Spiorad na Mara Offshore Wind Farm

Offshore Project

Environmental Impact Assessment Report

Annex 14.2.2: SeabORD Report, Volume 2c

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1 INTRODUCTION

1.1 OVERVIEW

1.1.1.1 This annex of the **Appendix 14.2: Displacement Report, Volume 2c** of the Environmental Impact Assessment Report (EIAR) presents the SeabORD analysis undertaken for the proposed offshore elements of the Spiorad na Mara Offshore Wind Farm (hereafter referred to as 'the Offshore Project') with respect to Marine and Nearshore Ornithology.

1.1.1.2 At the time of writing, NatureScot currently advises that, where feasible, outputs from the MATLAB version of SeabORD may be used to provide context to displacement assessments. However, the matrix approach remains the primary method for all species and seasons. Accordingly, the EIAR assessment relies on the displacement matrices and mortality estimates presented in **Appendix 14.2, Volume 2c**, while the SeabORD results included here are provided solely to inform contextual discussion within the **Offshore Report to Inform Appropriate Assessment (RIAA)**

1.1.1.3 This annex should be read in conjunction with the **EIAR** and **Offshore RIAA** and the relevant parts of the following chapters and appendices:

- **Appendix 5.2: Response to Scoping Opinion, Volume 1c;**
- **Chapter 14: Marine and Nearshore Ornithology, Volume 2a;**
- **Appendix 14.1: Ornithology Baseline Report, Volume 2c;**
- **Appendix 14.2, Volume 2c;**
- **Appendix 14.6 : EIA Ornithology Consultation, Volume 2c;**
- **Offshore RIAA Appendix C: Consultation;**
- **Offshore RIAA Appendix D: Offshore Ornithology Apportioning**.

1.1.2 BACKGROUND

1.1.2.1 Seabirds can be impacted by offshore wind farm developments in a number of ways, including collision, displacement, barrier effects and disturbance, as well as indirect impacts such as changes to prey availability. Disturbance as the result of activities during the construction, Operation and Maintenance (O&M) and decommissioning phases of an offshore wind farm has the potential to displace seabirds from an area of sea in which the activity is occurring. In relation to offshore wind farm development, displacement is defined as a reduction in the number of seabirds occurring within or immediately adjacent to an offshore wind farm (Furness *et al.*, 2013).

1.1.2.2 Species differ greatly in their susceptibility to disturbance. Species sensitivity to disturbance in response to offshore wind farms has been quantified by Garthe and Hüppop (2004), Furness *et al.* (2013), Bradbury *et al.* (2014) and Wade *et al.* (2016). During the O&M phase, the presence of operational wind turbines has the potential to directly disturb seabirds leading to displacement from the Offshore Project Turbine Area, including a buffer around it. As the result of disturbance, displaced birds may move to areas already occupied by other birds and thus face higher intra/inter-specific competition due to a higher density of individuals competing for the same resource. Alternatively, displaced birds may be forced to move into areas of lower quality (e.g. areas of lower prey availability). Such disturbance and resulting displacement could ultimately affect their demographic fitness (i.e. survival rates and breeding productivity) as well as potentially impacting on other birds in areas that displaced birds move to. Changes in mortality levels of displaced birds have been established for waders (e.g. Burton *et al.*, 2006).

1.1.2.3 While there is a general lack of empirical evidence on the consequence of displacement of seabirds, particularly in terms of both their survival and productivity, modelling approaches have been used to infer potential impacts. For example, Individual-Based Models (IBMs) have been employed in studies of waterbirds such as waders, geese and seaducks to simulate changes in mortality linked to altered energy budgets (Pettifor *et al.*, 2000; West *et al.*, 2003; Kaiser *et al.*, 2002). In the context of seabirds and offshore wind farms, IBMs such as the SeabORD model are increasingly used to predict the consequences of displacement on individual fitness and population dynamics (Topping and Petersen, 2011). However, such applications remain limited compared to other taxa, partly due to data constraints and the complexity of modelling seabird behaviour and energetics at sea.

1.1.2.4 NatureScot has produced guidance specific to the assessment of displacement in Scottish waters (NatureScot, 2023). NatureScot (2023) recommends the use of both the displacement matrix approach and SeabORD.

1.1.2.5 SeabORD is an IBM developed by the UK Centre for Ecology and Hydrology to evaluate the bio-energetic costs of seabird responses to distributional changes, both at the individual and population levels. The model simulates the flight paths of individual birds from breeding colonies to potential foraging areas under scenarios with and without the presence of wind farms (Searle *et al.*, 2018). These simulations are combined with bio-energetic equations to estimate percentage body mass loss, providing insights into the birds' survival and productivity during the breeding season. Theoretical annual mortality estimates are then derived based on the predicted body masses of individuals at the end of the breeding season.

1.1.3 PURPOSE OF THIS ANNEX

1.1.3.1 This annex presents estimated mortality for kittiwake, guillemot and razorbill, during the chick-rearing period, based on outputs from the SeabORD tool. SeabORD (Searle *et al.*, 2018) was developed to predict both direct and indirect impacts of displacement and barrier effects from offshore wind farms on seabirds.

1.1.3.2 It should be noted that puffin is also included within the SeabORD tool, however due to extensive computational run times, models could not be completed within the project timeframe (see Section 2.2.1).

1.1.3.3 This annex outlines the methodology and findings used to assess seabird displacement and barrier effects during the O&M phase of the Offshore Project for inclusion within the **Offshore RIAA**. These assessments are first undertaken as part of the EIAR application, following NatureScot guidance (NatureScot, 2023), and the resulting outputs are then used directly within the **Offshore RIAA**. However, the EIAR displacement and barrier effect assessment (Sections 14.9.3 and 14.9.5 of **Chapter 14, Volume 2a**) primarily uses matrix-based approach with the SeabORD tool (Searle et al., 2018) provided for additional context, if required. Use of SeabORD for this purpose was requested within the Project Scoping Opinion (**Appendix 5.2: Response to Scoping Opinion, Volume 1c and Offshore RIAA Appendix C: Consultation**) and subsequently confirmed in writing by NatureScot (NatureScot, 2025).

1.1.3.4 This annex focusses exclusively on the use of the SeabORD tool to model displacement and barrier effects on:

- Kittiwake *Rissa tridactyla;*
- Guillemot *Uria aalge;*
- Razorbill *Alca torda*.

1.1.3.5 These species (with the addition of puffin) are currently the only ones for which SeabORD can predict distributional responses during the chick-rearing period. For the purposes of this assessment, the SeabORD chick-rearing outputs are used to represent the wider breeding season, enabling direct comparison with the matrix-based assessment once results have been apportioned to the relevant SPA and non-SPA sites. This approach is consistent with standard guidance and established practice. The corresponding matrix-based displacement estimates for the full breeding, migratory, and non-breeding periods for the same species, are presented in **Appendix 14.2, Volume 2c**, with the apportioned outputs included within the **Offshore RIAA**. This enables the SeabORD breeding-season (chick-rearing) outputs to be compared on a like-for-like basis with the breeding-season outputs from the matrix method.

1.1.3.6 The structure of this annex is as follows:

- Section 1 Introduction: Introduces the Offshore Project, summarises the purpose of this annex, and provides background on the use of SeabORD in the assessment process;
- Section 2 Methodology: Sets out the approach used for SeabORD. This includes descriptions of the input parameters and model calibration;
- Section 3 Results: Presents the quantitative SeabORD estimates for guillemot, razorbill, and kittiwake. In addition, SeabORD-derived mortality rates are presented alongside values reported in the literature, together with model runtimes;

- Section 4 Discussion: Describes the findings set out within section 3 as well as limitations around the use of SeabORD;
- Section 5 Glossary and Terms and Abbreviations: Sets out the key terms and abbreviations introduced in this annex;
- Section 6 References: Sets out the details of the reports, research papers and literature referred to in this annex.

2 METHODOLOGY

2.1 OVERVIEW

2.1.1.1 SeabORD modelling was conducted for the Offshore Project in line with NatureScot guidance (NatureScot, 2023a). Models were run using SeabORD version 1.3 (Marine Scotland Science, 2024). This is currently the most up-to-date publicly available version of the model.

2.1.1.2 The SeabORD tool can be applied using 2 approaches, depending on the availability of Global Positioning System (GPS) data for the species and colony in question:

- **GPS-informed foraging distribution**: If sufficient GPS data are available for the focal species and colony, the foraging distribution can be modelled using statistical methods such as a Generalised Additive Model (GAM), as described in Searle *et al.* (2014);

- **Distance-based foraging distribution**: When GPS data are unavailable or limited, foraging distribution is determined using a simpler distance-decay relationship, which assumes that foraging density decreases as the distance from the colony increases. Distance decay utilises Woodward *et al.* (2019) foraging ranges.

2.1.1.3 The current publicly available version of the SeabORD model (Searle *et al.*, 2018) was originally developed for use in the Forth and Tay region and therefore incorporates GPS tracking data specific to that area. As the Offshore Project lies outside the Forth and Tay region and the available version of SeabORD does not include more recent or region-specific tracking data, the model has been applied using the distance-based foraging approach. In addition, because the tool was not initially designed for the current Study Area (see Section 14.4.2 of **Chapter 14, Volume 2a** for further details), an additional setup run was undertaken to generate the appropriate spatial map region for the assessment. Calibration was also required to ensure that prey levels reflected appropriate environmental conditions (i.e. poor, moderate, and good; see Section 2.3 for further details), as is standard whenever new colony or distribution data are introduced.

2.1.1.4 The model was run on an Offshore Project-only basis, meaning that cumulative impacts including other developments in the area were not assessed. This was due to the current version of the tool only supporting 5 offshore wind farms, with the number of built and proposed offshore wind farms included within the cumulative assessment significantly exceeding this.

2.1.1.5 The following sections provide detailed information included within the SeabORD assessment:

- Section 2.2 (Input Parameters) covers colony-specific information input to the model, the modelling region for each species, the proportion of the population modelled, foraging and prey information, and the displacement and barrier effects applied;

- Section 2.3 (Model Calibration) describes the steps undertaken to calibrate the model and presents the resulting input parameters used in the final model runs;
- Section 2.3.3 (Bioenergetics in the Model) discusses the bioenergetic parameters used within SeabORD for each species. The bioenergetics framework defines how adult and chick energy requirements, body mass, and behavioural responses are represented through time, including thresholds governing foraging behaviour, breeding success, chick provisioning, and mortality;
- Section 2.4 (Annual Mortality Output) summarises the model outputs, including estimates of annual adult and chick mortality under baseline and impact scenarios, alongside additional behavioural and energetic metrics used to derive mortality estimates and quantify the potential effects of the Offshore Project.

2.2 INPUT PARAMETERS

2.2.1 COLONY AND SPECIES INFORMATION

2.2.1.1 SeabORD was run to model impacts on the breeding colonies of selected Special Protection Areas (SPAs) for each species. The choice of SPAs to model in SeabORD was based on the outputs of the Habitats Regulation Assessment (HRA) Screening Report (Spiorad na Mara Limited, 2024) and the findings of **Offshore RIAA Appendix D**. For each species, SPAs with the highest breeding-season apportioning values were selected, as these represent the colonies predicted to experience the greatest potential impacts.

2.2.1.2 The SeabORD tool can incorporate data from up to 6 colonies to simulate competition effects at different foraging locations. To ensure the baseline model accurately reflects expected chick survival and adult mass loss during a moderate year for each colony, the model requires calibration for each individual colony. As a result, separate models must be run for each colony, with other colonies included only to account for competition effects in the simulations.

2.2.1.3 Since SPA colonies typically consist of sub-colonies, and SeabORD version 1.3 allows a maximum of 6 colonies per run, sub-colonies were combined, and all birds were assumed to forage from the centroid of the combined colonies within the SPA. The final locations used for each colony are provided in **Table 2-1**.

2.2.1.4 The 6-colony limitation also meant that competition effects for all identified SPAs assessed for kittiwake could not be included within a single model. To address this, SPAs were ranked based on their apportioning weights (as calculated within the **Offshore RIAA Appendix D**), with those with the lowest apportioning value ranked lowest and therefore excluded from competition simulations for other colonies. The final set of colonies included for competition effects in each model is presented in **Table 2-1**.

2.2.1.5 It should be noted that East Caithness Cliffs SPA was initially the second highest ranked SPA for kittiwake. However, due to its location relative to the Offshore Project, computational run times became prohibitive. 1 scenario out of 30 took approximately 4 weeks to complete during calibration at 10 percent of the population. Extrapolating this linearly for a full run indicates it would require several months. As a result, the computational requirements were considered unfeasible, and the full SeabORD model run was not progressed. Consequently, East Caithness Cliffs SPA was removed from the assessment, and North Rona and Sula Sgeir SPA was included instead. Section 3.6 provides further detail on model runtimes and associated limitations.

2.2.1.6 SeabORD requires the number of breeding pairs at each colony as an input parameter. The most recent available population counts (up to and including data from 2024) from the Seabird Monitoring Programme (SMP) were obtained for use within modelling conducted in April 2025 (**Table 2-1**). Grey cells within **Table 2-1** indicate that the SPA was not considered within SeabORD for that species, following the colony selection process outlined above.

2.2.1.7 Correction factors were applied to the counts within the SMP which are provided as individuals (IND), to calculate the estimated number of breeding pairs for the relevant colonies for guillemot and razorbill. To correct these counts a factor of 0.67 was applied to estimate the number of breeding pairs following the approach described in Walsh *et al.* (1995). The correction factors applied to the counts obtained from the SMP to calculate the estimated number of breeding pairs for the relevant colonies are further detailed within the **Appendix 14.1, Volume 2c**. Kittiwake counts were measured in Apparently Occupied Nests (AON) for all the colonies included in the simulation, meaning that no correction factor was applied to the counts provided.

2.2.1.8 For puffin, the SeabORD modelling was not completed due to the extensive computational requirements associated with this species and the consequent runtime demands. By way of example, calibration runs for Flannan Isles SPA required in excess of 1 week to complete for a single scenario. Full model runs take substantially longer to complete than calibration runs. Given that up to 40 model runs of 10 iterations each were required for each SPA, full SeabORD model runs were projected to require at least 2 months per SPA, with more distant SPAs likely to require longer runtimes. Running additional scenarios across multiple colonies would therefore have necessitated a prolonged modelling period extending well beyond the application drafting timeframe, and was not achievable within the overall programme.

2.2.1.9 These limitations reflect not only project-specific timescales but also the practical constraints of available computational capacity, which must be balanced across multiple species, colonies, and assessment scenarios within a single application. As such, completion of puffin SeabORD models alongside the other species assessed was not achievable within the overall drafting timeframe.

2.2.1.10 Given these constraints, puffin was excluded from the SeabORD assessment. Statutory Nature Conservation Bodies (SNCBs) are aware of the substantial computational demands associated with SeabORD and of the practical challenges this presents, particularly where species-specific models require exceptionally long runtimes that cannot always be accommodated within application timelines.

Table 2-1: SPA location and total number of pairs of key species per site, taken from the Seabird Monitoring Database (SMP) (JNCC, 2024).

SPA	Longitude	Latitude	Guillemot (pairs)	Razorbill (pairs)	Kittiwake (pairs)
North Caithness Cliffs SPA	-3.3710	58.6707	N/A	N/A	5,571
Handa SPA	-5.1900	58.3806	36,625	5,499	3,749
Cape Wrath SPA	-4.9026	58.6134	25,533	2,175	3,622
Flannan Isles SPA	-7.5944	58.2880	3,773	766	825
Shiant Isles SPA	-6.3660	57.9002	N/A	5,379	1,075
North Rona and Sula Sgeir SPA	-6.0050	59.1019	5,177	265	712
Sule Skerry and Sule Stack SPA	-4.4558	59.0483	6,030	N/A	N/A
St Kilda SPA	-8.4886	57.8682	N/A	549	N/A
Canna and Sanday SPA	-6.5410	57.0566	N/A	N/A	N/A

2.2.2 MODEL REGION

2.2.2.1 The selected SPAs were used to define the model region, representing the spatial extent for running the model (i.e. the 'bounding box'). The bounding box was determined by buffering each SPA colony with the mean-maximum foraging range plus 1 standard deviation, as specified by Woodward *et al.* (2019) (**Table 2-3**). The outermost coordinates of these buffers (north, east, south, and west) were then used to establish the boundaries of the model region. This is in line with the distance-decay method within the SeabORD report (Searle *et al.* 2018).

2.2.2.2 The size of the model's bounding box can significantly affect computational run time, particularly for species with extensive foraging ranges (e.g. kittiwake) and multiple SPAs included in the analysis. In such cases, the resulting bounding boxes can be very large, leading to prolonged model simulations. To optimise computational efficiency, the bounding box for kittiwake (**Table** 2-2) was refined to exclude areas of sea to the south and north. These peripheral areas lie between the mean-maximum foraging range and the mean-maximum plus 1 standard deviation, and therefore fall within the extended foraging range likely used only by a small proportion of the population.. It is considered unlikely that the majority of individuals would regularly forage in these more distant areas. All model regions were delineated using Geographic Information System (GIS) tools.

Table 2-2: Bounding box utilised within SeabORD for each individual species.

Species	North (degrees)	East (degrees)	South (degrees)	West (degrees)
Guillemot	60.0026	-2.7541	57.3656	-9.3923
Razorbill	60.2800	-2.6860	56.7530	-9.7740
Kittiwake	61.3875	-1.6780	56.6220	-9.9300

2.2.3 PROPORTION OF POPULATION

2.2.3.1 SeabORD allows users to adjust the number of birds included in simulations by specifying a fraction of the population to be modelled. The total population size is defined in an input file, and the specified fraction is applied as a simple multiplier to these values.

2.2.3.2 While model outputs are relatively insensitive to the fraction of the total population simulated, running a smaller proportion, such as 10%, is valuable during test mode for quickly assessing initial outputs. However, for final analyses, especially in ‘multiple’ or ‘batch’ models, it is recommended to simulate as large a fraction of the population as computationally feasible. This ensures that uncertainty is accurately quantified and that results are as representative as possible.

2.2.3.3 In the final simulation runs, 25% of the population was modelled for all species: i.e. guillemot, razorbill and kittiwake. The complexity of their simulations meant that only 25% of their populations could be modelled due to long run times (see Section 3.5 for details on run time per model). This was driven by the large number of individuals being modelled for guillemot and razorbill and the extensive sea area within the foraging range for kittiwake. For these 3 species, the results are presented both for the modelled fraction and scaled up to represent the full population.

2.2.4 FORAGING AND PREY LOCATION

2.2.4.1 Using GPS data is recommended for determining the foraging ranges and prey distributions of seabird populations during SeabORD simulations (Mobbs *et al.*, 2018). GPS data enable the calculation of relative seabird densities, allowing the model to infer site-specific foraging ranges, proportions of individuals, and prey distributions. This approach provides more accurate inputs by incorporating heterogeneous prey distributions rather than relying on generalised estimates from the literature.

2.2.4.2 However, in the absence of suitable GPS data, the distance decay method was employed to estimate foraging locations, and prey distribution was assumed to be uniform. This uniform distribution does not reflect the heterogeneous nature of real-world prey availability or how central place foragers exploit resources, such as the patterns described by Ashmole's halo effect (Ashmole, 1963).

2.2.4.3 The foraging ranges for each species, applied in the distance decay method within SeabORD, are presented in **Table 2-3**. These ranges follow NatureScot guidance (2023), which recommends using the mean maximum foraging range plus 1 standard deviation (SD), as reported in Woodward *et al.* (2019).

2.2.4.4 Within the SeabORD model, the default value of 95% was used to ensure that the majority of seabird foraging trips are captured within the estimated foraging range (**Table 2-3**). This choice reflects a balance between accuracy and practicality, aiming to include nearly all foraging activities without overcomplicating the model. By using 95%, the model provides a comprehensive and realistic representation of seabird behaviour, ensuring that the majority of foraging trips are accounted for. The use of 95% is also consistent with statistical practices, where a 95% confidence interval is commonly used to indicate a high level of certainty in estimates (Bevans, 2020).

Table 2-3: Foraging ranges used within SeabORD (NatureScot, 2023; Woodward *et al.*, 2019)

Species	Foraging range (km)	Percentage in foraging range
Guillemot	95.2	95%
Razorbill	122.2	95%
Kittiwake	300.6	95%

2.2.5 DISPLACEMENT AND BARRIER EFFECT RATES

2.2.5.1 2 parameters within SeabORD define the displacement and barrier effect behaviour of simulated birds:

- **Displacement susceptibility:** This parameter specifies the percentage of the population susceptible to displacement. Displacement-susceptible birds are displaced from the Offshore Renewable Development (ORD) (ORD footprint plus user-defined border) when their chosen foraging location lies within this region and subsequently select a new foraging location within a user-defined buffer around the ORD;
- **Barrier susceptibility**: This parameter represents the percentage of displacement-susceptible birds that are also susceptible to barrier effects. Barrier-susceptible birds avoid flying through the ORD footprint and instead navigate around it to reach foraging locations beyond the displacement zone.

2.2.5.2 The current displacement rates recommended by NatureScot (2023a), as shown in **Table 2-4**, were applied within the assessment.

2.2.5.3 The displacement border zone was defined as the Offshore Project Turbine Area plus a 2 km buffer. This buffer represents the area within which displacement effects are assumed to occur. Displaced birds were then assumed to select alternative foraging locations within a 5 km buffer surrounding the displacement zone. Both buffer values are consistent with the default configuration and guidance of the SeabORD tool.

2.2.5.4 A 100% barrier rate was applied, meaning all displacement-susceptible birds were assumed to also avoid crossing the area during their flight.

2.2.5.5 The SeabORD tool provides 2 methods to simulate the movement of barrier-susceptible birds around the displacement zone:

- **Perimeter Method**: Barrier-susceptible birds travel to the edge of the displacement zone, follow its perimeter, and resume their original trajectory once they exit the displacement zone.
- **A* Pathfinding Method**: An algorithm calculates the shortest possible path around the displacement zone to the final foraging location.

2.2.5.6 For this assessment, the perimeter method was used as a precautionary approach while minimising computational intensity. Although the A* pathfinding method calculates a more efficient path, it significantly increases computational demand and relies on the assumption that birds can instantly identify and follow the optimal path, which may not reflect natural behaviour.

Table 2-4: Displacement and barrier rates utilised within SeabORD (NatureScot, 2023b).

Species	Displacement rate	Proportion of displacement-susceptible birds also susceptible to barrier effects
Guillemot	60%	100%
Razorbill	60%	100%
Kittiwake	30%	100%

2.3 MODEL CALIBRATION

2.3.1.1 Before conducting a full SeabORD analysis, calibration simulations were performed to establish the upper and lower prey level input parameters that determine prey availability for each species and SPA combination. These calibration simulations were run under baseline conditions (i.e. without the presence of additional wind farms) to ensure prey distribution accurately reflects 'moderate' conditions.

2.3.1.2 The calibration process involved conducting trial simulations with 10% of the population for each species and SPA combination. The only input parameters differing from those used in the final paired simulations were the upper and lower prey quantity values, which generated a uniform prey distribution. A range of prey values were tested, and the outputs of each calibration run were reviewed to evaluate whether the observed adult mass loss and chick survival rates fell within the recommended thresholds shown in **Table 2-5.** For guillemot, a total of 48 calibration runs were required, whilst razorbill and kittiwake each required a total of 68 calibration runs, to determine the prey boundaries across all SPAs.

2.3.1.3 Moderate prey year conditions, as derived from Mobbs *et al.* (2018), were characterised by:

- Adult body mass loss falling within specified lower and upper thresholds (**Table 2-5**);
- Chick survival rates exceeding a predefined threshold representative of a typical moderate breeding season.

2.3.1.4 Values below the moderate prey condition were classified as "poor" prey years, while values above the moderate prey conditions were classified as "good" prey years.

Table 2-5: Adult percentage body mass loss and percentage chick survival used to determine prey values used in the final paired simulations. Values taken from Mobbs *et al.* (2018)

Species	Adult Mass Loss (%)		Chick Survival (%)
	Lower boundary	Upper boundary	Lower boundary
Guillemot	3.5	10.5	49
Razorbill	3.5	10.5	50
Kittiwake	5	15	11

2.3.2 PAIRED SIMULATIONS

2.3.2.1 Once the prey quantity ranges were determined through the calibration process, they were applied to run the final simulations (**Table 2-6**). These simulations consisted of 10 paired runs, with prey quantities selected from within the calibrated ranges using random stratification. This method of prey level selection incorporates uncertainty into the model outputs by generating effects across a range of moderate prey conditions (Searle *et al.*, 2018).

2.3.2.2 For each prey quantity, the model simulated a full breeding season under baseline conditions (i.e. in the absence of the Offshore Project) and impact conditions (i.e. with the Offshore Project). This approach resulted in a total of 20 breeding season simulations, as each pair included both baseline and impact scenarios. The final outputs were calculated as the average results of the 10 simulations for each scenario.

2.3.2.3 The only parameters that differed between the calibration simulations and the final paired simulations were the prey quantity ranges and the proportions of the population included. These adjustments ensured that the final simulations accounted for realistic variability in prey availability and provided robust estimates of potential impacts.

Table 2-6: Prey quantity range used for each final paired simulation calculated within SeabORD

Species	Colony	Lower prey quantity (g per unit volume)	Upper prey quantity (g per unit volume)	Range of adult mass loss (%)	Range of chick survival (%)
Guillemot	Handa SPA	384	488	9.13 - 3.51	49.25 - 93.50
	Cape Wrath SPA	374	464	9.10 - 3.52	49.43 - 94.32
	Flannan Isles SPA	349	440	9.46 - 3.59	49.34 - 94.69
	North Rona and Sula Sgeir SPA	356	447	9.23 – 3.51	49.42 – 94.40
	Sule Skerry and Sule Stack SPA	346	431	9.33 – 3.54	50.25 – 96.02
Razorbill	Handa SPA	271	359	9.48 - 3.53	50.18 - 93.82
	Cape Wrath SPA	257	330	9.32 - 3.55	51.83 - 94.95
	Flannan Isles SPA	248	319	9.29 - 3.53	50.65 - 96.10
	Shiant Isles SPA	257	337	9.31 – 3.55	50.93 – 93.49
	North Rona and Sula Sgeir SPA	246	312	9.79 – 3.56	51.85 – 96.30
	St Kilda SPA	235	304	9.67 – 3.51	52.37 – 98.18
Kittiwake	North Caithness Cliffs SPA	215	290	10.13 - 5.18	11.49 - 84.92
	Handa SPA	252	318	9.74 – 5.06	12.27 – 91.20
	Cape Wrath SPA	238	297	10.00 – 5.05	11.88 – 92.82
	Flannan Isles SPA	215	280	9.84 – 5.01	12.89 – 80.72
	Shiant Isles SPA	243	316	10.06 – 5.07	11.11 – 78.70
	North Rona and Sula Sgeir SPA	212	282	10.49 – 5.08	12.68 – 92.96

2.3.3 BIOENERGETICS IN THE MODEL

2.3.3.1 At each timestep, adult birds were assigned a Daily Energy Expenditure (DEE). For the first timestep, the DEE was selected from a normal distribution of DEE values stored within SeabORD, and for subsequent timesteps, the DEE was set to match the energy expended by the individual in the previous timestep. The DEE of chicks was kept constant throughout the simulation.

2.3.3.2 The behavioural mode of each adult was determined by its body mass relative to its starting mass. If an adult's body mass was greater than 90% of its starting mass at the onset of chick-rearing, it would avoid leaving its chick unattended, even if it had not met its Daily Energy Requirements (DER). If the mass was between 90% and 80%, the adult would prioritise self-survival, potentially leaving the chick unattended to meet its DER. If the mass fell below 80% of its starting mass, the adult would abandon the breeding attempt, leading to chick mortality.

2.3.3.3 Should an adult’s body mass fall below 60% of its starting mass, indicating starvation, it was assumed to have died, and the partner would abandon the breeding attempt. For chicks, mortality occurred if their body mass fell below 60% of an optimally-provisioned chick. If an adult's time away from the nest exceeded a critical threshold, chick mortality due to exposure was assumed.

2.4 ANNUAL MORTALITY OUTPUT

2.4.1.1 For each species assessed, the model outputs estimates of annual adult and chick mortalities under both baseline conditions (i.e. in the absence of the Offshore Project) and impact scenarios (i.e. with the Offshore Project present). These outputs are used to quantify the potential additional mortalities associated with the presence of Offshore Project, by comparing baseline and impact values.

2.4.1.2 In addition to mortality estimates, SeabORD provides a range of other outputs reflecting behavioural and energetic responses to the Offshore Project. These include metrics such as the number of adults exposed to the Offshore Project, changes in adult body mass by the end of the breeding season, total distance travelled, and foraging trip frequency. These indicators provide insight into how displacement influences individual energy budgets and, in turn, survival and reproductive success.

2.4.1.3 Notably, it is possible for SeabORD to predict a beneficial impact of an offshore wind farm, such as improved adult mass or chick survival, relative to baseline conditions. These results can arise in cases where displacement reduces foraging distances (e.g. if birds were displaced toward more accessible or productive foraging areas). While these effects are biologically possible, a precautionary approach should be adopted if results are used within any subsequent assessments (e.g. within the **Offshore RIAA** and/or Population Viability Analysis (PVA)), with such values considered effectively zero, thereby avoiding the assumption of any positive ecological gain. For reporting purposes, beneficial results are presented as negative values.

2.4.1.4 To predict annual adult survival, SeabORD uses each individual's body mass at the end of the breeding season and applies a logistic function that relates post-breeding mass to the probability of overwinter survival (Searle *et al.*, 2018). This model requires 2 parameters: the 'baseline' survival and the slope that links changes in adult mass to survival probability. Both values are predefined within SeabORD and remain constant across simulations.

2.4.1.5 The baseline survival corresponds to the mean value derived from sites with observed data on annual adult survival, as compiled by the developers of SeabORD. The slope and shape of the logistic curve, which determines how sensitive survival probability is to body condition, are also fixed within the model. SeabORD uses these values to convert the energetic outcomes of each simulated individual into population-level mortality estimates.

2.4.1.6 For species where less than 100% of the population was simulated, the number of mortalities predicted by SeabORD was scaled up to 100% using a scaling factor of 1/proportion of the population simulated. This scaling factor assumes a linear relationship between the number of mortalities and the proportion of the population simulated.

3 RESULTS

3.1 INTRODUCTION

3.1.1.1 SeabORD results for guillemot (Section 3.2), razorbill (Section 3.3), and kittiwake (Section 3.4) are presented within the relevant species-specific subsections below. The tables within each species' section provide estimates of additional mortality across poor, moderate, and good prey year types, both with and without the Offshore Project present (**Table 3-1**, **Table 3-4** and **Table 3-7**).

3.1.1.2 In addition, mean chick mortality with and without the Offshore Project is presented for moderate prey years only for each species (**Table 3-2**, **Table 3-5** and **Table 3-8**), as this is the only prey year type for which chick mortality is output by the SeabORD model and represents the most appropriate estimate for use within the **Offshore RIAA**.

3.1.1.3 Finally, changes in trip distance, number of trips and adult body mass with and without the Offshore Project are also presented for moderate prey years only for each species (**Table 3-3**, **Table 3-6** and **Table 3-9**), reflecting the available model outputs. These outputs are used within the calculation of additional mortality and is therefore included to provide context for the mortality estimates.

3.1.1.4 The scaled mortalities presented in the tables below have been adjusted using a scaling factor of 1/0.25, reflecting that 25% of the population was modelled within the SeabORD simulations. The terms "wind farm" or "development" used within the tables refer to the Offshore Project; this terminology has been retained to ensure consistency with the SeabORD model outputs.

3.1.1.5 Section 3.5 presents the mortality rates estimated by SeabORD alongside mortality rates reported in the literature. This comparison is used to inform the discussion of SeabORD model limitations presented in Section 4. In addition, Section 3.6 provides the model runtimes for each scenario, which further supports the discussion around the computational limitations and practical constraints associated with the SeabORD modelling approach.

3.2 GUILLEMOT

Table 3-1: Modelled impacts of the Offshore Project scenario on adult guillemot during 'poor', 'moderate', and 'good' environmental conditions (prey year type).

SPA	Prey year type	Adults not surviving the year						Difference in scaled mortalities between baseline and scenario	Additional mortality (%) due to the presence of the wind farm (95% confidence intervals)
		No wind farm present			Wind farm present				
		Mean	SD	Scaled mortalities	Mean	SD	Scaled mortalities		
Handa SPA	Poor	4364.700	55.560	17458.800	4396.600	73.176	17586.400	127.600	0.174 (-1.020, 1.368)
	Moderate	2305.200	45.738	9220.800	2326.600	61.160	9306.400	85.600	0.117 (-0.660, 0.894)
	Good	1777.300	32.311	7109.200	1793.300	47.084	7173.200	64.000	0.087 (-0.559, 0.734)
Cape Wrath SPA	Poor	2748.200	46.958	10992.800	2749.000	46.068	10996.000	3.200	0.006 (-0.015, 0.027)
	Moderate	1424.900	30.293	5699.600	1425.100	30.355	5700.400	0.800	0.002 (-0.013, 0.016)
	Good	1116.800	15.150	4467.200	1116.900	15.191	4467.600	0.400	0.001 (-0.013, 0.014)
Flannan Isles SPA	Poor	351.900	11.060	1407.600	352.300	10.802	1409.200	1.600	0.021 (-0.085, 0.127)
	Moderate	183.200	9.449	732.800	183.700	9.358	734.800	2.000	0.027 (-0.040, 0.093)
	Good	126.800	6.015	507.200	127.600	6.257	510.400	3.200	0.042 (-0.100, 0.185)
North Rona and Sula Sgeir SPA	Poor	489.900	10.214	1959.600	490.400	10.617	1961.600	2.000	0.019 (-0.089, 0.127)
	Moderate	226.900	6.437	907.600	226.900	6.740	907.600	0.000	0.000 (-0.061, 0.061)
	Good	193.100	4.332	772.400	193.200	4.237	772.800	0.400	0.004 (-0.025, 0.033)
Sule Skerry and Sule Stack SPA	Poor	554.600	22.873	2218.400	554.400	22.673	2217.600	-0.800	-0.007 (-0.040, 0.027)
	Moderate	263.900	10.290	1055.600	263.900	10.290	1055.600	0.000	0.000 (0.000, 0.000)
	Good	233.500	7.028	934.000	233.500	7.028	934.000	0.000	0.000 (0.000, 0.000)

Table 3-2: Mean predicted guillemot chick mortalities (scaled to represent the whole population) and survival rates during the chick-rearing season with and without displacement and barrier effects from the Offshore Project during a moderate prey year.

SPA	Chicks not surviving season						Additional mortalities (chicks)	Additional chick mortality (%) due to the presence of the wind farm (95% confidence intervals)
	No wind farm present			Wind farm present				
	Mean	SD	Scaled mean mortalities	Mean	SD	Scaled mean mortalities		
Handa SPA	1690.900	1113.991	6763.600	1705.800	1106.832	6823.200	59.600	0.163 (-0.389, 0.715)
Cape Wrath SPA	1098.400	788.289	4393.600	1100.500	787.881	4402.000	8.400	0.033 (-0.015, 0.081)
Flannan Isles SPA	157.700	115.944	630.800	159.700	115.835	638.800	8.000	0.212 (-0.247, 0.671)
North Rona and Sula Sgeir SPA	230.700	156.178	922.800	231.300	156.030	925.200	2.400	0.046 (-0.131, 0.224)
Sule Skerry and Sule Stack SPA	243.800	176.781	975.200	243.900	176.863	975.600	0.400	0.007 (-0.043, 0.056)

Table 3-3: SeabORD outputs for guillemot at the 5 SPA colonies modelled in a moderate prey year.

SPA	Number of adult birds in simulation (individuals)	Additional distance flown when development present vs not present (km)	Difference in the total number of trips carried out with and without wind farm	Number of adults directly impacted by the development (displaced or barriered at least once)	Scenario (wind farm present/not present)	Initial adult body mass (g)	Final adult body mass (g)	Difference in body mass (g)	Reduction in body mass due to presence of wind farm (g)
Handa SPA	18,312	5.084	-0.014	977	Not present	919.935	863.839	56.096	0.618
					Present	919.935	863.220	56.715	
Cape Wrath SPA	12,766	0.017	-0.002	235	Not present	919.862	862.161	57.702	0.007
					Present	919.862	862.153	57.709	
Flannan Isles SPA	1,886	0.517	-0.012	338	Not present	919.545	862.910	56.635	0.087
					Present	919.545	862.823	56.722	
North Rona and Sula Sgeir SPA	2,588	0.062	-0.003	62	Not present	920.629	864.055	56.573	0.012
					Present	920.629	864.044	56.585	
Sule Skerry and Sule Stack SPA	3,016	-0.021	-0.001	9	Not present	920.803	864.320	56.484	-0.006
					Present	920.803	864.326	56.478	

3.3 RAZORBILL

Table 3-4: Modelled impacts of the Offshore Project scenario on adult razorbill during 'poor', 'moderate', and 'good' environmental conditions (prey year type).

SPA	Prey year type	Adults not surviving the year						Difference in scaled mortalities between baseline and scenario	Additional mortality (%) due to the presence of the wind farm (95% confidence intervals)
		No wind farm present			Wind farm present				
		Mean	SD	Scaled mortalities	Mean	SD	Scaled mortalities		
Handa SPA	Poor	756.400	7.919	3025.600	771.700	43.643	3086.800	61.200	0.556 (-3.316, 4.429)
	Moderate	419.600	12.176	1678.400	431.500	42.022	1726.000	47.600	0.433 (-2.243, 3.108)
	Good	259.200	9.864	1036.800	268.000	28.837	1072.000	35.200	0.320 (-1.932, 2.572)
Cape Wrath SPA	Poor	244.200	6.460	976.800	244.300	6.550	977.200	0.400	0.009 (-0.060, 0.078)
	Moderate	126.200	5.266	504.800	126.200	5.266	504.800	0.000	0.000 (0.000, 0.000)
	Good	71.600	2.171	286.400	71.600	2.171	286.400	0.000	0.000 (0.000, 0.000)
Flannan Isles SPA	Poor	84.200	3.048	336.800	84.000	3.232	336.000	-0.800	-0.052 (-0.443, 0.339)
	Moderate	35.200	1.814	140.800	35.200	1.814	140.800	0.000	0.000 (0.000, 0.000)
	Good	11.800	1.135	47.200	11.700	1.059	46.800	-0.400	-0.026 (-0.221, 0.169)
Shiant Isles SPA	Poor	630.600	5.168	2522.400	632.000	5.312	2528.000	5.600	0.052 (-0.081, 0.185)
	Moderate	344.500	3.779	1378.000	344.900	4.886	1379.600	1.600	0.015 (-0.111, 0.141)
	Good	183.200	8.715	732.800	183.500	7.764	734.000	1.200	0.011 (-0.145, 0.167)
North Rona and Sula Sgeir SPA	Poor	31.100	1.287	124.400	31.000	1.247	124.000	-0.400	-0.076 (-0.644, 0.493)
	Moderate	13.400	0.843	53.600	13.300	0.949	53.200	-0.400	-0.076 (-0.644, 0.493)
	Good	6.200	0.422	24.800	6.200	0.422	24.800	0.000	0.000 (0.000, 0.000)
St Kilda SPA	Poor	63.600	3.950	254.400	63.600	3.950	254.400	0.000	0.000 (0.000, 0.000)
	Moderate	26.300	1.252	105.200	26.500	1.179	106.000	0.800	0.073 (-0.292, 0.438)
	Good	17.300	0.823	69.200	17.300	0.823	69.200	0.000	0.000 (0.000, 0.000)

Table 3-5: Mean predicted razorbill chick mortalities (scaled to represent the whole population) and survival rates during the chick-rearing season with and without displacement and barrier effects from the Offshore Project

SPA	Chicks not surviving season						Additional mortalities (chicks)	Additional chick mortality (%) due to the presence of the wind farm (95% confidence intervals)
	No wind farm present			Wind farm present				
	Mean	SD	Scaled mean mortalities	Mean	SD	Scaled mean mortalities		
Handa SPA	247.800	160.597	991.200	252.000	159.776	1008.000	16.800	0.305 (-0.452, 1.063)
Cape Wrath SPA	80.700	56.087	322.800	80.700	56.087	322.800	0.000	0.000 (0.000, 0.000)
Flannan Isles SPA	28.900	20.496	115.600	29.600	20.326	118.400	2.800	0.365 (-0.469, 1.199)
Shiant Isles SPA	252.400	138.133	1009.600	256.700	137.861	1026.800	17.200	0.320 (-0.053, 0.692)
North Rona and Sula Sgeir SPA	13.200	10.229	52.800	13.200	10.229	52.800	0.000	0.000 (0.000, 0.000)
St Kilda SPA	21.100	21.850	84.400	21.200	21.801	84.800	0.400	0.073 (-0.475, 0.621)

Table 3-6: SeabORD outputs for razorbill at the 6 SPA colonies modelled, using 100% of the population in a moderate prey year.

SPA	Number of adult birds in simulation (individuals)	Additional distance flown when development present vs not present (km)	Difference in the total number of trips carried out with and without wind farm	Number of adults directly impacted by the development (displaced or barriered at least once)	Scenario (wind farm present/not present)	Initial adult body mass (g)	Final adult body mass (g)	Difference in body mass (g)	Reduction in body mass due to presence of wind farm (g)
Handa SPA	2,750	17.879	-0.035	255	Not present	583.672	548.137	35.535	0.868
					Present	583.672	547.269	36.403	
Cape Wrath SPA	1,088	0.116	0.000	23	Not present	582.923	547.426	35.497	-0.001
					Present	582.923	547.428	35.495	
Flannan Isles SPA	384	0.956	-0.016	102	Not present	583.219	546.591	36.628	0.121
					Present	583.219	546.470	36.749	
Shiant Isles SPA	2,690	-0.093	-0.017	218	Not present	582.604	546.609	35.996	0.046
					Present	582.604	546.563	36.042	
North Rona and Sula Sgeir SPA	132	0.094	-0.001	5	Not present	581.981	544.805	37.177	0.020
					Present	581.981	544.785	37.196	
St Kilda SPA	274	0.098	-0.002	10	Not present	583.712	544.337	39.375	0.022
					Present	583.712	544.315	39.397	

3.4 KITTIWAKE

Table 3-7: Modelled impacts of the Offshore Project scenario on adult kittiwake during 'poor', 'moderate', and 'good' environmental conditions (prey year type).

SPA	Prey year type	Adults not surviving the year						Difference in scaled mortalities between baseline and scenario	Additional mortality (%) due to the presence of the wind farm (95% confidence intervals)
		No wind farm present			Wind farm present				
		Mean	SD	Scaled mortalities	Mean	SD	Scaled mortalities		
North Caithness Cliffs SPA	Poor	1046.200	29.085	4184.800	1046.300	29.163	4185.200	0.400	0.004 (-0.023, 0.031)
	Moderate	698.100	21.886	2792.400	698.000	22.111	2792.000	-0.400	-0.004 (-0.052, 0.045)
	Good	406.200	19.441	1624.800	406.200	19.441	1624.800	0.000	0.000 (0.000, 0.000)
Handa SPA	Poor	802.100	16.934	3208.400	804.500	14.661	3218.000	9.600	0.128 (-0.925, 1.181)
	Moderate	541.400	13.150	2165.600	543.500	13.485	2174.000	8.400	0.112 (-0.385, 0.609)
	Good	316.900	14.625	1267.600	318.900	15.466	1275.600	8.000	0.107 (-0.607, 0.820)
Cape Wrath SPA	Poor	713.500	12.222	2854.000	713.700	12.365	2854.800	0.800	0.011 (-0.044, 0.066)
	Moderate	483.600	7.382	1934.400	483.700	7.166	1934.800	0.400	0.006 (-0.036, 0.047)
	Good	279.800	3.327	1119.200	280.100	3.281	1120.400	1.200	0.017 (-0.047, 0.080)
Flannan Isles SPA	Poor	171.100	6.244	684.400	171.200	6.143	684.800	0.400	0.024 (-0.401, 0.449)
	Moderate	115.100	4.012	460.400	115.300	4.057	461.200	0.800	0.049 (-0.194, 0.291)
	Good	58.000	2.625	232.000	58.100	2.767	232.400	0.400	0.024 (-0.401, 0.449)

SPA	Prey year type	Adults not surviving the year						Difference in scaled mortalities between baseline and scenario	Additional mortality (%) due to the presence of the wind farm (95% confidence intervals)
		No wind farm present			Wind farm present				
		Mean	SD	Scaled mortalities	Mean	SD	Scaled mortalities		
Shiant Isles SPA	Poor	237.200	3.553	948.800	237.200	3.765	948.800	0.000	0.000 (-0.208, 0.208)
	Moderate	138.000	5.395	552.000	137.800	5.574	551.200	-0.800	-0.037 (-0.223, 0.149)
	Good	93.100	5.953	372.400	93.200	5.846	372.800	0.400	0.019 (-0.121, 0.158)
North Rona and Sula Sgeir SPA	Poor	127.700	8.152	510.800	127.600	8.113	510.400	-0.400	-0.028 (-0.239, 0.183)
	Moderate	81.900	4.122	327.600	81.900	4.122	327.600	0.000	0.000 (0.000, 0.000)
	Good	49.200	4.050	196.800	49.300	4.191	197.200	0.400	0.028 (-0.183, 0.239)

Table 3-8: Mean predicted kittiwake chick mortalities (scaled to represent the whole population) and survival rates during the chick-rearing season with and without displacement and barrier effects from the Offshore Project during a moderate prey year.

SPA	Chicks not surviving season						Additional mortalities (chicks)	Additional chick mortality (%) due to the presence of the wind farm (95% confidence intervals)
	No wind farm present			Wind farm present				
	Mean	SD	Scaled mean mortalities	Mean	SD	Scaled mean mortalities		
North Caithness Cliffs SPA	644.100	322.602	2576.400	645.000	322.626	2580.000	3.600	0.065 (-0.085, 0.214)
Handa SPA	395.000	245.453	1580.000	397.600	243.856	1590.400	10.400	0.277 (-0.929, 1.484)
Cape Wrath SPA	359.000	211.855	1436.000	359.800	212.087	1439.200	3.200	0.088 (-0.182, 0.359)
Flannan Isles SPA	94.300	47.986	377.200	94.900	47.878	379.600	2.400	0.291 (-0.514, 1.097)
Shiant Isles SPA	118.000	61.998	472.000	118.500	61.787	474.000	2.000	0.186 (-0.279, 0.651)
North Rona and Sula Sgeir SPA	69.400	46.987	277.600	69.600	47.002	278.400	0.800	0.112 (-0.450, 0.674)

Table 3-9: SeabORD outputs for kittiwake at the 6 SPA colonies modelled, in a moderate prey year

SPA	Number of adult birds in simulation (individuals)	Additional distance flown when development present vs not present (km)	Difference in the total number of trips carried out with and without wind farm	Number of adults directly impacted by the development (displaced or barriered at least once)	Scenario (wind farm present/not present)	Initial adult body mass (g)	Final adult body mass (g)	Difference in body mass (g)	Reduction in body mass due to presence of wind farm (g)
North Caithness Cliffs SPA	2,786	-0.828	-0.009	74	Not present	373.605	343.922	29.683	0.005
					Present	373.605	343.917	29.688	
Handa SPA	1,874	20.985	-0.020	276	Not present	372.819	343.621	29.198	0.180
					Present	372.819	343.441	29.378	
Cape Wrath SPA	1,812	-0.333	-0.008	154	Not present	372.358	342.757	29.601	0.008
					Present	372.358	342.749	29.609	
Flannan Isles SPA	412	-2.524	-0.057	96	Not present	371.460	341.556	29.904	0.067
					Present	371.460	341.489	29.972	
Shiant Isles SPA	538	-2.646	-0.036	33	Not present	374.618	345.681	28.938	0.019
					Present	374.618	345.662	28.956	
North Rona and Sula Sgeir SPA	356	1.026	-0.005	34	Not present	374.658	344.700	29.958	0.016
					Present	374.658	344.684	29.974	

3.5 MORTALITY RATES CALCULATED USING SEABORD

3.5.1.1 **Table 3-10** summarises annual mortality rates for guillemot, razorbill, and kittiwake across multiple SPAs under different prey availability scenarios. Mortality estimates are presented as both non-scaled outputs from SeabORD simulations and scaled values, comparing conditions with and without the Offshore Project present. These results provide an overview of potential changes in mortality risk associated with the presence of the Offshore Project across the 3 species and 3 prey scenarios.

Table 3-10: Mortality rates calculated using SeabORD simulations (non-scaled) and using scaled mortality estimates with and without the wind farm present for all SPAs and species modelled.

Species	SPA	Prey Scenario	Non-scaled annual mortality (%)		Scaled annual mortality (%)	
			Wind farm not present	Wind farm present	Wind farm not present	Wind farm present
Guillemot	Handa SPA	Poor	5.96%	6.00%	23.83%	24.01%
		Moderate	3.15%	3.18%	12.59%	12.70%
		Good	2.43%	2.45%	9.71%	9.79%
	Cape Wrath SPA	Poor	5.38%	5.38%	21.53%	21.53%
		Moderate	2.79%	2.79%	11.16%	11.16%
		Good	2.19%	2.19%	8.75%	8.75%
	Flannan Isles SPA	Poor	4.66%	4.67%	18.65%	18.67%
		Moderate	2.43%	2.43%	9.71%	9.74%
		Good	1.68%	1.69%	6.72%	6.76%
	North Rona and Sula Sgeir SPA	Poor	4.73%	4.74%	18.93%	18.95%
		Moderate	2.19%	2.19%	8.77%	8.77%
		Good	1.86%	1.87%	7.46%	7.46%
	Sule Skerry and Sule Stack SPA	Poor	4.60%	4.60%	18.39%	18.39%
		Moderate	2.19%	2.19%	8.75%	8.75%
		Good	1.94%	1.94%	7.74%	7.74%
Razorbill	Handa SPA	Poor	6.88%	7.02%	27.51%	28.07%

Species	SPA	Prey Scenario	Non-scaled annual mortality (%)		Scaled annual mortality (%)	
			Wind farm not present	Wind farm present	Wind farm not present	Wind farm present
		Moderate	3.82%	3.92%	15.26%	15.69%
		Good	2.36%	2.44%	9.43%	9.75%
	Cape Wrath SPA	Poor	5.61%	5.62%	22.46%	22.46%
		Moderate	2.90%	2.90%	11.60%	11.60%
		Good	1.65%	1.65%	6.58%	6.58%
	Flannan Isles SPA	Poor	5.50%	5.48%	21.98%	21.93%
		Moderate	2.30%	2.30%	9.19%	9.19%
		Good	0.77%	0.76%	3.08%	3.05%
	Shiant Isles SPA	Poor	5.86%	5.87%	23.45%	23.50%
		Moderate	3.20%	3.21%	12.81%	12.82%
		Good	1.70%	1.71%	6.81%	6.82%
	North Rona and Sula Sgeir SPA	Poor	5.87%	5.85%	23.47%	23.40%
		Moderate	2.53%	2.51%	10.11%	10.04%
		Good	1.17%	1.17%	4.68%	4.68%
	St Kilda SPA	Poor	5.79%	5.79%	23.17%	23.17%
		Moderate	2.40%	2.41%	9.58%	9.65%
		Good	1.58%	1.58%	6.30%	6.30%
Kittiwake	North Caithness Cliffs SPA	Poor	9.39%	9.39%	37.56%	37.56%
		Moderate	6.27%	6.26%	25.06%	25.06%
		Good	3.65%	3.65%	14.58%	14.58%
	Handa SPA	Poor	10.70%	10.73%	42.79%	42.92%
		Moderate	7.22%	7.25%	28.88%	28.99%

Species	SPA	Prey Scenario	Non-scaled annual mortality (%)		Scaled annual mortality (%)	
			Wind farm not present	Wind farm present	Wind farm not present	Wind farm present
		Good	4.23%	4.25%	16.91%	17.01%
	Cape Wrath SPA	Poor	9.85%	9.85%	39.40%	39.41%
		Moderate	6.68%	6.68%	26.70%	26.71%
		Good	3.86%	3.87%	15.45%	15.47%
	Flannan Isles SPA	Poor	10.37%	10.38%	41.48%	41.50%
		Moderate	6.98%	6.99%	27.90%	27.95%
		Good	3.52%	3.52%	14.06%	14.08%
	Shiant Isles SPA	Poor	11.03%	11.03%	44.13%	44.13%
		Moderate	6.42%	6.41%	25.67%	25.64%
		Good	4.33%	4.33%	17.32%	17.34%
	North Rona and Sula Sgeir SPA	Poor	8.97%	8.96%	35.87%	35.84%
		Moderate	5.75%	5.75%	23.01%	23.01%
		Good	3.46%	3.46%	13.82%	13.85%

3.6 MODEL RUNTIME

3.6.1.1 The total runtime for each model was extensive, with guillemot, razorbill and kittiwake models taking multiple days to complete, even with only modelling 25% of the population. Run times for SeabORD modelling carried out for the Offshore Project are presented in **Table 3-11.** Total run time for the work was roughly 80 days, demonstrating the computational intensity of the process.

3.6.1.2 Given these extensive runtimes, increasing the modelled population beyond 25% for some species would be impractical. Assuming a linear relationship between the proportion of the population modelled and runtime, kittiwake models, which required over 653 hours (~27.2 days) for a full run at 25%, would take an estimated 2,612 hours (108.8 days) at 100%, which is an infeasible duration within Offshore Project constraints. Similarly, razorbill models, particularly for Handa Islands SPA, already took 187.2 hours at 25%, meaning a 100% run would require approximately 748.8 hours

(31.2 days). Even guillemot models at Handa SPA, which took 108.0 hours at 25%, would scale up to 432.0 hours (18 days) at 100%.

3.6.1.3 Additionally, the runtime disparity across species and SPAs indicates that larger-scale simulations could place an unsustainable demand on computational resources, particularly for species like kittiwakes, which require significantly longer processing times than razorbills. It is also important to note that this estimation assumes a linear increase in runtime with population size, though in practice, computational demands could scale non-linearly due to factors such as increased memory usage or model complexity. Under a linear assumption, scaling all species and SPAs to 100% would require around 319 days of processing time. These constraints highlight the necessity of balancing model resolution with practical feasibility.

Table 3-11: Total runtime for calibration and final runs per species and SPA.

Species	SPA	Run type	Number of runs	Percentage of population modelled (%)	Duration (hours)
Guillemot	Handa SPA	Calibration Run	10	10	27.0
		Full run	10	25	108.0
	Cape Wrath SPA	Calibration Run	9	10	23.6
		Full run	10	25	52.8
	Flannan Isles SPA	Calibration Run	10	10	31.2
		Full run	10	25	45.6
	North Rona and Sula Sgeir SPA	Calibration Run	8	10	19.2
		Full run	10	25	55.2
	Sule Skerry and Sule Stack SPA	Calibration Run	9	10	20.4
		Full run	10	25	58.8
Razorbill	Handa SPA	Calibration Run	13	10	16.2
		Full run	10	25	187.2
	Cape Wrath SPA	Calibration Run	7	10	11.5
		Full run	10	25	136.8
		Calibration Run	11	10	8.9

Species	SPA	Run type	Number of runs	Percentage of population modelled (%)	Duration (hours)
	Flannan Isles SPA	Full run	10	25	134.4
	Shiant Isles SPA	Calibration Run	16	10	12.6
		Full run	10	25	81.6
	North Rona and Sula Sgeir SPA	Calibration Run	11	10	8.8
		Full run	10	25	69.6
	St Kilda SPA	Calibration Run	10	10	9.7
		Full run	10	25	81.6
Kittiwake	North Caithness Cliffs SPA	Calibration Run	14	10	14.9
		Full run	10	25	168.1
	Handa SPA	Calibration Run	13	10	12.7
		Full run	10	25	206.4
	Cape Wrath SPA	Calibration Run	7	10	9.4
		Full run	10	25	98.4
	Flannan Isles SPA	Calibration Run	7	10	6.4
		Full run	10	25	69.6
	Shiant Isles SPA	Calibration Run	18	10	11.5
		Full run	10	25	62.4
	North Rona and Sula Sgeir SPA	Calibration Run	9	10	8.1
		Full run	10	25	48.2
				Total run time (hours)	1,916.8
				Total run time (days)	79.87

4 DISCUSSION OF THE SEABORD MODEL

4.1.1.1 The SeabORD model determines adult abandonment based on a percentage body mass loss during breeding and assumes that if one parent abandons a chick, the other will too, removing them from further timesteps. Additionally, the modelling process is designed to assume that individuals cannot regain mass during chick-rearing. This therefore means that mortality predictions may be affected, as in reality, abandoned individuals could regain energy and body mass, potentially altering annual mortality estimates based on body condition.

4.1.1.2 In addition, the baseline mortality rates incorporated into SeabORD (**Table 3-10**) have been found to be higher than those reported within literature (i.e. Horswill and Robinson (2015)), likely due to the slope parameter in the mass-survival relationship being steeper than suggested by more geographically relevant data (Vallejo *et al.*, 2022). For example, when considering additional mortalities in the absence of a wind farm, the scaled mortalities for Handa SPA range from 7,109.2 to 17,458.8 within a population of 36,625 pairs (73,250 individuals), resulting in an adult baseline mortality rate of 9.70 to 23.8% (**Table 3-10**). However, published estimates indicate a lower adult baseline mortality rate for guillemot at 6.1% (Horswill and Robinson, 2015). A similar trend is observed for razorbill, where SeabORD predicts a higher baseline mortality (**Table 3-10**) compared to the published estimates of 10.5%, and for kittiwake, where published rates are around 10.6% (Horswill and Robinson, 2015).

4.1.1.3 The model assumes a uniform prey distribution due to the lack of GPS data for all SPA colonies, despite real-world variability. Ashmole's Halo theory suggests prey depletion near colonies due to predation, which SeabORD does not account for unless colony-specific GPS data is available. Sensitivity analysis by Natural Power indicated that assuming uniform prey distribution increased adult kittiwake mortality in 'good' years and chick mortality during chick-rearing (Vallejo et al., 2022). SeabORD calibration uses “moderate” prey availability as a baseline to represent average conditions, but this does not capture the extremes of good or poor prey years, which can significantly influence energetic stress and displacement effects. As a result, uniform prey assumptions may lead to over- or under-estimation of mortality risk depending on actual prey conditions (Searle *et al.,* 2022).

4.1.1.4 In some cases, SeabORD predicts higher additional seabird mortalities in ‘good’ prey years compared to ‘moderate’ years, or ‘moderate’ years compared to ‘poor’ years, which can seem counterintuitive. This is because the metric considered here is additional mortalities as a result of the ORD, not overall mortality. Therefore, it is possible that in a ‘good’ year, the ORD has a bigger impact because most birds would otherwise survive, so displacement or increased competition leads to more additional deaths. By contrast, in poor prey years, baseline mortality is already high, and only birds foraging closest to the colony survive, leaving little or no additional mortality attributable to the ORD. An extreme example would be in a year with no prey, all birds die regardless of the presence of an ORD (i.e. zero additional mortality), whereas in a good year, the presence of the ORD causes some birds to die due to longer foraging trips or heightened competition. It is important to note that these are biologically plausible explanations for the patterns observed, rather than definitive drivers of the specific model outputs, which cannot be determined from the available data. Furthermore, greater prey availability can heighten competition among conspecifics near colonies, potentially offsetting the benefits of a good prey year and leading to increased mortality (Vallejo *et al.*, 2022).

4.1.1.5 In some instances, the SeabORD results indicate a negative value for additional mortality in the presence of the ORD, i.e. increased survival in the presence of the ORD. Such outcomes are biologically possible, as discussed in Section 2.4 and Searle *et al*. (2022), as displacement could lead birds to forage in areas that are overall more favourable, rather than foraging patterns following an ideal free distribution, which assumes perfect knowledge of prey and competitor distribution and is therefore unlikely in reality. However, negative values in the SeabORD results can also arise due to stochasticity in the model, when the true effect size is small. Therefore, for assessment and reporting purposes, negative values are treated as zero additional mortalities to ensure consistency with other offshore wind farm assessments (e.g., Vallejo et al., 2022) and to avoid misrepresenting population-level impacts.

4.1.1.6 SeabORD scales mortality estimates using a factor of 1/proportion of the population simulated, but it is unclear whether mortality scales linearly with population size, adding uncertainty to the final estimates. Additionally, many parameters rely on expert judgment rather than empirical data, leading to potential oversimplifications and unquantified uncertainty (Vallejo *et al.*, 2022; Searle *et al.*, 2022). When parameters are based on empirical data, it is unclear how applicable those data are (e.g. applying values from the Forth & Tay region to elsewhere).

4.1.1.7 Future updates to SeabORD are expected to address some of these issues, particularly in handling uncertainty (Searle *et al.*, 2022). The assessment presented here was undertaken using the most up-to-date version of SeabORD available at the time of analysis. As noted in paragraph 1.1.1.2, the SeabORD outputs are provided for contextual information only and are used to support the wider assessment rather than as standalone determinative results.

5 GLOSSARY OF TERMS AND ABBREVIATIONS

5.1.1.1 A list of key terms and acronyms used in this annex are provided in **Table 5-1** and **Table 5-2.**

Table 5-1 Acronyms and Abbreviations

Term	Definition
AON	Apparently Occupied Nests
DEE	Daily Energy Expenditure
DER	Daily Energy Requirements
EIA	Environmental Impact Assessment
EIAR	Environmental Impact Assessment Report
GIS	Geographic Information System
GPS	Global Positioning System
HRA	Habitats Regulations Assessment
IBM	Individual-Based Model
IND	Individuals counted at the colony
O&M	Operation and Maintenance
ORD	Offshore Renewable Development
PVA	Population Viability Analysis
RIAA	Report to Inform Appropriate Assessment
SD	Standard Deviation
SMP	Seabird Monitoring Programme
SNCBs	Statutory Nature Conservation Bodies
SPA	Special Protection Area

Table 5-2 Glossary

Term	Meaning
Additional mortality (%)	Additional mortality (%) is the increase in the mortality rates caused by the presence of the wind farm.
Buffer	The buffer area surrounds the Turbine Area individuals that are susceptible to displacement will be assigned a foraging location in the buffer if their original randomly selected foraging location is within the wind farm footprint.
Displacement	Individuals that are not able to forage within the wind farm footprint and must find a new foraging location are classified as suffering from displacement.
Model region	Extent of the region of interest used in a SeabORD run.

Term	Meaning
Offshore Project	Components of the Project seaward of Mean High Water Springs (MHWS). Includes Array Area plus Offshore Cable Area of Search.
Paired simulation	Paired simulations simulate two breeding seasons. The only difference between the two simulations is the presence of the wind farm. If multiple pairs are run in the same simulation, each pair of simulations will have a unique prey quantity value selected.
Prey year type	The year type can be 'poor', 'moderate' or 'good', these classifications represent the environmental conditions during the year, and classifications use values expected during moderate environmental conditions. 'Poor' classifications occur when the % body mass loss of adults is higher and % chick survival is lower than those observed during a typical year. 'Good' years are only classified based on the % body mass loss of adults
SeabORD	Modelling tool developed by Searle *et al.* (2018) to assess displacement and barrier effects of ORD.
Turbine Area	A reduced area within the Array Area where above water surface infrastructure would be located i.e. wind turbines or Offshore Substation Platform. Developed and refined through environmental assessment.

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