 150:413-415

Harris, M.P. & Wanless, S. (2011) The Puffin. T & A.D. Poyser.

Hilton GM, Ruxton GD, Furness RW, Houston DC (2000a) Optimal digestion strategies in seabirds: a modeling approach. Evol. Ecol. Res 2: 207-230.

Hilton GM, Furness RW, Housten DC (2000b) A comparative study of digestion in North Atlantic seabirds. J Avian Biol 31: 36-46

Pennycuick CJ (1989). Bird flight performance: a practical calculation manual. Oxford Univ. Press, Oxford.

12 Appendix E - Algorithm for Simulating the Number of Birds within a Snapshot

For the k-th snapshot:

Step 1 - Choose a day, $j(k)$, on which to undertake the snapshot. Within the tool the choice of day is specified by the user.

Step 2 - Use the model output to calculate the proportion of time that each bird, i, is simulated to spend foraging within the ORD ($p_{ij(k)}$) and flying across the ORD ($q_{ij(k)}$). Note that these proportions will both be zero for birds that never interact with the ORD.

Step 3 - Simulate the behaviour of the individual at a random time within the day by simulating from a multinomial distribution with sample size one:

$$y_{ik} \sim \text{Multinomial}\left(1, \left(1 - p_{ij(k)} - q_{ij(k)}, p_{ij(k)}, q_{ij(k)}\right)\right)$$

This variable takes the value two if the bird is foraging within the ORD at the time of the snapshot, the value three if it is flying across the ORD, and the value one otherwise.

Step 4 - Calculate the total number of birds foraging within the ORD at the time of the snapshot to be

$$o_k = \sum_{i=1}^{n} I(y_{ik} = 2)$$

and the total number of birds flying across the ORD at the time of the snapshot to be

$$l_k = \sum_{i=1}^{n} I(y_{ik} = 3)$$

13 Appendix F - Results for Other Species

13.1 Razorbill Results

For Razorbills we present results for the Forth Islands based on mapped bird densities and prey availability derived from local GPS tracking data (Method i: Mapped older GPS tracking data flight removed). We completed ten paired simulations of the fictional ORDs assessed cumulatively (WFA+WFB+WFC):

- Mapped bird density and mapped prey based on local GPS data with flight removed
- Barrier type: perimeter
- Probability of displacement: 0.6
- Probability of barrier effect: 1.0 (all displacement-susceptible birds are also subject to barrier effects)
- ORD footprint border: 0.5km
- 10 paired simulations of matched baseline-ORD runs
- 100% of the total population
- Forth Islands in the Forth-Tay region

Table 13-1: For bird densities and prey availability derived from local GPS tracking data, the percent mass loss (%) of adult birds during the chick-rearing period in baseline runs (no ORDs present) and their corresponding classifications into ‘good’ (G, dark green cells), ‘moderate’ (M, light green cells) and ‘poor’ (P) prey conditions. Colonies: ‘FI’ = Forth Islands.

	230		**250**		**270**		**280**	
Colony	Type	%	Type	%	Type	%	Type	%
FI	M	10.4	M	10.1	M	8.7	M	7.4

13.1.1 Overall population-Level Effect (P1)

Table 13-2: Population-level impact (P1) of the fictitious ORDs upon razorbill adult and chick mortality rates for all birds in the simulated population (100% of total population) at the Forth Islands SPA in the Forth Tay region. All runs performed with 100% of the population using ten matched pairs of runs, and assume a 0.5 km border around each footprint. The impact is defined as the percent additional mortality due to the wind farm (percentage points; e.g., 0.1 is 0.1% additional mortality; and 1.5 is 1.5% additional mortality). A positive value implies an increase in additional mortality when the WF is present. Table shows cumulative effects for results for bird and prey densities derived from local GPS data when all three ORDs were included (WFA+WFB+WFC Mapped).

Forth Islands WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Total number of birds simulated (100% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
230 (M)	6934	0.08 (-0.07, 0.22)	0.85 (0.28, 1.43)
250 (M)	6934	0.11 (-0.09, 0.30)	1.84 (1.21, 2.47)
270 (M)	6934	0.17 (-0.11, 0.46)	0.91 (0.48, 1.34)
280 (M)	6934	0.17 (-0.01, 0.35)	0.34 (0.23, 0.45)

13.1.2 Metrics on the Fate of Individual Birds

Table 13-3: Impact of all three wind farms upon adult mortality rates for birds that are directly impacted by the wind farm at any point during the breeding season in terms of either displacement or barrier effects (“Dir I2”) and those that are not (“Non I1”). The number of birds in each category is presented along with the percentage of the population in brackets. Results are based on ten paired runs of 100% of the total population, and relate to razorbills from Forth Islands SPA in the Forth Tay region. Impact is defined as the mean percent additional mortality for each set of birds with associated 95% prediction intervals. Positive values represent an increase in mortality associated with the impact, negative values represent a decrease in mortality associated with the impact. Table shows cumulative effects for results for bird and prey densities derived from local GPS data when all three ORDs were included (WFA+WFB+WFC).

Forth Islands WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
230 (M)	4001.6 (57.7%)	2932.4 (42.3%)	0.18 (-0.09, 0.44)	-0.06 (-0.17, 0.05)
250 (M)	4001.6 (57.7%)	2932.4 (42.3%)	0.19 (-0.15, 0.53)	-0.01 (-0.08, 0.07)
270 (M)	4001.6 (57.7%)	2932.4 (42.3%)	0.30 (-0.20, 0.81)	-0.01 (-0.07, 0.06)
280 (M)	4001.6 (57.7%)	2932.4 (42.3%)	0.29 (-0.02, 0.59)	0.003 (-0.06, 0.06)

Table 13-4: Mapped older GPS tracking data flight removed Impact of each wind farm upon adult and chick mortality rates for: a) birds that experience displacement but not barrier effects ("Disp", I3), b) birds that experience barrier but not displacement effects ("Bar", I4), c) birds that experience both ("Both", I5). Results are based on ten paired runs of 100% of the total population, and relate only to razorbills from the Forth Islands SPA in the Forth Tay region. Impact is defined as the mean percent additional mortality for each set of birds with associated 95% prediction intervals. Positive values represent an increase in mortality associated with the impact, negative values represent a decrease in mortality associated with the impact. Table shows cumulative effects for results for bird and prey densities derived from local GPS data when all three ORDs were included (WFA+WFB+WFC Mapped).

Forth Islands WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
230 (M)	120.7 (1.7%)	1644.5 (23.7%)	2236.4 (32.3%)	0.23 (-1.81, 2.27)	0.11 (-0.34, 0.55)	0.23 (-0.21, 0.67)
250 (M)	118.4 (1.7%)	1644.5 (23.7%)	2238.7 (32.3)	0.33 (-1.22, 1.88)	0.18 (-0.33, 0.70)	0.19 (-0.26, 0.64)
270 (M)	117.7 (1.7%)	1644.5 (23.7%)	2239.4 (32.3%)	0.50 (-1.16, 2.16)	0.21 (-0.38, 0.79)	0.36 (-0.39, 1.12)
280 (M)	119.5 (1.7%)	1644.5 (23.7%)	2237.6 (32.3%)	0.17 (-0.70, 1.05)	0.31 (-0.18, 0.80)	0.28 (-0.05, 0.60)

13.2 Guillemot Results

For common guillemots we present results for all SPA colonies in the Forth-Tay region based on mapped bird densities and prey availability derived from local GPS tracking data (Method i: Mapped older GPS tracking data flight removed). We completed ten paired simulations of the fictional ORDs assessed cumulatively (WFA+WFB+WFC):

- Mapped bird density and mapped prey based on local GPS data with flight removed;
- Barrier type: perimeter;
- Probability of displacement: 0.6;
- Probability of barrier effect: 1.0 (all displacement-susceptible birds are also subject to barrier effects);
- ORD footprint border: 0.5 km;
- 10 paired simulations of matched baseline-ORD runs;
- 30% of the total population;
- All SPAs in the Forth-Tay region.

Table 13-5: For bird densities and prey availability derived from local GPS tracking data, the percent mass loss (%) of adult birds during the chick-rearing period in baseline runs (no ORDs present) and their corresponding classifications into 'good' (G, dark green cells), 'moderate' (M, light green cells) and 'poor' (P) prey conditions. Colonies: 1 = Buchan Ness, 2 = Fowlsheugh, 3 = Forth Islands, 4 = St Abbs Head.

	260		**275**		**290**		**320**	
Colony	Type	%	Type	%	Type	%	Type	%
1	M	10.57	M	10.32	M	9.71	G	0.28
2	M	10.47	M	10.27	M	9.73	G	3.26
3	M	10.11	M	9.09	M	6.28	G	1.65
4	G	1.27	G	0.51	G	0.21	G	0.03

13.2.1 Overall Population-Level Effect (P1)

Table 13-6: Population-level impact (P1) of all three fictional ORDs upon common guillemot adult and chick mortality rates for all birds in the simulated population (30% of total population) at all SPA colonies in the Forth Tay region. All runs performed with 30% of the population using ten matched pairs of runs, and assume a 0.5 km border around each footprint. The impact is defined as the percent additional mortality due to the wind farm (percentage points; e.g., 0.1 is 0.1% additional mortality; and 1.5 is 1.5% additional mortality). A positive value implies an increase in additional mortality when the WF is present. Table shows cumulative effects for results for bird and prey densities derived from local GPS data when all three ORDs were included (WFA+WFB+WFC Mapped).

Buchan Ness WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	**Total number of birds simulated (30% of total population)**	**Mean impact on adult mortality (percentage points, 95% prediction interval)**	**Impact on chick mortality (percentage points, 95% prediction interval)**
260 (M)	10652.0	0.00	0.00
275 (M)	10652.0	0.00	0.00
290 (M)	10652.0	0.00	0.00
320 (G)	10652.0	0.00	0.00

Fowlsheugh WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

	Total number of birds simulated (30% of total population)	**Mean impact on adult mortality (percentage points, 95% prediction interval)**	**Impact on chick mortality (percentage points, 95% prediction interval)**
260 (M)	22366.0	0.00	0.00
275 (M)	22366.0	0.00	0.00
290 (M)	22366.0	0.00	0.00
320 (G)	22366.0	0.00	0.00

Forth Islands WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

	Total number of birds simulated (30% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
260 (M)	11934.0	0.003 (-0.11, 0.12)	2.23 (1.78, 2.69)
275 (M)	11934.0	0.17 (0.10, 0.25)	3.25 (2.77, 3.74)
290 (M)	11934.0	0.31 (0.16, 0.47)	0.50 (0.35, 0.65)
320 (G)	11934.0	0.18 (0.07, 0.30)	0.00

St Abbs Head WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

	Total number of birds simulated (30% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
260 (G)	17448.0	0.003 (-0.004, 0.01)	0.00
275 (G)	17448.0	0.002 (-0.01, 0.01)	0.00
290 (G)	17448.0	0.002 (-0.01, 0.01)	0.00
320 (G)	17448.0	0.00	0.00

13.2.2 Metrics on the Fate of Individual Birds

Table 13-7: Impact of each wind farm upon adult mortality rates for birds that are directly impacted by the wind farm at any point during the breeding season in terms of either displacement or barrier effects (“Dir I2”) and those that are not (“Non I1”). The number of birds in each category is presented along with the percentage of the population in brackets. Results are based on ten paired runs of 30% of the total population, and relate to common guillemots from all SPA colonies in the Forth Tay region. Impact is defined as the mean percent additional mortality for each set of birds with associated 95% prediction intervals. Positive values represent an increase in mortality associated with the impact, negative values represent a decrease in mortality associated with the impact. Table shows cumulative effects for results for bird and prey densities derived from local GPS data when all three ORDs were included (WFA+WFB+WFC).

Buchan Ness WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
260 (M)	0	10652 (100%)	0.00	0.00
275 (M)	0	10652 (100%)	0.00	0.00
290 (M)	0	10652 (100%)	0.00	0.00
320 (G)	0	10652 (100%)	0.00	0.00

Fowlsheugh WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
260 (M)	2.0 (<1%)	22364.6 (>99%)	0.00	0.00
275 (M)	2.0 (<1%)	22364.6 (>99%)	0.00	0.00
290 (M)	2.0 (<1%)	22364.6 (>99%)	0.00	0.00
320 (G)	2.0 (<1%)	22364.6 (>99%)	0.00	0.00

Forth Islands WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
260 (M)	6184.3 (51.8%)	5749.7 (48.2%)	0.05 (-0.15, 0.25)	-0.05 (-0.16, 0.06)
275 (M)	6184.3 (51.8%)	5749.7 (48.2%)	0.35 (0.17, 0.52)	-0.01 (-0.07, 0.05)
290 (M)	6184.3 (51.8%)	5749.7 (48.2%)	0.60 (0.31, 0.89)	0.01 (-0.03, 0.04)
320 (G)	6184.3 (51.8%)	5749.7 (48.2%)	0.36 (0.13, 0.59)	-0.01 (-0.05, 0.03)

St Abbs Head WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
260 (G)	1430.0 (8.2%)	16018.0 (91.8%)	0.04 (-0.12, 0.21)	0.00 (-0.01, 0.01)
275 (G)	1430.0 (8.2%)	16018.0 (91.8%)	0.03 (-0.12, 0.17)	0.00
290 (G)	1430.0 (8.2%)	16018.0 (91.8%)	0.03 (-0.09, 0.15)	0.00
320 (G)	1430.0 (8.2%)	16018.0 (91.8%)	0.00	0.00

Table 13-8: Impact of each wind farm upon adult and chick mortality rates for: a) birds that experience displacement but not barrier effects (“Disp”, I3), b) birds that experience barrier but not displacement effects (“Bar”, I4), c) birds that experience both (“Both”, I5). Results are based on ten paired runs of 30% of the total population, and relate only to common guillemots from SPA colonies in the Forth Tay region. Impact is defined as the mean percent additional mortality for each set of birds with associated 95% prediction intervals. Positive values represent an increase in mortality associated with the impact, negative values represent a decrease in mortality associated with the impact. Table shows cumulative effects for results for bird and prey densities derived from local GPS data when all three ORDs were included (WFA+WFB+WFC Mapped).

Buchan Ness WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
260 (M)	0	0	0	0	0	0
275 (M)	0	0	0	0	0	0
290 (M)	0	0	0	0	0	0
320 (G)	0	0	0	0	0	0

Fowlsheugh WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
260 (M)	1.7	1.0	1.0	0.00	0.00	0.00
275 (M)	1.7	1.0	1.0	0.00	0.00	0.00
290 (M)	1.7	1.0	1.0	0.00	0.00	0.00
320 (G)	1.7	1.0	1.0	0.00	0.00	0.00

Forth Islands WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
260 (M)	639.1 (5.4%)	2373.8 (19.9%)	3171.4 (26.6%)	0.06 (-0.50, 0.62)	0.05 (-0.34, 0.43)	0.05 (-0.27, 0.38)
275 (M)	641.5 (5.4%)	2373.8 (19.9%)	3169.0 (26.6%)	-0.10 (-0.66, 0.46)	0.40 (-0.06, 0.85)	0.40 (0.12, 0.67)
290 (M)	639.3 (5.4%)	2373.8 (19.9%)	3171.2 (26.6%)	0.11 (-0.20, 0.42)	0.60 (0.27, 0.92)	0.70 (0.36, 1.05)
320 (G)	635.3 (5.4%)	2373.8 (19.9%)	3175.2 (26.6%)	0.06 (-0.30, 0.42)	0.32 (0.04, 0.59)	0.45 (0.13, 0.77)

St Abbs Head WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category			Impact on adult mortality (percentage points)		
	Disp I3	Bar I4	Both I5	Disp I3	Bar I4	Both I5
260 (G)	857.3 (4.9%)	394.0 (2.3%)	178.7 (1.0%)	0.04 (-0.10, 0.17)	0.03 (-0.17, 0.22)	0.11 (-0.72, 0.94)
275 (G)	857.3 (4.9%)	394.0 (2.3%)	182.3 (1.0%)	0.001 (-0.13, 0.14)	0.08 (-0.21, 0.37)	0.05 (-0.33, 0.43)
290 (G)	854.8 (4.9%)	394.0 (2.3%)	181.2 (1.0%)	0.00	0.10 (-0.32, 0.52)	0.00
320 (G)	854.1 (4.9%)	394.0 (2.3%)	181.9 (1.0%)	0.00	0.00	0.00

13.3 Atlantic Puffin Results

For Atlantic puffins we present results for the Forth Islands based on mapped bird densities and prey availability derived from local GPS tracking data with flight removed (method i: Mapped older GPS tracking data flight removed). We completed ten paired simulations of the fictional ORDs assessed cumulatively (WFA+WFB+WFC):

- Mapped bird density and mapped prey based on local GPS data with flight removed;
- Barrier type: perimeter;r
- Probability of displacement: 0.6;
- Probability of barrier effect: 1.0 (all displacement-susceptible birds are also subject to barrier effects);
- ORD footprint border: 0.5 km;
- 10 paired simulations of matched baseline-ORD runs;
- 30% of the total population;
- Forth Islands in the Forth-Tay region.

Table 13-9: For bird densities and prey availability derived from local GPS tracking data, the percent mass loss (%) of adult birds during the chick-rearing period in baseline runs (no ORDs present) and their corresponding classifications into ‘good’ (G, dark green cells), ‘moderate’ (M, light green cells) and ‘poor’ (P) prey conditions. Colonies: ‘FI’ = Forth Islands.

	130		145		175	
Colony	Type	%	Type	%	Type	%
FI	M	8.73	M	5.25	G	0.45

13.3.1 Overall Population-Level Effect (P1)

Table 13-10: Population-level impact (P1) of the fictitious ORDs upon Atlantic puffin adult and chick mortality rates for all birds in the simulated population (100% of total population) at the Forth Islands SPA in the Forth Tay region. All runs performed with 30% of the population using ten matched pairs of runs, and assume a 0.5 km border around each footprint. The impact is defined as the percent additional mortality due to the wind farm (percentage points; e.g., 0.1 is 0.1% additional mortality; and 1.5 is 1.5% additional mortality). A positive value implies an increase in additional mortality when the WF is present. Table shows cumulative effects for results for bird and prey densities derived from local GPS data when all three ORDs were included (WFA+WFB+WFC Mapped).

Forth Islands WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Total number of birds simulated (30% of total population)	Mean impact on adult mortality (percentage points, 95% prediction interval)	Impact on chick mortality (percentage points, 95% prediction interval)
130 (M)	31374	1.92 (1.75, 2.09)	1.37 (1.19, 1.55)
145 (M)	31374	2.41 (2.23, 2.60)	0.05 (0.02, 0.09)
175 (G)	31374	0.98 (0.82, 1.14)	0.00

13.3.2 Metrics on the Fate of Individual Birds

Table 13-11: Impact of all three wind farms upon adult mortality rates for birds that are directly impacted by the wind farm at any point during the breeding season in terms of either displacement or barrier effects (“Dir I2”) and those that are not (“Non I1”). The number of birds in each category is presented along with the percentage of the population in brackets. Results are based on ten paired runs of 30% of the total population, and relate to Atlantic puffin from Forth Islands SPA in the Forth Tay region. Impact is defined as the mean percent additional mortality for each set of birds with associated 95% prediction intervals. Positive values represent an increase in mortality associated with the impact, negative values represent a decrease in mortality associated with the impact. Table shows cumulative effects for results for bird and prey densities derived from local GPS data when all three ORDs were included (WFA+WFB+WFC).

Forth Islands WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	Mean Number of birds per category		Impact on adult mortality (percentage points)	
	Dir I2 (%)	Non I1 (%)	Dir (I2)	Non (I1)
130 (M)	18791.0 (59.9%)	12583.0 (40.1%)	3.23 (2.95, 3.51)	-0.05 (-0.15, 0.06)
145 (M)	18791.0 (59.9%)	12583.0 (40.1%)	4.05 (3.77, 4.34)	-0.04 (-0.12, 0.04)
175 (G)	18791.0 (59.9%)	12583.0 (40.1%)	1.63 (1.38, 1.88)	0.001 (-0.02, 0.03)

Table 13-12: Impact of each wind farm upon adult and chick mortality rates for: a) birds that experience displacement but not barrier effects ("Disp", I3), b) birds that experience barrier but not displacement effects ("Bar", I4), c) birds that experience both ("Both", I5). Results are based on ten paired runs of 30% of the total population, and relate only to Atlantic puffin from the Forth Islands SPA in the Forth Tay region. Impact is defined as the mean percent additional mortality for each set of birds with associated 95% prediction intervals. Positive values represent an increase in mortality associated with the impact, negative values represent a decrease in mortality associated with the impact. Table shows cumulative effects for results for bird and prey densities derived from local GPS data when all three ORDs were included (WFA+WFB+WFC Mapped).

Forth Islands WFA+WFB+WFC (Mapped older GPS tracking data flight removed):

PREY LEVEL	**Mean Number of birds per category**			**Impact on adult mortality (percentage points)**		
	Disp I3	**Bar I4**	**Both I5**	**Disp I3**	**Bar I4**	**Both I5**
130 (M)	0	52.2 (<1%)	18738.8 (>99%)	0	1.97 (-1.98, 5.92)	3.24 (2.96, 3.52)
145 (M)	0	52.2 (<1%)	18738.8 (>99%)	0	2.20 (-3.62, 8.01)	4.06 (3.78, 4.34)
175 (G)	0	52.2 (<1%)	18738.8 (>99%)	0	1.68 (-1.57, 4.92)	1.63 (1.38, 1.88)

NERC SCIENCE OF THE ENVIRONMENT

PEER

INVESTORS IN PEOPLE

Athena SWAN Bronze Award

BANGOR
Centre for Ecology & Hydrology
Environment Centre Wales
Deiniol Road
Bangor
Gwynedd
LL57 2UW
United Kingdom
T: +44 (0)1248 374500
F: +44 (0)1248 362133

EDINBURGH
Centre for Ecology & Hydrology
Bush Estate
Penicuik
Midlothian
EH26 0QB
United Kingdom
T: +44 (0)131 4454343
F: +44 (0)131 4453943

LANCASTER
Centre for Ecology & Hydrology
Lancaster Environment Centre
Library Avenue
Bailrigg
Lancaster
LA1 4AP
United Kingdom
T: +44 (0)1524 595800
F: +44 (0)1524 61536

WALLINGFORD - Headquarters
Centre for Ecology & Hydrology
Maclean Building
Benson Lane
Crowmarsh Gifford
Wallingford
Oxfordshire
OX10 8BB
United Kingdom
T: +44 (0)1491 838800
F: +44 (0)1491 692424

enquiries@ceh.ac.uk
www.ceh.ac.uk