Morven North Offshore Wind Array Project

Environmental Impact Assessment Report

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

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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 North Offshore Wind Array Project (hereafter 'Morven North') 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 North. 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*);
- 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:

- **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) 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.

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 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 Appraisal (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	20,381		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

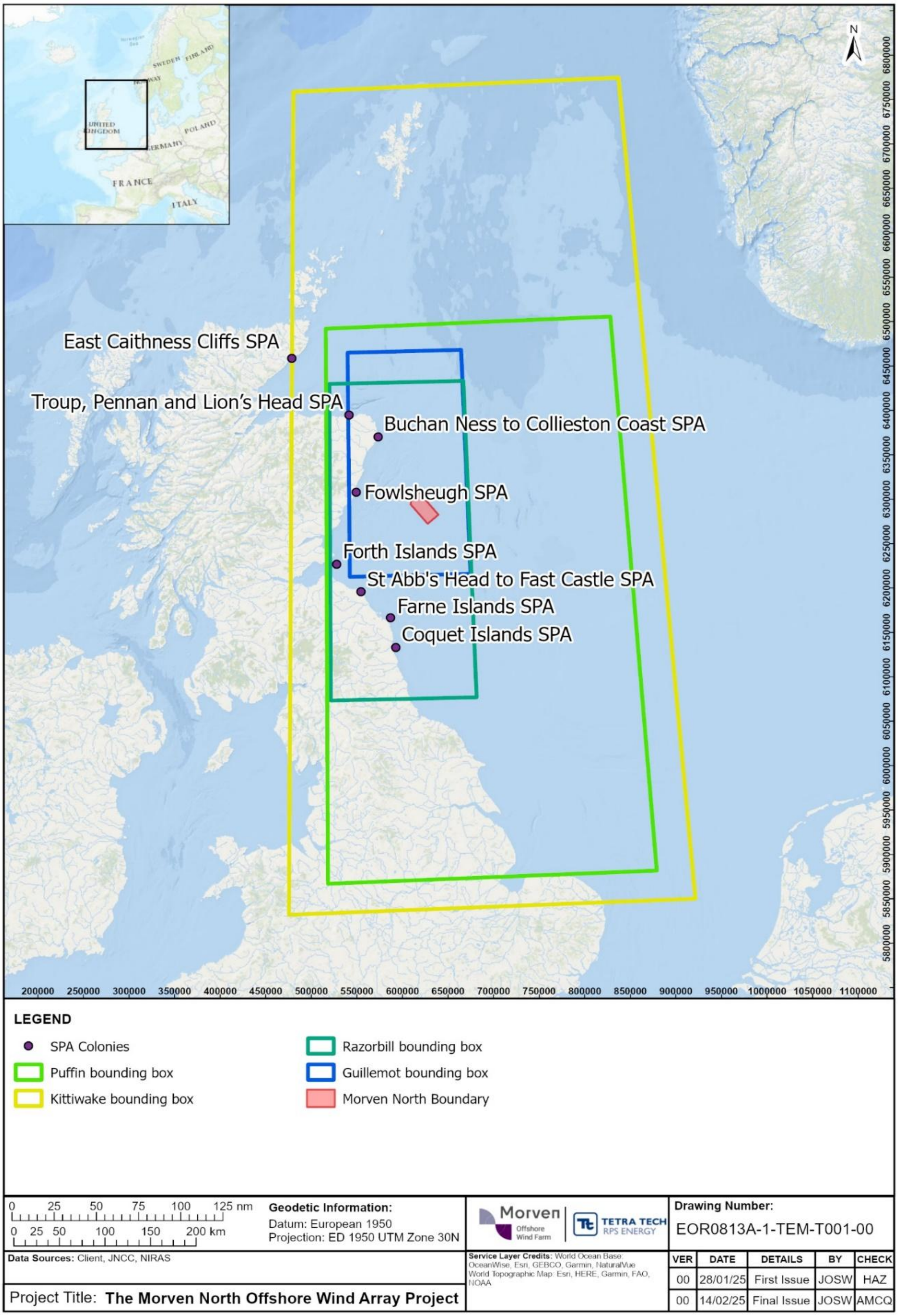

Figure 2.1: SeabORD bounding box for guillemot, razorbill, kittiwake and puffin

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 population

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%
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:

- **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);
- **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 Offshore Renewable Development (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 North 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 North 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); and
- 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	
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 North) and impact conditions (with Morven North). 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	Buchan Ness to Collieston Coast SPA	9.23 – 3.56	49.75 – 94.85	362	451
	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 judgement 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 North) and impact scenarios (i.e. with Morven North present). These outputs are used to quantify the potential additional mortalities associated with the presence of Morven North, 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 North. These include metrics such as the number of adults exposed to Morven North, 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 North 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 North
		Morven North not present			Morven North present				
		Mean	SD	Scaled mortalities	Mean	SD	Scaled Mortalities		
Buchan Ness to Collieston Coast SPA	Poor	1791.400	19.311	8957.000	1795.900	17.362	8979.500	22.500	0.055 (-0.307, 0.417)
	Moderate	927.800	6.828	4639.000	930.000	7.513	4650.000	11.000	0.027 (-0.111, 0.165)
	Good	706.400	3.718	3532.000	709.400	7.183	3547.000	15.000	0.037 (-0.075, 0.148)
Fowlsheugh SPA	Poor	4753.800	18.165	23769.000	4759.600	20.609	23798.000	29.000	0.027 (-0.053, 0.107)
	Moderate	2475.100	20.262	12375.500	2479.200	20.406	12396.000	20.500	0.019 (-0.024, 0.062)
	Good	1903.000	20.342	9515.000	1905.500	18.063	9527.500	12.500	0.012 (-0.031, 0.054)

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 North during a moderate prey year

SPA	Mean number of chicks not surviving season (no Morven North)	chicks not surviving season stdev (no Morven North)	Scaled mean number of chicks not surviving season (no Morven North)	Mean number of chicks not surviving season (Morven North present)	Chicks not surviving season stdev (Morven North present)	Scaled mean number of chicks not surviving season (Morven North present)	Additional mortalities (chicks)	Additional chick mortality (%) due to the presence of Morven North
Buchan Ness to Collieston Coast SPA	688.400	525.455	3442.000	689.000	525.762	3445.000	3.000	0.015 (10.112, 0.141)
Fowlsheugh SPA	1891.800	1342.910	9459.000	1897.800	1340.662	9489.000	30.000	0.055 (-0.087, 0.198)

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 North is present vs not present (km)	Difference in the total number of trips carried out with and without Morven North	number of adults directly impacted by Morven North (displaced or barriered at least once)	Scenario (Morven North 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 North (g)
Buchan Ness to Collieston Coast SPA	8,152	2.351	0.002	592	Not present	920.648	862.577	58.071	0.200
					present	920.648	862.377	58.271	
Fowlsheugh SPA	21,722	0.698	-0.010	4,576	Not present	919.451	862.103	57.348	0.086
					present	919.451	862.017	57.434	

3.2 Razorbill

Table 3.4: Modelled impacts of Morven North 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 North
		No Morven North present		Wind farm present			
		Mean	SD	Mean	SD		
Fowlsheugh SPA	Poor	4652.200	114.139	4679.000	89.275	26.800	0.151 (-0.417, 0.718)
	Moderate	2685.500	60.451	2711.300	49.076	25.800	0.145 (-0.260, 0.551)
	Good	1484.900	29.233	1499.200	21.280	14.300	0.080 (-0.123, 0.284)
Forth Islands SPA	Poor	1661.500	69.650	1665.500	68.533	4.000	0.078 (-0.029, 0.185)
	Moderate	2680.600	85.811	2683.900	86.937	3.300	0.019 (0.042, 0.117)
	Good	1463.700	38.947	1468.100	37.681	4.400	0.025 (-0.029, 0.079)
St Abb's Head to Fast Castle SPA	Poor	906.700	16.872	909.100	15.716	2.400	0.058 (-0.093, 0.209)
	Moderate	501.900	12.965	504.400	10.814	2.500	0.061 (-0.100, 0.221)
	Good	296.300	4.762	297.100	4.748	0.800	0.019 (-0.081, 0.210)

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 North, using 100% of the population

SPA	mean number of chicks not surviving season (no Morven North)	Chicks not surviving season stdev (no Morven North)	mean number of chicks not surviving season (Morven North present)	chicks not surviving season stdev (Morven North present)	additional mortalities (chicks)	Additional chick mortality (%) due to the presence of Morven North
Fowlsheugh SPA	834.000	515.195	841.600	518.768	7.600	0.086 (-0.069, 0.240)
Forth Islands SPA	675.700	459.261	682.900	460.824	7.200	0.172 (-0.031, 0.375)
St Abb's Head to Fast Castle SPA	278.700	187.104	281.300	187.809	2.600	0.126 (-0.085, 0.337)

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 Morven North is present vs not present (km)	Difference in the total number of trips carried out with and without Morven North	Number of adults directly impacted by Morven North (displaced or barriered at least once)	Scenario (Morven North 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 North (g)
Fowlsheugh SPA	17,770	4.543	-0.020	4,977	Not present	582.720	554.331	28.389	0.283
					present	582.720	554.048	28.672	
Forth Islands SPA	8,376	2.815	0.004	758	Not present	582.644	546.451	36.193	0.094
					present	582.644	546.357	36.287	
St Abb's Head to Fast Castle SPA	4,130	2.684	0.005	337	Not present	582.467	547.797	34.670	0.099
					present	582.467	547.698	34.769	

3.3 Kittiwake

Table 3.7: Modelled impacts of Morven North 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 North
		Morven North not present			Morven North present				
		Mean	SD	Scaled mortalities	Mean	SD	Scaled mortalities		
Buchan Ness to Collieston Coast SPA	Poor	2049.000	26.390	10245.000	2049.900	25.710	10249.500	4.500	0.017 (-0.040, 0.073)
	Moderate	1385.800	15.576	6929.000	1386.800	15.332	6934.000	5.000	0.018 (-0.043, 0.080)
	Good	831.700	15.875	4158.500	833.000	15.811	4165.000	6.500	0.024 (-0.051, 0.099)
Fowlsheugh SPA	Poor	2455.700	12.311	12278.500	2458.600	11.626	12293.000	14.500	0.047 (-0.114, 0.208)
	Moderate	1693.300	15.692	8466.500	1695.600	16.794	8478.000	11.500	0.037 (-0.112, 0.186)
	Good	1025.600	13.285	5128.000	1028.100	15.110	5140.500	12.500	0.040 (-0.88, 0.168)
Forth Islands SPA	Poor	1023.000	7.860	5115.000	1023.300	7.747	5116.500	1.500	0.011 (-0.213, 0.234)
	Moderate	728.600	7.662	3643.000	729.300	6.075	3646.500	3.500	0.025 (-0.202, 0.251)
	Good	410.100	6.027	2050.500	411.500	4.994	2057.500	7.000	0.049 (-0.114, 0.213)
St Abb's Head to Fast Castle SPA	Poor	843.700	4.398	4218.500	844.700	3.268	4223.500	5.000	0.045 (-0.167, 0.256)
	Moderate	563.400	4.551	2817.000	563.500	4.625	2817.500	0.500	0.004 (-0.157, 0.166)
	Good	317.600	3.950	1588.000	318.400	3.273	1592.000	4.000	0.036 (-0.143, 0.214)
	Poor	2968.900	56.394	14844.500	2968.700	56.502	14843.500	-1.000	-0.003 (-0.042, 0.037)

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 North
		Morven North not present			Morven North present				
		Mean	SD	Scaled mortalities	Mean	SD	Scaled mortalities		
East Caithness Cliffs SPA	Moderate	2048.600	69.712	10243.000	2048.400	69.433	10242.000	-1.000	-0.003 (-0.048, 0.042)
	Good	1273.000	59.974	6365.000	1274.000	60.063	6370.000	5.000	0.014 (-0.020, 0.048)
Troup, Pennan and Lion's Head SPA	Poor	1485.300	18.062	7426.500	1485.700	16.680	7428.500	2.000	0.010 (-0.129, 0.150)
	Moderate	1001.700	7.602	5008.500	1001.700	7.379	5008.500	0.000	0.000 (0.012, 0.028)
	Good	629.700	4.138	3148.500	630.000	3.944	3150.000	1.500	0.008 (-0.062, 0.077)

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 North during a moderate prey year

SPA	mean number of chicks not surviving season (no Morven North)	Chicks not surviving season stdev (no Morven North)	Scaled mean number of chicks not surviving season (no Morven North)	mean number of chicks not surviving season (Morven North present)	Chicks not surviving season stdev (Morven North present)	Scaled mean number of chicks not surviving season (Morven North Present)	Additional mortalities (chicks)	Additional chick mortality (%) due to the presence of Morven North
Buchan Ness to Collieston Coast SPA	1433.100	567.541	7165.500	1435.000	568.385	7175.000	9.500	0.070 (-0.158, 0.298)
Fowlsheugh SPA	1830.200	552.738	9151.000	1838.900	553.984	9194.500	43.500	0.281 (0.052, 0.509)
Forth Islands SPA	872.000	246.245	4360.000	877.700	246.231	4388.500	28.500	0.402 (0.165, 0.639)
St Abb's Head to Fast Castle SPA	654.700	197.238	3273.500	660.800	196.433	3304.000	30.500	0.545 (0.127, 0.962)
East Caithness Cliffs SPA	2345.700	769.697	11728.500	2346.200	769.565	11731.000	2.500	0.014 (-0.020, 0.048)
Troup, Pennan and Lion's Head SPA	1032.800	391.539	5164.000	1034.400	391.508	5172.000	8.000	0.081 (-0.071, 0.233)

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 North is present vs not present (km)	Difference in the total number of trips carried out with and without Morven North	number of adults directly impacted by Morven North (displaced or barriered at least once)	Scenario (Morven North 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 North (g)
Buchan Ness to Collieston Coast SPA	5418.000	0.417	-0.012	924.000	Not present	373.053	344.419	28.634	0.022
					present	373.053	344.397	28.656	
Fowlsheugh SPA	6194.000	6.185	-0.042	1658.000	Not present	372.392	344.285	28.106	0.081
					present	372.392	344.204	28.188	
Forth Islands SPA	2836.000	0.658	-0.103	612.000	Not present	371.752	343.630	31.612	0.051
					present	371.752	343.638	31.662	
St Abb's Head to Fast Castle SPA	2240.000	0.415	-0.090	437.000	Not present	372.433	344.807	27.626	0.010
					present	372.433	344.797	27.636	
East Caithness Cliffs SPA	7364.000	0.416	-0.001	335.000	Not present	371.990	343.654	28.336	0.006
					present	371.990	343.648	28.342	
Troup, Pennan and Lion’s Head SPA	3942.000	1.105	-0.005	433.000	Not present	373.002	344.664	28.338	0.001

3.4 Puffin

Table 3.10: Modelled impacts of Morven North 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 North
		Morven North not present			Morven North present				
		Mean	SD	Scaled mortalities	Mean	SD	Scaled mortalities		
Forth Islands SPA	Poor	3689.700	31.577	18448.500	3736.000	26.499	18680.000	231.500	0.251 (0.150, 0.353)
	Moderate	2625.700	15.938	13128.500	2659.000	10.509	13295.000	166.500	0.181 (0.057, 0.305)
	Good	1470.000	5.477	7350.000	1490.000	5.888	7450.000	100.000	0.109 (0.058, 0.159)
Farne Islands SPA	Poor	3518.400	20.018	17592.000	3534.300	21.166	17671.500	79.500	0.079 (0.038, 0.121)
	Moderate	2533.000	20.849	12665.000	2547.300	21.965	12736.500	71.500	0.071 (0.006, 0.136)
	Good	1344.600	4.248	6723.000	1354.900	4.909	6774.500	51.500	0.051 (0.011, 0.092)
Coquet Islands SPA	Poor	1457.000	51.504	7285.000	1460.300	52.959	7301.500	16.500	0.047 (-0.036, 0.130)
	Moderate	1050.900	37.054	5254.500	1056.100	36.574	5280.500	26.000	0.074 (0.017, 0.131)
	Good	612.800	26.935	3064.000	615.000	28.566	3075.000	11.000	0.031 (-0.041, 0.104)

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 North during a moderate prey year

SPA	Mean number of chicks not surviving season (no Morven North)	Chicks not surviving season stdev (no Morven North)	Scaled mean number of chicks not surviving season (no Morven North)	Mean number of chicks not surviving season (Morven North present)	Chicks not surviving season stdev (Morven North present)	Scaled mean number of chicks not surviving season (Morven North present)	Additional mortalities (chicks)	Additional chick mortality (%) due to the presence of Morven North
Forth Islands SPA	889.200	447.454	4446.000	910.700	470.456	4553.500	107.500	0.233 (-0.366, 0.833)
Farne Islands SPA	919.700	414.203	4598.500	932.400	424.380	4662.000	63.500	0.127 (-0.141, 0.395)
Coquet Islands SPA	294.100	152.634	1470.500	297.100	154.838	1485.500	15.000	0.086 (-0.119, 0.290)

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 North present vs not present (km)	Difference in the total number of trips carried out with and without Morven North	Number of adults directly impacted by Morven North (displaced or barriered at least once)	Scenario (Morven North 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 North (g)
Forth Islands SPA	18430.000	22.356	0.021	8346.000	Not present	392.832	368.428	24.404	0.890
					present	392.832	367.538	25.294	
Farne Islands SPA	20042.000	7.801	-0.002	6375.000	Not present	392.699	369.408	23.291	0.178
					present	392.699	369.230	23.469	
Coquet Islands SPA	7016.000	3.631	-0.003	1669.000	Not present	392.933	368.989	23.944	0.100
					present	392.933	368.889	24.044	

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 North are presented in Table 3.13. Total run time for the work was roughly 67 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, 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%, which is an infeasible duration 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). Even guillemot models at Buchan Ness to Collieston Coast SPA, which took 76.8 hours at 20%, would scale up to 384 hours (16 days) at 100%.

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 over 300 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	Buchan Ness to Collieston Coast SPA	Calibration Run	10	10	36.0
		Full run	10	20	76.8
	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)	1,605.1
				Total run time (days)	66.8

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 Morven North, 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 ecological interactions: while prey availability may be higher, displacement from Morven North has the potential to 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 Morven North 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 judgement 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 North present for all Special Protected areas As 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 North not present	Morven North present	Morven North not present	Morven North present
Guillemot	Buchan Ness to Collieston Coast SPA	Poor	21.97%	22.03%	21.97%	22.03%
		Moderate	11.38%	11.41%	11.38%	11.41%
		Good	8.67%	8.70%	8.66%	8.70%
	Fowlsheugh SPA	Poor	21.88%	21.91%	21.88%	21.91%
		Moderate	11.39%	11.41%	11.39%	11.41%
		Good	8.76%	8.77%	8.76%	8.77%
Razorbill	Fowlsheugh SPA	Poor	N/A	N/A	26.18%	26.33%
		Moderate	N/A	N/A	15.11%	15.26%
		Good	N/A	N/A	8.36%	8.44%
	Forth Islands SPA	Poor	N/A	N/A	19.84%	19.88%
		Moderate	N/A	N/A	32.00%	32.04%
		Good	N/A	N/A	17.47%	17.53%
	St Abb's Head to Fast Castle SPA	Poor	N/A	N/A	21.95%	22.01%
		Moderate	N/A	N/A	12.15%	12.21%
		Good	N/A	N/A	7.17%	7.19%
Kittiwake	Buchan Ness to Collieston Coast SPA	Poor	37.82%	37.83%	37.81%	37.83%
		Moderate	25.58%	25.60%	25.57%	25.59%
		Good	15.35%	15.37%	15.35%	15.37%
	Fowlsheugh SPA	Poor	39.65%	39.69%	39.65%	39.70%
		Moderate	27.34%	27.37%	27.34%	27.38%
		Good	16.56%	16.60%	16.56%	16.60%
	Forth Islands SPA	Poor	36.07%	36.08%	36.07%	36.08%
		Moderate	25.69%	25.72%	25.69%	25.72%
		Good	14.46%	14.51%	14.46%	14.51%
	St Abb's Head to Fast Castle SPA	Poor	37.67%	37.71%	37.65%	37.70%
		Moderate	25.15%	25.16%	25.14%	25.15%
		Good	14.18%	14.21%	14.17%	14.21%
	East Caithness Cliff's SPA	Poor	40.32%	40.31%	40.31%	40.31%
		Moderate	27.82%	27.82%	27.82%	27.81%
		Good	17.29%	17.30%	17.29%	17.30%

Species	SPA	Prey Scenario	Non-Scaled Annual Mortality (%)		Scaled Annual Mortality (%)	
			Morven North not present	Morven North present	Morven North not present	Morven North present
	Troup, Pennan and Lion's Head SPA	Poor	37.68%	37.69%	37.69%	37.70%
		Moderate	25.41%	25.41%	25.42%	25.42%
		Good	15.97%	15.98%	15.98%	15.98%
Puffin	Forth Islands SPA	Poor	20.02%	20.27%	20.02%	20.27%
		Moderate	14.25%	14.43%	14.25%	14.43%
		Good	7.98%	8.08%	7.98%	8.08%
	Farne Islands SPA	Poor	17.56%	17.63%	17.56%	17.64%
		Moderate	12.64%	12.71%	12.64%	12.71%
		Good	6.71%	6.76%	6.71%	6.76%
	Coquet Islands SPA	Poor	20.77%	20.81%	20.77%	20.81%
		Moderate	14.98%	15.05%	14.98%	15.05%
		Good	8.73%	8.77%	8.73%	8.77%

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