Morven South Offshore Wind Array Project

Environmental Impact Assessment Report

Volume 3, Annex 11.5: Offshore Ornithology Displacement Modelling Report (SeabORD)

MVCNS-J1201-RPS-10121
May 2026

B01

Document status

Version	Purpose of document	Authored by	Checker	Approved by	Date
FINAL	Application	TTRPSEL	TTRPSEL	MvOWL	May 2026

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1 Introduction

1.1 Context

1.1.1.1 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, operations and maintenance 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).

1.1.1.2 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 operation and maintenance (O&M) phase, the presence of operational wind turbines has the potential to directly disturb seabirds leading to displacement from the Morven South Offshore Wind Array Project (hereafter 'Morven South') Boundary, including a buffer around it.

1.1.1.3 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).

1.1.1.4 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.

1.1.1.5 NatureScot have 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.

1.1.1.6 SeabORD is an individual-based model developed by the Centre for Ecology and Hydrology (CEH) to evaluate the bio-energetic costs of seabird responses to distributional changes, both at the individual and population levels, expressed as estimated mortalities. The model simulates the flight paths of individual birds from Special Protection Area (SPA) sub-sites (or 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.

1.2 Purpose and scope

1.2.1.1 This technical appendix outlines the methodology and findings used to assess displacement and barrier effects on seabirds during the operation of Morven South. To evaluate these effects, the SeabORD tool (Searle *et al*., 2018) was used alongside a matrix-based approach, following guidance from NatureScot (NatureScot, 2023), with use of SeabORD requested within the Morven Option Lease Agreement Site Scoping Opinion (hereafter ‚Morven Site Scoping Opinion') (Scottish Government, 2023), with its application further confirmed in writing from NatureScot (NatureScot, 2025). This report focusses exclusively on the use of the SeabORD tool to model displacement and barrier effects on:

- Kittiwake (*Rissa tridactyla*);
- Common guillemot (*Uria aalge*);
- Razorbill (*Alca torda); and*
- Puffin (*Fratercula arctica*).

1.2.1.2 Currently, the four species outlined above are the only species for which SeabORD can predict the impact of distributional responses during the chick-rearing/breeding season. Details of the matrix-based approach, which provides assessments for the breeding season, migratory period, and non-breeding season for these species, are presented in Volume 3, Annex 11.4: Offshore Ornithology Displacement Modelling Report (Matrix Approach) to enable comparisons with the SeabORD results during the chick-rearing/breeding season.

2 Methods

2.1.1.1 SeabORD modelling was conducted for the proposed project in line with NatureScot guidance (NatureScot, 2023a). Models were run using SeabORD version 1.3 (SeabORD, 2024). This is currently the most up-to-date publicly available version of the model at the time of writing.

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

1. 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);
2. 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) is parameterised specifically for the Forth and Tay region. This version relies on older data from 2010 to 2014 and requires a manual calibration step to ensure prey levels accurately reflect poor, moderate, and good conditions for seabirds. Calibration is necessary whenever new bird distribution maps or colonies are introduced. The SeabORD tool does not incorporate more recent tracking data current.

2.1.1.4 The model was run on a 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 five offshore wind farms, with the number of built and proposed offshore wind farms in the Forth and Tay region far exceeding this.

2.2 Input parameters

2.2.1 Special Protected Areas and species information

2.2.1.1 The choice of SPAs to model in SeabORD (version 1.3) was based on the outputs of the Habitats Regulation Assessment (HRA) screening report (Volume 1, Chapter 1: Morven Option Lease Agreement Site: HRA Stage 1 Screening Report of the HRA).

2.2.1.2 The SeabORD tool can incorporate data from up to six 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 six 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 six-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 in Volume 2, Annex 3.1: Report to Inform Appropriate Assessment: Apportioning of the HRA), 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 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 January 2025 (Table 2.1). Grey cells within Table 2.1 indicate that the species was not considered within SeabORD for that SPA either due to the SPA being outside the species foraging range or because the species is not a qualifying feature of that site.

2.2.1.6 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) and applied to population data for these two species in the two most recent national seabird censuses (Mitchell *et al*., 2004 and Burnell *et al*., 2023). 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 Volume 3, Annex 11.1: Offshore Ornithology Baseline Characterisation Report. For puffin, counts are provided as Apparently Occupied Burrows (AOB) (a measure of breeding pairs) and so no correction factor was applied. 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.

Table 2.1: Special Protected Area locations and total number of pairs of key species per site, taken from the Seabird Monitoring Database Sectoral Marine Plan (JNCC, 2024)

SPA	Longitude	Latitude	Guillemot (pairs)	Razorbill (pairs)	Kittiwake (pairs)	Puffin (pairs)
Buchan Ness to Collieston Coast SPA	-1.7819	57.4667			13,547	
Fowlsheugh SPA	-2.1970	56.9094	54,306	8,885	15,483	
Forth Islands SPA	-2.5499	56.1830		4,188	7,090	46,076
St Abb's Head to Fast Castle SPA	-2.1331	55.9018		2,065	5,602	
East Caithness Cliffs SPA	-3.3630	58.2640			18,411	
Farne Islands SPA	-1.62369	55.6337				50,103
Coquet Islands SPA	-1.54077	55.3347				17,541
Troup, Pennan and Lion's Head SPA	-2.3109	57.6928			9,853	

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 one 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 bounding box can influence the computable runtime of the model, with species that have large foraging ranges (e.g. kittiwake) and multiple SPAs included within the model resulting in large bounding boxes and therefore extensive modelling time. To reduce the amount of computable runtime required for the simulations, the bounding box (Table 2.2) for all species only encompassed sea area on the east coast (i.e. the North Sea), with the part of the bounding box that extended across land to the west and included the Atlantic Ocean and Irish Sea excluded (Figure 2.1). This was calculated within Geographic Information System (GIS).

Table 2.2: Bounding box utilised within SeabORD for each individual species

Species	North (Degrees)	East (Degrees)	South (Degrees)	West (Degrees)
Guillemot	58.321464	-0.195621	56.054442	-2.322318
Razorbill	58.006663	-0.179331	54.804162	-2.667541
Kittiwake	60.961282	3.230376	52.635746	-3.362232
Puffin	58.566122	2.643702	52.950317	-2.728105

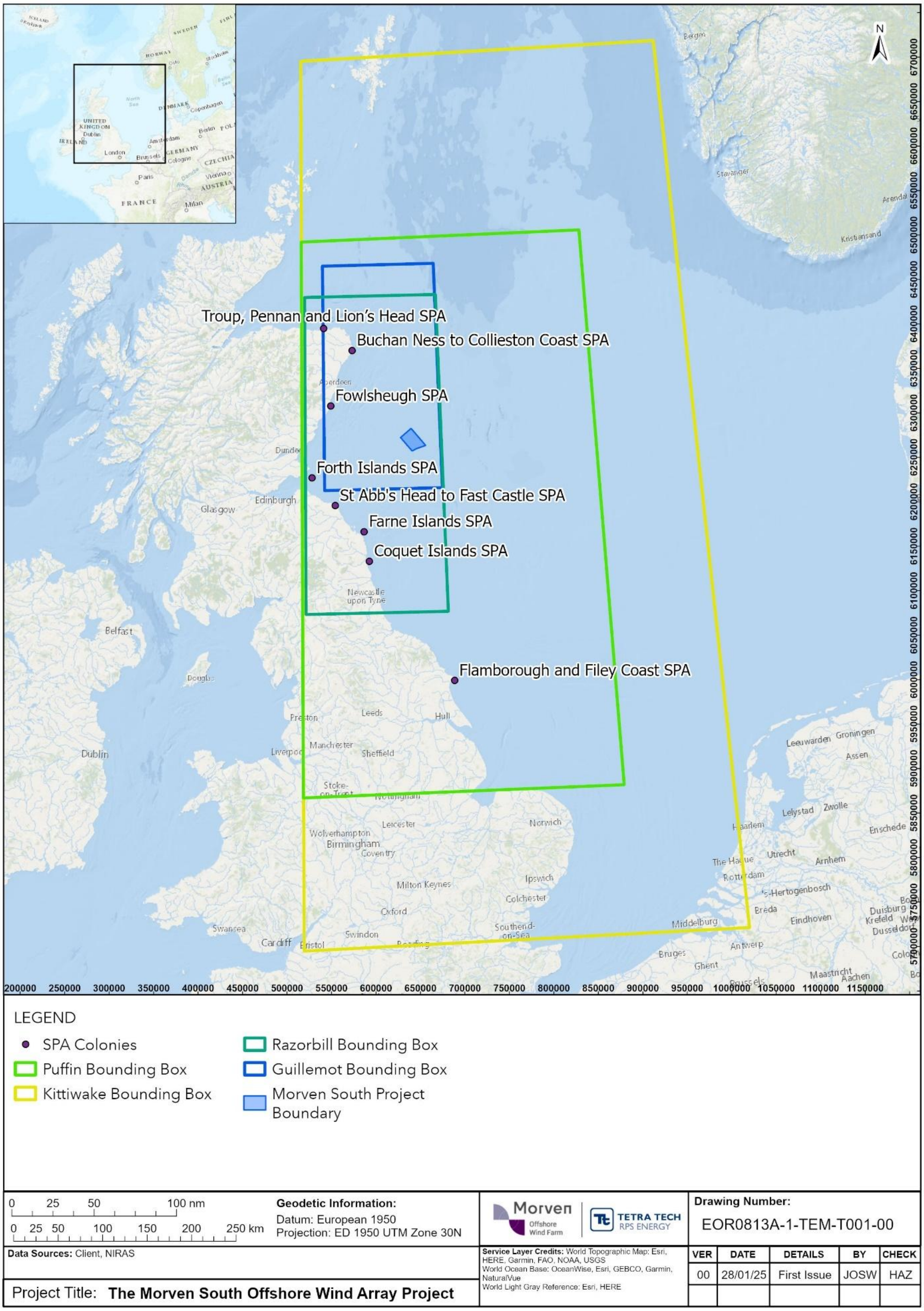

Figure 2.1: SeabORD bounding box for guillemot, razorbill, kittiwake, and puffin

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, 100% of the razorbill population was successfully modelled. For puffin, guillemot, and kittiwake, the complexity of their simulations meant that only 20% 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 the extensive sea area within the foraging range for kittiwake and puffin. For these three 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). Despite this, the uniform prey distribution assumption did not affect the simulation results, as all foraging locations were assigned the same prey availability.

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 one 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%

Species	Foraging Range (km)	Percentage in Foraging Range
Kittiwake	300.6	95%
Puffin	265.4	95%

2.2.5 Displacement and mortality rates

2.2.5.1 The current displacement and barrier effect rates recommended by NatureScot (2023b), as shown in Table 2.4, were applied within the assessment. Two parameters within SeabORD define the displacement and barrier behaviour of simulated birds:

1. Displacement susceptibility: This parameter specifies the percentage of the population susceptible to displacement. Displacement-susceptible birds are those whose preferred foraging locations fall within the Offshore Renewable Development (ORD) footprint and are subsequently displaced to alternative foraging locations (Searle *et al*., 2018).
2. 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 (Searle *et al*., 2018).

2.2.5.2 The displacement zone was defined as Morven South plus a 2km buffer, while displaced birds were assumed to select alternative foraging locations within a 5km buffer surrounding the displacement zone.

2.2.5.3 A 100% barrier rate was applied, meaning all birds displaced from foraging within the Morven South Boundary were assumed to also avoid crossing the area during their flight. This precautionary assumption ensures that any potential effects of barrier behaviour on bird energetics and flight paths are fully captured in the model.

2.2.5.4 The SeabORD tool provides two 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 foraging location.

2.2.5.5 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 Displaced also Barriered
Guillemot	60%	100%
Razorbill	60%	100%
Kittiwake	30%	100%
Puffin	60%	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 18 calibration runs were required, while razorbill needed 23, kittiwake 61 and puffin 27, to determine the prey boundaries across all SPAs.

2.3.1.3 Moderate 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.

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
Puffin	3.5	10.5	50

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 ten 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 selected prey quantity, the model simulated a full breeding season under baseline conditions (without Morven South) and impact conditions (with Morven South). 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 ten 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	Range of adult mass loss (%)	Range of chick survival (%)	Lower prey quantity (g per unit volume)	Upper prey quantity (g per unit volume)
Guillemot	Fowlsheugh SPA	9.14 – 3.53	49.55 – 94.07	364	455
Razorbill	Fowlsheugh SPA	9.43 – 3.59	50.17 – 94.60	260	336
	Forth Islands SPA	9.48 - 3.55	49.88 - 95.23	216	286
	St Abb's Head to Fast Castle SPA	9.54 – 3.57	52.17 – 95.17	236	302
Kittiwake	Buchan Ness to Collieston Coast SPA	9.58 - 5.00	11.51 – 75.13	231	298
	Fowlsheugh SPA	9.28 - 5.07	11.63 - 68.41	243	304
	Forth Islands SPA	9.30 – 5.04	11.28 - 59.94	222	292
	St Abb's Head to Fast Castle SPA	9.52 – 5.03	11.43 – 65.00	224	295
	East Caithness Cliffs SPA	9.47 – 5.05	11.93 – 67.90	233	318
	Troup, Pennan and Lion's Head SPA	9.56 -5.06	11.27 -73.30	236	303
Puffin	Forth Islands SPA	10.47 – 3.56	74.76 – 94.36	277	345
	Farne Islands SPA	10.44 – 3.52	77.72 – 94.69	274	335
	Coquet Islands SPA	10.34 – 3.52	78.56 – 95.15	286	348

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.

2.3.3.4 For burrow-nesting puffins, once a chick's mass deficit reached 80% of the DER, it ventured to the burrow entrance, increasing its risk of predation. If the mass deficit was below 60%, the chick was assumed to have died. These thresholds were based on expert judgment and empirical data, following similar logic to Langton *et al*. (2014) (Searle *et al*., 2018).

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 Morven South) and impact scenarios (i.e. with Morven South present). These outputs are used to quantify the potential additional mortalities associated with the presence of Morven South, 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 Morven South. These include metrics such as the number of adults exposed to Morven South, 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, in some simulations, SeabORD predicted modest benefits under impact scenarios, such as improved adult mass or chick survival, relative to baseline conditions. These results typically arose in cases where displacement reduced foraging distance (e.g. if birds were displaced toward more accessible or productive foraging areas). While these effects are biologically plausible, a precautionary approach should be adopted if results are used within any subsequent assessments (e.g. Population Viability Analysis (PVA)), with such values set to zero, thereby avoiding the assumption of any positive ecological gain.

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 two 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 Guillemot

Table 3.1: Modelled impacts of Morven South scenario on adult guillemot during 'poor', 'moderate', and 'good' environmental conditions (prey year type). Scaled mortalities were calculated using a scaling factor of 1/0.2

SPA	Prey year type	Adults not surviving the year						Difference in scaled mortalities between baseline and scenario	Additional mortality (%) due to the presence of Morven South
		Morven South not present			Morven South not present				
		Mean	SD	Scaled mortalities	Mean	SD	Scaled mortalities		
Fowlsheugh SPA	Poor	4803.200	17.662	24016.000	4803.500	19.643	24017.500	1.500	0.001 (-0.030, 0.032)
	Moderate	2426.300	18.980	12131.500	2428.100	21.533	12140.500	9.000	0.008 (-0.032, 0.049)
	Good	1878.700	13.124	9393.500	1878.800	13.223	9394.000	0.500	0.000 (-0.012, 0.012)

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 Morven South During a moderate prey year

SPA	Mean number of chicks not surviving season (no Morven South)	chicks not surviving season stdev (no Morven South)	scaled mean number of chicks not surviving season (no Morven South)	mean number of chicks not surviving season (Morven South present)	chicks not surviving season stdev (Morven South present)	scaled mean number of chicks not surviving season (Morven South present)	Additional mortalities (chicks)	Additional chick mortality (%) due to the presence of Morven South
Fowlsheugh SPA	1927.500	1371.876	9637.500	1929.900	1372.972	9649.500	12.000	0.022 (-0.026, 0.071)

Table 3.3: SeabORD outputs for guillemot at the two Special Protected Area colonies modelled in a moderate prey year

SPA	Number of adult birds in simulation (individuals)	Additional distance flown when Morven South is present vs not present (km)	Difference in the total number of trips carried out with and without Morven South	Number of adults directly impacted by Morven South (displaced or barriered at least once)	Scenario (Morven South 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 Morven South (g)
Fowlsheugh SPA	21722.000	0.422	-0.001	1303.000	Not present	919.574	861.799	57.775	0.040
					Present	919.574	861.759	57.815	

3.2 Razorbill

Table 3.4: Modelled impacts of Morven South scenario on adult razorbill during 'poor', 'moderate', and 'good' environmental conditions (prey year type). scaled mortalities are equal to estimated mortalities due to the use of the full population

SPA	Prey year type	Adults not surviving the year				Difference in mortalities between baseline and scenario	Additional mortality (%) due to the presence of Morven South
		Morven South not present		Morven Suth present			
		Mean	SD	Mean	SD		
Fowlsheugh SPA	Poor	4655.400	107.999	4661.100	108.806	5.700	0.018 (-0.011, 0.075)
	Moderate	2686.800	58.195	2691.000	57.386	4.200	0.024 (-0.018, 0.065)
	Good	1485.300	28.304	1486.900	27.926	1.600	0.009 (-0.028, 0.046)
Forth Islands SPA	Poor	1661.500	69.650	1662.100	68.981	0.600	0.007 (-0.033, 0.048)
	Moderate	915.400	43.421	917.200	43.435	1.800	0.021 (-0.013, 0.056)
	Good	494.600	21.864	495.300	21.823	0.700	0.008 (-0.015, 0.032)
St Abb's Head to Fast Castle SPA	Poor	906.700	16.872	908.800	16.137	2.100	0.051 (-0.028, 0.130)
	Moderate	501.900	12.965	503.500	12.510	1.600	0.039 (-0.029, 0.106)
	Good	296.300	4.762	296.500	4.428	0.200	0.005 (-0.054, 0.064)

Table 3.5: Mean predicted razorbill chick mortalities and survival rates during the chick-rearing season with and without displacement and barrier effects from Morven South, using 100% of the population

SPA	Mean number of chicks not surviving season (no Morven South)	Chicks not surviving season stdev (no Morven South)	Mean number of chicks not surviving season (Morven South present)	Chicks not surviving season stdev (Morven South present)	Additional mortalities (chicks)	Additional chick mortality (%) due to the presence of Morven South
Fowlsheugh SPA	834.100	515.113	836.900	515.888	2.800	0.032 (-0.024, 0.088)
Forth Islands SPA	675.700	459.261	677.300	460.577	1.600	0.038 (-0.098, 0.175)
St Abb's Head to Fast Castle SPA	278.700	187.104	279.600	187.359	0.900	0.044 (-0.022, 0.109)

Table 3.6: SeabORD Outputs for razorbill at the three Special Protected Area 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)
Fowlsheugh SPA	17770.000	0.876	0.001	1817.000	Not present	582.720	554.225	28.496	0.038
					present	582.720	554.187	28.533	
Forth Islands SPA	8376.000	1.034	0.002	430.000	Not present	582.644	546.451	36.193	0.035
					present	582.644	582.644	36.229	
St Abb's Head to Fast Castle SPA	4130.000	1.504	0.004	299.000	Not present	582.467	547.797	34.671	0.066
					present	582.467	547.731	34.737	

3.3 Kittiwake

Table 3.7: Modelled impacts of Morven South scenario on adult kittiwake during 'poor', 'moderate', and 'good' environmental conditions (prey year type). Scaled mortalities were calculated using a scaling factor of 1/0.2

SPA	Prey year type	Adults not surviving the year						Difference in scaled mortalities between baseline and scenario	Additional mortality (%) due to the presence of Morven South
		Morven South not present			Morven South present				
		Mean	SD	Scaled mortalities	Mean	SD	Scaled mortalities		
Buchan Ness to Collieston Coast SPA	Poor	2049.000	26.390	10245.000	2049.200	26.246	10246.000	1.000	0.004 (-0.073, 0.080)
	Moderate	1385.800	15.576	6929.000	1387.100	15.466	6935.500	6.500	0.024 (-0.018, 0.066)
	Good	831.700	15.875	4158.500	832.200	15.208	4161.000	2.500	0.009 (-0.054, 0.072)
Fowlsheugh SPA	Poor	2455.700	12.311	12278.500	2456.600	11.587	12283.000	4.500	0.015 (-0.063, 0.092)
	Moderate	1693.300	15.692	8466.500	1693.900	14.579	8469.500	3.000	0.010 (-0.069, 0.089)
	Good	1025.600	13.285	5128.000	1026.300	14.135	5131.500	3.500	0.011 (-0.046, 0.069)
Forth Islands SPA	Poor	1023.000	7.860	5115.000	1024.200	8.244	5121.000	6.000	0.042 (-0.081, 0.166)
	Moderate	728.600	7.662	3643.000	730.300	6.977	3651.500	8.500	0.060 (-0.052, 0.172)
	Good	410.100	6.027	2050.500	410.600	6.114	2053.000	2.500	0.018 (-0.081, 0.166)
St Abb's Head to Fast Castle SPA	Poor	843.700	4.398	4218.500	844.600	4.006	4223.000	4.500	0.040 (-0.087, 0.167)
	Moderate	563.400	4.551	2817.000	564.200	4.492	2821.000	4.000	0.036 (-0.150, 0.221)
	Good	317.600	3.950	1588.000	317.500	3.749	1587.500	-0.500	-0.004 (-0.083, 0.074)
	Poor	2968.900	56.394	14844.500	2969.100	56.446	14845.500	1.000	0.003 (-0.023, 0.028)
	Moderate	2048.600	69.712	10243.000	2049.300	69.717	10246.500	3.500	0.010 (-0.025, 0.044)

SPA	Prey year type	Adults not surviving the year						Difference in scaled mortalities between baseline and scenario	Additional mortality (%) due to the presence of Morven South
		Morven South not present			Morven South present				
		Mean	SD	Scaled mortalities	Mean	SD	Scaled mortalities		
East Caithness Cliffs SPA	Good	1273.000	59.974	6365.000	1273.700	59.803	6368.500	3.500	0.010 (-0.031, 0.050)
Troup, Pennan and Lion’s Head SPA	Poor	1485.300	18.062	7426.500	1485.600	16.946	7428.000	1.500	0.008 (-0.106, 0.121)
	Moderate	1001.700	7.602	5008.500	1001.700	7.528	5008.500	0.000	0.000 (-0.028, 0.028)
	Good	629.700	4.138	3148.500	630.000	3.887	3150.000	1.500	0.008 (-0.049, 0.065)

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 Morven South during a moderate prey year

SPA	Mean number of chicks not surviving season (no Morven South)	Chicks not surviving season stdev (no Morven South)	Scaled mean number of chicks not surviving season (no Morven South)	Mean number of chicks not surviving season (Morven South present)	Chicks not surviving season stdev (Morven South present)	Scaled mean number of chicks not surviving season (Morven South present)	Additional mortalities (chicks)	Additional chick mortality (%) due to the presence of Morven South
Buchan Ness to Collieston Coast SPA	1433.100	567.541	7165.500	1433.800	567.906	7169.000	3.500	0.026 (-0.117, 0.169)
Fowlsheugh SPA	1830.200	552.738	9151.000	1833.100	552.204	9165.500	14.500	0.094 (-0.057, 0.244)
Forth Islands SPA	872.000	246.245	4360.000	873.300	245.623	4366.500	6.500	0.092 (-0.046, 0.229)
St Abb's Head to Fast Castle SPA	654.700	197.238	3273.500	657.000	196.309	3285.000	11.500	0.205 (-0.242, 0.653)
East Caithness Cliffs SPA	2345.700	769.697	11728.500	2346.100	770.000	11730.500	2.000	0.011 (-0.043, 0.065)
Troup, Pennan and Lion's Head SPA	1032.800	391.539	5164.000	1034.100	391.064	5170.500	6.500	0.006 (-0.033, 0.165)

Table 3.9: SeabORD outputs for kittiwake at the six Special Protected Area colonies modelled in a moderate prey year

SPA	Number of adult birds in simulation (individuals)	Additional distance flown when Morven South present vs not present (km)	Difference in the total number of trips carried out with and without Morven South	Number of adults directly impacted by Morven South (displaced or barriered at least once)	Scenario (Morven South 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 Morven South (g)
Buchan Ness to Collieston Coast SPA	5418.000	0.692	-0.008	565.000	Not present	373.053	344.419	28.634	0.017
					present	373.053	344.403	28.650	
Fowlsheugh SPA	6194.000	0.901	-0.015	1104.000	Not present	372.392	344.285	28.106	0.026
					present	372.392	344.259	28.132	
Forth Islands SPA	2836.000	1.469	-0.025	425.000	Not present	371.752	343.630	28.122	0.017
					present	371.752	343.613	28.138	
St Abb's Head to Fast Castle SPA	2240.000	2.596	-0.020	391.000	Not present	372.433	344.807	27.626	0.027
					present	372.433	344.780	27.654	
East Caithness Cliffs SPA	7364.000	0.420	0.000	190.000	Not present	371.990	343.654	28.336	0.005
					present	371.990	343.649	28.341	
Troup, Pennan and Lion's Head SPA	3942.000	0.507	-0.005	289.000	Not present	373.002	344.664	28.338	0.001
					present	373.002	344.665	28.337	

3.4 Puffin

Table 3.10: Modelled impacts of Morven South scenario on adult puffin during 'poor', 'moderate', and 'good' environmental conditions (prey year type). Scaled mortalities were calculated using a scaling factor of 1/0.2

SPA	Prey year type	Adults not surviving the year						Difference in scaled mortalities between baseline and scenario	Additional mortality (%) due to the presence of Morven South
		Morven South not present			Morven South present				
		Mean	SD	Scaled mortalities	Mean	SD	Scaled mortalities		
Forth Islands SPA	Poor	3689.700	31.577	18448.500	3702.900	27.735	18514.500	66.000	0.072 (0.010, 0.133)
	Moderate	2625.700	15.938	13128.500	2641.200	16.219	13206.000	77.500	0.084 (0.020, 0.148)
	Good	1470.000	5.477	7350.000	1477.100	6.806	7385.500	35.500	0.039 (-0.005, 0.082)
Farne Islands SPA	Poor	3524.400	24.075	17622.000	3537.400	18.674	17687.000	65.000	0.065 (-0.024, 0.153)
	Moderate	2540.300	29.436	12701.500	2553.200	25.918	12766.000	64.500	0.064 (0.002, 0.127)
	Good	1345.300	7.364	6726.500	1351.000	7.180	6755.000	28.500	0.028 (-0.002, 0.059)
Coquet Islands SPA	Poor	1457.000	51.504	7285.000	1463.900	53.382	7319.500	34.500	0.098 (0.026, 0.170)
	Moderate	2550.300	30.159	5254.500	2562.200	27.414	5279.000	24.500	0.044 (-0.035, 0.175)
	Good	612.800	26.935	3064.000	615.100	29.730	3075.500	11.500	0.033 (-0.081, 0.147)

Table 3.11: Mean predicted puffin chick mortalities (scaled to represent the whole population) and survival rates during the chick-rearing season with and without displacement and barrier effects from Morven South during a moderate prey year

SPA	Mean number of chicks not surviving season (no Morven South)	Chicks not surviving season stdev (no Morven South)	Scaled mean number of chicks not surviving season (no Morven South)	Mean number of chicks not surviving season (Morven South present)	Chicks not surviving season stdev (Morven South present)	Scaled mean number of chicks not surviving season (Morven South present)	Additional mortalities (chicks)	Additional Chick mortality (%) due to the presence of Morven South
Forth Islands SPA	889.200	447.454	4446.000	898.100	454.165	4490.500	44.500	0.097 (-0.089, 0.282)
Farne Islands SPA	865.000	399.182	4325.000	874.900	407.822	4374.500	49.500	0.099 (-0.114, 0.312)
Coquet Islands SPA	294.100	152.634	1470.500	296.400	155.286	1482.000	11.500	0.066 (-0.131, 0.262)

Table 3.12: SeabORD outputs for puffin at the three Special Protected Area colonies modelled in a moderate prey year

SPA	Number of adult birds in simulation (individuals)	Additional distance flown when Morven South present vs not present (km)	Difference in the total number of trips carried out with and without Morven South	Number of adults directly impacted by Morven South (displaced or barriered at least once)	Scenario (Morven South 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 Morven South (g)
Forth Islands SPA	18430.000	8.659	0.008	6263.000	Not present	392.643	369.054	23.589	0.164
					present	392.643	392.643	23.753	
Farne Islands SPA	20042.000	8.607	0.003	7004.000	Not present	392.699	371.027	21.673	0.156
					present	392.699	370.871	21.828	
Coquet Islands SPA	7016.000	5.114	0.003	1963.000	Not present	392.933	368.989	23.944	0.115
					present	392.933	368.874	24.059	

3.5 Model runtime

3.5.1.1 As stated, the total runtime for each model was extensive, with guillemot, kittiwake and puffin models taking multiple days to complete, even with only modelling 20% of the population. Run times for SeabORD modelling carried out for Morven South are presented in Table 3.13. Total run time for the work was roughly 62 days, demonstrating the computational intensity of the process.

3.5.1.2 Given these extensive runtimes, increasing the modelled population beyond 20% for some species would be impractical. Assuming a linear relationship between the proportion of the population modelled and runtime, kittiwake models for Buchan Ness to Collieston Coast SPA, which required over 420 hours (~17.5 days) for a full run at 20%, would take an estimated 2,102 hours (~87.6 days) at 100%, making it infeasible within project constraints. Similarly, puffin models, particularly for Coquet Islands SPA, already took 134.4 hours at 20%, meaning a 100% run would require approximately 672 hours (~28 days). Other species and SPAs show a similar pattern: kittiwake models at Fowlsheugh SPA would require 516 hours (~21.5 days), while guillemot models at Fowlsheugh SPA, which took 28.8 hours at 20%, would scale up to 144 hours (~6 days) at 100%. The trend remains consistent across species, with significantly higher estimated runtimes for full population modelling, further reinforcing the impracticality of scaling up beyond 20%.

3.5.1.3 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 or puffins. 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. Even under a linear assumption, scaling all species and SPAs to 100% would require approximately 311 days of processing time. These constraints highlight the necessity of balancing model resolution with practical feasibility.

Table 3.13 Total runtime for calibration and final runs per species and Special Protected Area

Species	SPA	Run type	Number of runs	Percentage of population modelled (%)	Duration (average hours)
Guillemot	Fowlsheugh SPA	Calibration Run	8	10	14.4
		Full run	10	20	28.8
Razorbill	Fowlsheugh SPA	Calibration Run	6	10	2.3
		Full run	10	100	81.6
	Forth Islands SPA	Calibration Run	7	10	6.1
		Full run	10	100	158.4
	St Abb's Head to Fast Castle SPA	Calibration Run	10	10	22.0
		Full run	10	100	69.6
Kittiwake		Calibration Run	10	10	19.1

Species	SPA	Run type	Number of runs	Percentage of population modelled (%)	Duration (average hours)
	Buchan Ness to Collieston Coast SPA	Full run	10	20	420.4
	Fowlsheugh SPA	Calibration Run	8	10	50.4
		Full run	10	20	103.2
	Forth Islands SPA	Calibration Run	7	10	4.7
		Full run	10	20	67.2
	St Abb's Head to Fast Castle SPA	Calibration Run	14	10	7.6
		Full run	10	20	36
	East Caithness Cliff's SPA	Calibration Run	12	10	5.0
		Full run	10	20	25.1
	Troup, Pennan and Lion's Head SPA	Calibration Run	10	10	3.3
		Full run	10	20	28.8
Puffin	Forth Islands SPA	Calibration Run	7	10	16.6
		Full run	10	20	40.8
	Farne Islands SPA	Calibration Run	8	10	24.1
		Full run	10	20	79.2
	Coquet Islands SPA	Calibration Run	12	10	43.2
		Full run	10	20	134.4
				Total run time (hours)	1,492.3
				Total run time (days)	62.17

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 have been found to be higher than those reported in PVA (Table 4.1), 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 adult guillemot in the absence of a wind farm, the scaled mortalities for Buchan Ness to Collieston Coast SPA range from 3,532 to 8,957 within a population of 20,381 pairs (40,762 individuals), resulting in an adult baseline mortality rate of 0.088 to 0.219 (Table 4.1). However, published estimates indicate a lower adult baseline mortality rate for guillemot at 0.061 (Horswill and Robinson, 2015). This trend of higher baseline mortality rates in SeabORD compared to baseline mortality estimates used in PVA is also observed for razorbill, kittiwake and puffin.

4.1.1.3 The model also 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).

4.1.1.4 Further, the model's distance decay function does not align with the example GPS-based distribution maps in the Forth and Tay area that were originally used for the development of the SeabORD model (Searle *et al*., 2018), likely due to the model's inability to capture patchy prey distributions affecting real foraging patterns. SeabORD also selects foraging sites randomly, whereas in reality, birds consider competition and prey abundance.

4.1.1.5 In some cases, SeabORD predicts higher seabird mortality in 'good' prey years compared to 'moderate' years, which can seem counterintuitive. This is the result of complex interactions: while prey availability may be higher, displacement from wind farms can force birds to forage further or in less productive areas, increasing energetic costs. Furthermore, greater prey availability can heighten competition among conspecifics and other predators, potentially offsetting the benefits of a good prey year and leading to increased mortality (Vallejo *et al*., 2022). A similar dynamic can apply to poor prey years, where predicted mortality may sometimes be lower than in moderate years. In these cases, birds may adjust their foraging strategies in response to prey scarcity, potentially reducing exposure to wind farm impacts or competition pressures. This can result in lower mortality estimates in some poor prey scenarios, despite the overall lower availability of food.

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).

4.1.1.7 Future updates to SeabORD are expected to address some of these issues, particularly in handling uncertainty (Searle *et al*., 2022).

Table 4.1: Mortality rates calculated using SeabORD simulations (non-scaled) and using scaled mortality estimates with and without Morven Sout present for all Special Protected Areas and species modelled. greyed-out cells marked as n/a for razorbill indicate that scaling was not required, as the modelling was conducted using 100% of the population

Species	SPA	Prey scenario	Non-scaled annual mortality (%)		Scaled annual mortality (%)	
			Morven South not present	Morven South present	Morven South not present	Morven South present
Guillemot	Fowlsheugh SPA	Poor	22.11%	22.11%	22.11%	22.11%
		Moderate	11.17%	11.18%	11.17%	11.18%
		Good	8.65%	8.65%	8.65%	8.65%
Razorbill	Fowlsheugh SPA	Poor	N/A	N/A	26.20%	26.23%
		Moderate	N/A	N/A	15.12%	15.14%
		Good	N/A	N/A	8.36%	8.37%
	Forth Islands SPA	Poor	N/A	N/A	19.84%	19.84%
		Moderate	N/A	N/A	10.93%	10.95%
		Good	N/A	N/A	5.90%	5.91%
	St Abb's Head to Fast Castle SPA	Poor	N/A	N/A	21.95%	22.00%
		Moderate	N/A	N/A	12.15%	12.19%
		Good	N/A	N/A	7.17%	7.18%
Kittiwake	Buchan Ness to Collieston Coast SPA	Poor	37.82%	37.82%	37.81%	37.82%
		Moderate	25.58%	25.60%	25.57%	25.60%
		Good	15.35%	15.36%	15.35%	15.36%
	Fowlsheugh SPA	Poor	39.65%	39.66%	39.65%	39.67%
		Moderate	27.34%	27.35%	27.34%	27.35%
		Good	16.56%	16.57%	16.56%	16.57%
	Forth Islands SPA	Poor	36.07%	36.11%	36.07%	36.11%
		Moderate	25.69%	25.75%	25.69%	25.75%
		Good	14.46%	14.48%	14.46%	14.48%
	St Abb's Head to Fast Castle SPA	Poor	37.67%	37.71%	37.65%	37.69%
		Moderate	25.15%	25.19%	25.14%	25.18%
		Good	14.18%	14.17%	14.17%	14.17%
	East Caithness Cliff's SPA	Poor	40.32%	40.32%	40.31%	40.32%
		Moderate	27.82%	27.83%	27.82%	27.83%
		Good	17.29%	17.30%	17.29%	17.30%
	Troup, Pennan and Lion's Head SPA	Poor	37.68%	37.69%	37.69%	37.69%
		Moderate	25.41%	25.41%	25.42%	25.42%
		Good	15.97%	15.98%	15.98%	15.98%

Species	SPA	Prey scenario	Non-scaled annual mortality (%)		Scaled annual mortality (%)	
			Morven South not present	Morven South present	Morven South not present	Morven South present
Puffin	Forth Islands SPA	Poor	20.02%	20.09%	20.02%	20.09%
		Moderate	14.25%	14.33%	14.25%	14.33%
		Good	7.98%	8.01%	7.98%	8.01%
	Farne Islands SPA	Poor	17.59%	17.65%	17.59%	17.65%
		Moderate	12.67%	12.74%	12.68%	12.74%
		Good	6.71%	6.74%	6.71%	6.74%
	Coquet Islands SPA	Poor	20.77%	20.87%	20.77%	20.86%
		Moderate	36.35%	36.52%	14.98%	15.05%
		Good	8.73%	8.77%	8.73%	8.77%

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