BEATRICE OFFSHORE WIND FARM TECHNICAL REPORT IN SUPPORT OF ORNITHOLOGICAL HABITAT REGULATIONS ASSESSMENT

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1   INTRODUCTION

1. This technical report provides supporting information for the Habitat Regulations Assessment of the Beatrice Offshore Wind Farm. It comprises a description of the methodology used to apportion the peak breeding season abundance of each key species found in the Wind Farm Site to all Special Protection Area (SPA) populations within foraging range, based on size of colony, distance from Wind Farm Site and proportion of sea available within foraging range. Results are then used to determine which species should be considered at risk of Likely Significant Effects (LSEs). The second section details a stochastic population model developed for the East Caithness Cliffs SPA population of greater black-backed gulls. This model is used to explore the potential effects of additional mortality, caused by collisions with turbines, on this population.

2 SPA SEABIRDS AT RISK OF LIKELY SIGNIFICANT EFFECT

2.1 Method for apportioning seabird populations amongst candidate SPAs within foraging range

2. For several of the seabird species observed on the Beatrice Wind Farm Site there is more than one SPA from which they could originate. It is considered highly unlikely that all possible SPAs will have contributed equally to the wind farm birds, with factors such as distance and SPA population size having an important role in determining the proportion originating from each SPA. Thus a method has been developed to apportion birds from each species amongst SPAs. In addition, seabirds seen on the Wind Farm Sites may originate from seabird colonies which lie outside SPA boundaries. Therefore this assessment represents a worst case scenario where all seabirds seen on the Wind Farm Site are linked to SPAs.

3. This approach provides a means of estimating the extent of connectivity between the Wind Farm Site and the SPAs within the region. Outside the breeding season such connectivity is extremely difficult to establish, since most seabird species disperse widely from their breeding colonies. During the breeding season, the demands of reproduction are such that, as centrally-placed foragers, breeding adults are much more constrained in the areas over which they forage. Estimates of foraging range are available for many species and these permit initial selection of SPAs on the basis of distance to the Wind Farm Site. However, for species which are qualifying features of more than one SPA within foraging range of the Wind Farm Site this could potentially create a situation whereby SPAs located at different distances from the Wind Farm Site being assessed as equivalent in terms of risk of Likely Significant Effects (LSEs). A method was developed to estimate the relative contribution of candidate SPAs to the Wind Farm Site population which uses three variables; distance, SPA population size and the proportion of the total area within the species' foraging range which is sea. Using these metrics, the Wind Farm Site peak breeding season abundance can be apportioned amongst the candidate SPAs.

4. For each SPA a weight is calculated as follows:

SPA weight = SPA population / (distance to Wind Farm^2 × sea proportion)

5. Each individual weight is then divided by the sum of all SPA weights within foraging range (for that species), to estimate the proportion of the population on the Wind Farm Site expected to originate from each SPA.

SPA proportion = SPA weight / $\sum$SPA weights

6. This proportion can then be multiplied by the Wind Farm Site population estimate to calculate how many individuals originate from each SPA. Dividing this figure by the SPA population size provides an estimate of the percentage of each SPA's population which is present on the Wind Farm Site. This can be undertaken in relation to both displacement and collision risk. An example of this method is presented below.

7. Given a peak Wind Farm Site population for species a of 500 individuals, thought to derive from three possible SPAs (with respectively, population sizes of 5,000, 6,000 and 10,0000, located 10km, 20km and 50km from the Wind Farm Site, with 50%, 60% and 80% of the area within the species' foraging range consisting of sea), the following SPA proportions would be calculated as shown in Table 1.1.

Table 1.1 Example of SPA apportioning calculations, based on a species present in different numbers at three SPAs located at different distances and with different proportions of sea area available.

SPA details				Weight	SPA Proportion	SPA Population on site	Proportion of SPA population on site
Site name	Population	Distance to site	Proportion of sea within foraging range				
1	5000	10	0.5	100	0.77	385	0.077
2	6000	20	0.6	25	0.19	96	0.016
3	10000	50	0.8	5	0.04	19	0.002

Weight = population/(distance^2 × sea proportion)
SPA Proportion = SPA weight / ∑SPA weights
SPA population on site = SPA Proportion × peak population on site

8. Once the peak Wind Farm Site population is apportioned amongst possible SPAs, it is then possible to estimate the percentage of each SPA's population present on the Wind Farm Site. If more than 1 % of the SPA's population is estimated to have been present on the Wind Farm Site that population is considered to be at risk of an LSE. A threshold of 1 % was used as this confers a precautionary level in line with the typical range of natural variation observed in seabird demographic rates. To illustrate, in the case of gannet (a comparatively well studied seabird species), the estimated standard deviation (SD) on average adult survival between 1959 and 2002 was 1.2 (Wanless *et al.* 2006). Since survival is measured on a scale of 0-100 this effectively means that 95 % of the time natural variation in survival for this species will lie ± 2.35 % of the mean value (i.e. SD × 1.96).

9. Adult survival is the demographic rate to which long-lived, slow breeding species such as seabirds are most sensitive to changes in, with the consequence that it tends to vary the least of all demographic rates (for example, equivalent estimates of the standard deviation on reproduction and juvenile survival for gannet are 3.5 % and 7.8 %). The gannet population from which these estimates are derived has undergone steady growth over the period of study. Given this, in combination with the period of study, it is highly unlikely that smaller standard deviations on adult survival would be obtained for any other seabird species. Furthermore, the gannet population has maintained positive population growth (approx. 2 % per year) while its demographic rates have varied by more than 1 %. Therefore, it is reasonable to state that if fewer than 1 % of an SPA's population is present on the Wind Farm Site, the risk to the population of an LSE is sufficiently small that it can be regarded as negligible.

10. On this basis, SPAs for which more than 1 % of their population is estimated to be present on the Wind Farm Site will be considered at risk of LSEs and will therefore be included in the assessment. SPAs for which less than 1 % of their populations are estimated to be present on the Wind Farm Site will be considered not at risk of LSE and will therefore not receive further consideration.

11. Thus, in the example in Table 1.1, 7.7 % of the population of SPA 1, 1.6 % of the population from SPA 2 and 0.2 % of the population from SPA 3 were estimated to be present on the Wind Farm Site. Therefore, SPA 1 and SPA 2 would receive further assessment while SPA 3 would not.

12. Since this approach requires pooling of data for each species across SPAs, for those species which may originate from multiple SPAs this stage of the assessment was initially conducted from a species perspective in order to identify which SPAs should be retained and which excluded.

13. The seabird species for which multiple SPAs have been considered are: fulmar, gannet, kittiwake, great black-backed gull, herring gull, guillemot, razorbill and puffin. In the following tables for each species, the SPAs for which more than 1 % of their populations are estimated to have been present on the Wind Farm Site are highlighted.

14. While this process has only been conducted using the distances of the SPAs from the Beatrice Wind Farm Site, the location of the Moray Firth Round 3 Eastern Development Area (hereafter MORL EDA) in relation to the SPAs under consideration would not lead to any additional sites being included since the MORL EDA is located farther offshore. The peak breeding season population used in these calculations was either the peak number recorded on the Beatrice site alone, or the combined total (with the MORL EDA) recorded during the same month. Thus this assessment considers the potential effects of both the Beatrice site alone and the combined Beatrice and MORL EDA sites.

2.2 Seabird apportioning

Fulmar

15. The combined breeding season peak abundance for estimating displacement was estimated to be 2,719 individuals in May 2010 (Beatrice: 879; EDA: 1,840). No collision estimates were supplied for the EDA, therefore the breeding season estimate for Beatrice (at an avoidance rate of 99 %) was used; 11. Table 1.2 provides the estimated SPA breakdown of the on-site population.

Table 1.2 Estimated proportion of on-site fulmar population deriving from each SPA within foraging range of the Beatrice Wind Farm and the EDA. Highlighted cells indicate SPAs which contribute more than 1 % to the on-site population at potential risk of LSE.

SPA details				Weight	SPA Proportion	Proportion of SPA population at risk of:			
						Displacement		Collision mortality	
Site name	Population	Distance to site	Proportion of sea within foraging range			BOWL	BOWL & MORL	BOWL	BOWL & MORL
East Caithness Cliffs	28404	11	0.813	288.74	0.792	0.0244	0.0758	0.0003	NA
North Caithness Cliffs	28336	29	0.832	40.50	0.111	0.0034	0.0107	0.0000	
Hoy	70000	57	0.847	25.44	0.070	0.0009	0.0027	0.0000	
Troup, Pennan and Lions Head	8800	62	0.782	2.93	0.008	0.0013	0.0025	0.0000	
Copinsay	3220	63	0.851	0.95	0.003	0.0007	0.0022	0.0000	
Rousay	2480	94	0.866	0.32	0.001	0.0003	0.0011	0.0000	
Calf of Eday	3910	97	0.871	0.48	0.001	0.0003	0.0010	0.0000	
West Westray	2800	107	0.873	0.28	0.001	0.0002	0.0008	0.0000	
Fair Isle	70420	150	0.899	3.48	0.010	0.0001	0.0004	0.0000	
North Rona and Sula Sgeir	23000	206	0.865	0.63	0.002	0.0001	0.0002	0.0000	
Noss	12700	220	0.944	0.28	0.001	0.0001	0.0002	0.0000	
Forth Islands	1596	255	0.695	0.04	0.000	0.0001	0.0002	0.0000	
Hermaness, Saxa Vord and Valla Field	39078	296	0.979	0.46	0.001	0.0000	0.0001	0.0000	

Weight = population/(distance2 × sea proportion)
SPA Proportion = SPA weight / ∑SPA weights
SPA population on site = SPA Proportion × peak population on site

16. For fulmar no SPAs were considered to be at risk of LSEs due to collision mortality. For the Beatrice site alone the East Caithness Cliffs SPA population was estimated to be at risk of LSE due to displacement (2.4%), while for the combined Beatrice and MORL EDA sites both the East Caithness Cliffs SPA (7.6%) and the North Caithness Cliffs SPA (1.1%) populations were estimated to be at risk of LSE due to displacement.

Gannet

17. The combined breeding season peak abundance for estimating displacement was estimated to be 409 individuals in April 2010 (Beatrice: 12; EDA: 397), while the highest abundance recorded just on the Beatrice site was 159. The combined breeding season collision mortality estimate (at an avoidance rate of 99 %) was 135 individuals (Beatrice: 54; EDA: 81). Table 1.3 provides the estimated SPA breakdown of the on-site population.

18. For gannet, an additional SPA was included in the analysis which is not designated for this species (Troup, Pennan and Lion's Head SPA), but hosts a gannet colony which is located much closer to the Wind Farm Site than any other SPA. Given its proximity it seems likely that most birds observed on the Wind Farm Site will originate from this colony, therefore not including it in the calculations would lead to greater attribution of birds to farther afield sites than is appropriate.

Table 1.3 Estimated proportion of on-site gannet population deriving from each SPA within foraging range of the Beatrice Wind Farm and the EDA. Highlighted cells indicate SPAs which contribute more than 1 % to the on-site population at potential risk of LSE.

SPA details				Weight	SPA Proportion	Proportion of SPA population at risk of:			
						Displacement		Collision mortality	
Site name	Population	Distance to site	Proportion of sea within foraging range			BOWL	BOWL & MORL	BOWL	BOWL & MORL
Troup, Pennan and Lions Head	3094	62	0.737	1.09	0.198	0.0146	0.0367	0.0049	0.0124
Sule Skerry & Sule stack	9350	130	0.832	0.66	0.108	0.0018	0.0046	0.0006	0.0016
Fair Isle	3750	150	0.909	0.18	0.030	0.0013	0.0032	0.0004	0.0011
North Rona and Sula Sgeir	18450	206	0.852	0.51	0.083	0.0007	0.0018	0.0002	0.0006
Noss	17304	220	0.965	0.37	0.060	0.0006	0.0014	0.0002	0.0005
Forth Islands	96130	255	0.635	2.33	0.378	0.0006	0.0016	0.0002	0.0005
Hermaness, Saxa Vord and Valla Field	31266	296	0.988	0.36	0.059	0.0003	0.0008	0.0001	0.0003

Weight = population/(distance2 × sea proportion)
SPA Proportion = SPA weight / ∑SPA weights
SPA population on site = SPA Proportion × peak population on site

19. For gannet, none of the SPAs for which this species is a qualifying feature within foraging range were estimated to contribute more than 1 % of their populations to the peak number observed on the Wind Farm Sites, either for the Beatrice site alone or the combined number for Beatrice and MORL. Troup, Pennan and Lion's Head SPA contributed the largest

proportion of birds (approx. 20 % of the total seen on site) however gannet are not a qualifying feature of this SPA so no HRA is required.

Kittiwake

20. The combined breeding season peak abundance for estimating displacement was estimated to be 7,149 individuals in May 2010 (Beatrice: 496; MORL EDA: 6,653)), while the highest abundance recorded just on the Beatrice site was 771. The combined breeding season collision mortality estimate (at an avoidance rate of 99 %) was 225 individuals (Beatrice: 62; EDA: 163). Table 1.4 provides the estimated SPA breakdown of the on-site population.

Table 1.4 Estimated proportion of on-site kittiwake population deriving from each SPA within foraging range of the Beatrice Wind Farm and the EDA. Highlighted cells indicate SPAs which contribute more than 1 % to the on-site population at potential risk of LSE.

SPA details						Proportion of SPA population at risk of:			
						Displacement		Collision mortality	
Site name	Population	Distance to site	Proportion of sea within foraging range	Weight	SPA Proportion	BOWL	BOWL & MORL	BOWL	BOWL & MORL
East Caithness Cliffs	80820	11	0.813	821.57	0.929	0.0089	0.0822	0.0007	0.0026
North Caithness Cliffs	20294	29	0.832	29.00	0.033	0.0012	0.0116	0.0001	0.0004
Hoy	6000	57	0.847	2.18	0.002	0.0003	0.0029	0.0000	0.0001
Troup, Pennan and Lions Head	63200	62	0.641	25.65	0.029	0.0004	0.0033	0.0000	0.0001
Copinsay	19100	63	0.87	5.53	0.006	0.0003	0.0023	0.0000	0.0001

Weight = population/(distance2 × sea proportion)
SPA Proportion = SPA weight / ∑SPA weights
SPA population on site = SPA Proportion × peak population on site

21. For kittiwake no SPAs were considered to be at risk of LSEs due to collision mortality for either Beatrice alone or the combined abundance across Beatrice and MORL EDA. No SPAs were assessed as being at risk due to displacement for Beatrice alone, however East Caithness Cliffs SPA and the North Caithness Cliffs SPA populations were estimated to be at risk of LSE due to displacement for the combined abundance across Beatrice and MORL EDA, with up to 8.2 % and 1.2 % of their populations present during the breeding season.

Great black-backed gull

22. The combined breeding season peak abundance for estimating displacement was estimated to be 93 individuals in August 2010 (Beatrice: 37; EDA: 56), while the highest abundance recorded just on the Beatrice site was 37. The combined breeding season collision mortality estimate (at an avoidance rate of 99 %) was 100 individuals (Beatrice: 62; EDA: 38). Table 1.5 provides the estimated SPA breakdown of the on-site population.

Table 1.5 Estimated proportion of on-site great black-backed gull population deriving from each SPA within foraging range of the Beatrice Wind Farm and the EDA. Highlighted cells indicate SPAs which contribute more than 1 % to the on-site population at potential risk of LSE.

SPA details				Weight	SPA Proportion	Proportion of SPA population at risk of:			
						Displacement		Collision mortality	
Site name	Population	Distance to site	Proportion of sea within foraging range			BOWL	BOWL & MORL	BOWL	BOWL & MORL
East Caithness Cliffs	**360**	**11**	**0.562**	**5.29**	**0.816**	**0.0839**	**0.2108**	**0.1406**	**0.2267**
Hoy	**1140**	**57**	**0.752**	**0.47**	**0.072**	**0.0023**	**0.0059**	**0.0039**	**0.0063**
Copinsay	**2378**	**63**	**0.825**	**0.73**	**0.112**	**0.0017**	**0.0044**	**0.0029**	**0.0047**

Weight = population/(distance2 × sea proportion)
SPA Proportion = SPA weight / ∑SPA weights
SPA population on site = SPA Proportion × peak population on site

23. For great black-backed gull the East Caithness Cliffs SPA population was estimated to be at risk of LSE due to both displacement and collision mortality for both Beatrice alone (8 % and 14 % respectively) and for Beatrice and MORL EDA combined (21 % and 23 % respectively).

Herring gull

24. No density estimates were provided for herring gull for the EDA, so the peak breeding season estimate on the Beatrice site was used: 19 individuals. The combined breeding season collision mortality estimate (at an avoidance rate of 99 %) was 55 individuals (Beatrice: 29; EDA: 26). Table 1.6 provides the estimated SPA breakdown of the on-site population.

Table 1.6 Estimated proportion of on-site herring gull population deriving from each SPA within foraging range of the Beatrice Wind Farm and the EDA. Highlighted cells indicate SPAs which contribute more than 1 % to the on-site population at potential risk of LSE.

SPA details				Weight	SPA Proportion	Proportion of SPA population at risk of:			
						Displacement		Collision mortality	
Site name	Population	Distance to site	Proportion of sea within foraging range			BOWL	BOWL & MORL	BOWL	BOWL & MORL
East Caithness Cliffs	6786	11	0.653	85.88	0.968	0.0027	**NA**	0.0041	0.0078
Troup, Pennan and Lions Head	4220	49	0.615	2.86	0.032	0.0001	**NA**	0.0002	0.0004

Weight = population/(distance2 × sea proportion)
SPA Proportion = SPA weight / ∑SPA weights
SPA population on site = SPA Proportion × peak population on site

25. For herring gull, none of the SPAs within (or just beyond) the mean maximum foraging range were estimated to be at risk of LSEs for either displacement or collision mortality for either Beatrice alone or the combined estimates for Beatrice and MORL EDA.

Common guillemot

26. The combined breeding season peak abundance for estimating displacement was estimated to be 20,885 individuals in May 2010 (Beatrice: 5,180; EDA: 15,705), while the highest abundance recorded just on the Beatrice site was 5,180. No collision mortality estimates were provided for the EDA, so the Beatrice breeding season estimate was used here: 11 individuals. Table 1.7 provides the estimated SPA breakdown of the on-site population.

Table 1.7 Estimated proportion of on-site guillemot population deriving from each SPA within foraging range of the Beatrice Wind Farm and the EDA. Highlighted cells indicate SPAs which contribute more than 1 % to the on-site population at potential risk of LSE.

SPA details						Proportion of SPA population at risk of:			
						Displacement		Collision mortality	
Site name	Population	Distance to site	Proportion of sea within foraging range	Weight	SPA Proportion	BOWL	BOWL & MORL	BOWL	BOWL & MORL
East Caithness Cliffs	158985	11	0.813	1616.14	0.916	0.0298	0.1203	0.0001	NA
North Caithness Cliffs	70584	29	0.832	100.88	0.057	0.0042	0.0169	0.0000	NA
Hoy	26800	57	0.847	9.74	0.006	0.0011	0.0043	0.0000	0.0000
Troup, Pennan and Lions Head	44600	49	0.641	28.98	0.016	0.0019	0.0077	0.0000	0.0000
Copinsay	29450	63	0.87	8.53	0.005	0.0009	0.0034	0.0000	0.0000

Weight = population/(distance2 × sea proportion)
SPA Proportion = SPA weight / ∑SPA weights
SPA population on site = SPA Proportion × peak population on site

27. For guillemot no SPAs were considered to be at risk of LSEs due to collision mortality. The East Caithness Cliffs SPA population was estimated to be at risk of LSE due to displacement for both the Beatrice site abundance alone (3 % of the SPA population) and also for the combined abundance for Beatrice and MORL EDA (12 % of the SPA population). The North Caithness Cliffs SPA population was estimated to be at risk of LSE due to displacement for the combined abundance across both sites with 1.7 % of the population present.

Razorbill

28. The combined breeding season peak abundance for estimating displacement was estimated to be 5,525 individuals in May 2010 (Beatrice: 331; EDA: 5,194), while the highest abundance recorded just on the Beatrice site was 537. No collision mortality estimates were provided for the EDA, and the Beatrice breeding season estimate was 0, so no assessment of collision mortality was necessary. Table 1.8 provides the estimated SPA breakdown of the on-site population.

Table 1.8 Estimated proportion of on-site razorbill population deriving from each SPA within foraging range of the Beatrice Wind Farm and the EDA. Highlighted cells indicate SPAs which contribute more than 1 % to the on-site population at potential risk of LSE.

SPA details				Weight	SPA Proportion	Proportion of SPA population at risk of:			
						Displacement		Collision mortality	
Site name	Population	Distance to site	Proportion of sea within foraging range			BOWL	BOWL & MORL	BOWL	BOWL & MORL
East Caithness Cliffs	17830	11	0.813	181.25	0.980	**0.0295**	**0.3035**	NA	NA
North Caithness Cliffs	2643	29	0.832	3.78	0.020	0.0041	**0.0427**	NA	NA

Weight = population/(distance2 × sea proportion)
SPA Proportion = SPA weight / ∑SPA weights
SPA population on site = SPA Proportion × peak population on site

29. For razorbill no SPAs were considered to be at risk of LSEs due to collision mortality. The East Caithness Cliffs SPA population was estimated to be at risk of LSE due to displacement for both the Beatrice site abundance alone (3.0 % of the SPA population) and also for the combined abundance for Beatrice and MORL EDA (30 % of the SPA population). The North Caithness Cliffs SPA population was estimated to be at risk of LSE due to displacement for the combined abundance across both sites with 4.3 % of the population present.

Puffin

30. The combined breeding season peak abundance for estimating displacement was estimated to be 3,217 individuals in May 2010 (Beatrice: 166; EDA: 3,051), while the highest abundance recorded just on the Beatrice site was 1,455 No collision mortality estimates were provided for the EDA, and the Beatrice breeding season estimate was 0, so no assessment of collision mortality was necessary. Table 1.9 provides the estimated SPA breakdown of the on-site population.

Table 1.9 Estimated proportion of on-site puffin population deriving from each SPA within foraging range of the Beatrice Wind Farm and the EDA. Highlighted cells indicate SPAs which contribute more than 1 % to the on-site population at potential risk of LSE.

SPA details				Weight	SPA Proportion	Proportion of SPA population at risk of:			
						Displacement		Collision mortality	
Site name	Population	Distance to site	Proportion of sea within foraging range			BOWL	BOWL & MORL	BOWL	BOWL & MORL
East Caithness Cliffs	3500	11	0.813	35.58	0.611	0.2539	0.5613	NA	NA
North Caithness Cliffs	14090	29	0.832	20.14	0.346	0.0357	0.0789	NA	NA
Hoy	7000	57	0.847	2.54	0.044	0.0091	0.0201	NA	NA

Weight = population/(distance2 × sea proportion)
SPA Proportion = SPA weight / ∑SPA weights
SPA population on site = SPA Proportion × peak population on site

31. For puffin no SPAs were considered to be at risk of LSEs due to collision mortality. The East Caithness Cliffs SPA population was estimated to be at risk of LSE due to displacement for both the Beatrice site abundance alone (25 % of the SPA population) and also for the combined abundance for Beatrice and MORL EDA (56 % of the SPA population). The North Caithness Cliffs SPA population was estimated to be at risk of LSE due to displacement for both the Beatrice site abundance alone (3.6 % of the SPA population) and also for the combined abundance for Beatrice and MORL EDA (7.9 % of the SPA population).

Other species

32. For several of the species included for assessment there is only one SPA within foraging range (Arctic skua, great skua, shag and cormorant). It was unnecessary to apportion individuals from these species amongst potential candidate SPAs, however for each species an estimate of the percentage of the SPA population present on the Wind Farm Sites during the breeding season was made (Table 1.10).

Table 1.10 Estimated proportion of on-site population for great cormorant, European shag, Arctic skua and great skua deriving from qualifying SPAs within foraging range of the Beatrice Wind Farm and the EDA. Highlighted rows indicate SPAs which contribute more than 1 % to the on-site population at potential risk of LSE.

| Species | SPA | SPA population | Peak breeding season abundance | | Breeding season collision mortality | | Proportion of SPA population at risk of: | | | |
| | | | | | | | Displacement | | Collision mortality | |
			BOWL	BOWL & MORL	BOWL	BOWL & MORL	BOWL	BOWL & MORL	BOWL	BOWL & MORL
Great cormorant	East Caithness Cliffs	188	0	0	0	0	NA	NA	NA	NA
European shag	East Caithness Cliffs	2112	5	5	0	0	0.0023	0.0023	NA	NA
Arctic skua	Hoy	118	30	30	6	6	0.25	0.25	0.051	0.051
Great skua	Hoy	3946	57	542	13	13	0.014	0.14	0.003	0.003

Weight = population/(distance2 × sea proportion)
SPA Proportion = SPA weight / ∑SPA weights
SPA population on site = SPA Proportion × peak population on site

33. No LSEs were identified for the SPA populations of cormorant or shag, but the Hoy SPA population of Arctic skua was assessed to be at risk of LSEs for both displacement and collision mortality and the Hoy SPA population of great skua was assessed to be at risk of an LSE for displacement. In both cases these assessments applied to the abundance on the Beatrice site alone and also the combined Beatrice and MORL EDA sites.

3 GREAT BLACK-BACKED GULL POPULATION MODEL

3.1 Assessment of potential collision mortality impacts on great black-backed gull

34. The SPA population for which the greatest potential mortality impact was identified was the East Caithness Cliffs SPA population of great black-backed gulls. To fully explore the potential effect on this population from additional mortality a stochastic population model was developed. Details of this model are provided below.

3.1.1 Estimation of collision mortality

35. The collision risk modelling presented in the Ornithological Technical Appendix (14a) indicated that the majority of the collision mortality predicted to occur due to the Beatrice Wind Farm would take place outside the breeding season. During the non-breeding season it is not possible to determine the origin of birds with regards to their breeding colonies, therefore this non-breeding mortality is not considered here in the context of HRA.

36. Approximately 20 % of the mortality was predicted to occur during the breeding season, amounting to a combined (BOWL and MORL EDA) breeding season mortality of 100 individuals (at an avoidance rate of 99 %, Cook *et al.* 2012). The most recent estimate of the East Caithness Cliffs SPA population size was 180 pairs (SNH 2008). The predicted proportion of the mortality assigned to the East Caithness Cliffs SPA population was 81.6 % (Table 1.5), hence an annual loss of 82 individuals from this population.

37. Over 20 % of great black-backed gulls observed during the boat surveys were aged (as either adults or immature birds) on the basis of plumage. Across all surveys the percentage of adult birds was 39.5 % while during the breeding months (May – August inc.) this was 37.5 %. Therefore, only 37.5 % of the 82 individuals at risk of collision would be expected to be adults, which equates to 31 individuals. Furthermore, some proportion of these adults are likely to be non-breeding individuals.

38. Compared to other seabird species such as skuas (Catry *et al.* 1998) and auks (Harris and Wanless 1994), gulls have relatively large proportions of non-breeders in a population. Calladine and Harris (1996) estimated that within a lesser black-backed gull colony at the Isle of May, east of Scotland, 34 % of adults in 1993, and 40 % in 1994 did not breed. This was considered to be a 'normal' period, unaffected by culling measures which occurred in some other years. These results are similar to those from other studies of gull populations. Kadlec and Drury (1968) estimated that 15-30 % of adult North American herring gulls did not breed in any one year, and Pugesek and Diem (1990) estimated that 36 % of Californian gulls did not breed. Samuels and Ladino (1984) estimated that 45 % of herring gulls did not breed in a North American study.

39. It could therefore be reasonably concluded that as a conservative estimate, for every two breeding birds recorded, another non-breeding individual is present within the SPA population. Since the SPA population estimate is based on breeding pairs, this effectively increases the East Caithness Cliffs SPA population from 360 individuals to around 540. This would mean that approximately one in three adult birds at risk of collision would be a non-breeder, assuming that all birds from the SPA use the site equally. In reality it is very likely that the proportion of non-breeders encountered will increase with distance offshore, since these individuals are not constrained by the demands of incubation and feeding chicks. Therefore non-breeders are more likely to spend longer periods of time farther away from the colony, and range more widely than breeders.

40. Consequently, of the estimated 31 adults at risk of collision, it is likely that no more than 20 would be breeding birds.

3.1.2 Population model structure

41. The stochastic population model produced for the great black backed gull followed best practice methods (e.g. selection of appropriate probability distributions for survival and reproduction). The model was based on the best available demographic data (Table 2.1). However, this species has not been well studied, and consequently it was necessary to use values from closely related species for certain demographic rates. This is discussed below.

Table 2.1 Input parameters used for in the great black backed gull population model.		
Parameter	**Value used (SD)**	**Source**
Juvenile survival rate	0.82 (0.03)	Wanless *et al.* (1996)
Age of first breeding	4 years	Snow and Perrins (1998)
Adult survival rate	0.93 (0.025)	Garthe & Hüppop (2004), Maclean *et al.* (2007)
Proportion of breeding adults in population	0.66	Calladine and Harris (1996) Reeves and Furness 2002 Poot *et al.* (2011).
Reproductive rate	1.11 (0.54) chicks per pair	Mavor et al (2008)
Sex ratio	50:50	NA
Migration	Closed population	NA
Density dependence	Density independent	NA

42. Juvenile survival was taken from a study of herring gulls (Wanless et al. 1996). The compound four year rate of 0.45 (0 – 4 years) given by Wanless et al. (1996) for survival from fledging to breeding age was converted to an annual rate for use in the model by raising it to the power 0.25.

43. Age of first breeding was set at 4 (Snow and Perrins 1998). Average adult survival was reported in Garthe and Hüppop (2004). No estimate of variance was found for this species, therefore the equivalent rate estimated for herring gulls (0.025, Wanless et al. 1996) was used.

44. Calladine and Harris (1996) estimated that 66 % of breeding age lesser black backed gulls on the Isle of May bred each year. This level of non-breeding is also consistent with estimates provided elsewhere for seabirds (e.g. Reeves and Furness 2002, Poot et al. 2011).

45. There is no information on rates of exchange (i.e. immigration and emigration) between breeding colonies, so a closed population was assumed. Similarly there is no information on which to base density dependent population regulation, hence the model was density independent. While this is clearly unrealistic in the longer term, for the benefits of short term modelling of small populations, the risks from violating this assumption were considered to be small.

46. The population was modelled on an annual time step, using a five age class model, split into the following age classes: 0-1, 1-2, 2-3, 3-4, 4+. It was assumed that only the final age class (4+) breeds. The model was structured around a post-breeding census (i.e. each census of the modelled population occurs immediately after the breeding season). The same underlying survival rate was used for the first 4 age classes, although for each age class an independent random rate was generated at each time step during simulations.

47. The presence of birds in the final non-breeding age class (3-4) in the region during the breeding season, and therefore the potential for them to be in collision with turbines was

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incorporated by applying additional mortality to the last two age classes in proportion to their relative proportions (this was typically in the ratio of 1 : 9).

48. Environmental stochasticity was modelled using the mean rates and the standard deviations as listed in Table 2.1. Survival rates were drawn from a beta distribution, and brood sizes from a lognormal distribution. These distributions were used as they generate random numbers with characteristics appropriate to the demographic rates (i.e. survival between 0 and 1, and brood sizes which cannot be negative).

49. Demographic stochasticity on survival was modelled using a binomial process, whereby the number of individuals which survive from one time step to the next was estimated using a binomial function (Akcakaya 1991). Thus, the number of individuals alive at time t+1 is generated by a 'coin-toss' process, using the number of individuals alive at time t and the randomly generated survival rate for that time step (as described in the preceding point).

50. [NB: The difference between environmental and demographic stochasticity can be thought of as follows: Environmental stochasticity generates random values for the probability of survival from one time step to the next. Demographic stochasticity generates random numbers of individuals which survive from one time step to the next for any given survival probability. Thus environmental stochasticity models variable environments (e.g. weather effects) while demographic stochasticity models the effects of chance, which are increasingly important as the population size falls.]

51. Additional mortality, applied to the final two age classes in proportion to their presence in the population was modelled across a range of values, from 0 to 50 at intervals of 5. Juveniles were not included in the mortality as the extent to which birds of this age class are associated with any particular SPA is not know and birds of this age typically disperse quite widely. In order to reflect the fact that collision mortality would be more likely to operate as a per capita rate, rather than an absolute value, the actual number killed at each time step was made proportional to the population size as follows. At the beginning of each simulation, the number to be killed each year (as set for that simulation, between 0 and 50) was used to calculate the proportion of the sub-adult and adult age classes which this number represented. For example, if the additional mortality was set at 10 individuals, this would be split between the sub-adult and adult age classes and then the proportion of the initial population that these numbers represented calculated. In this manner the additional absolute mortality (as generated by collision risk modelling) was converted into a proportional mortality rate. Thus, additional mortality remained at the same proportional level relative to the population size throughout the simulation, whether the population increased or decreased.

52. The average population growth rate, and the upper and lower 95% confidence intervals were calculated across all simulations (5,000). Since matrix based population models such as this one are prone to producing unrepresentative outputs for the first few time-steps (while the stable age distribution is achieved), the growth rate was estimated between the 5th and final year of each 25 year simulation.

3.1.3   Model results

53. From an initial population size of 180 pairs, the model predicted a mean annual population growth rate of 6.7 % (95 % confidence interval: 4.3 % - 9.5 %, Figure 1). Thus the average prediction would be for an increase in the number of breeding pairs from 180 to 725 after 25 years, assuming no additional mortality due to collisions with turbines.
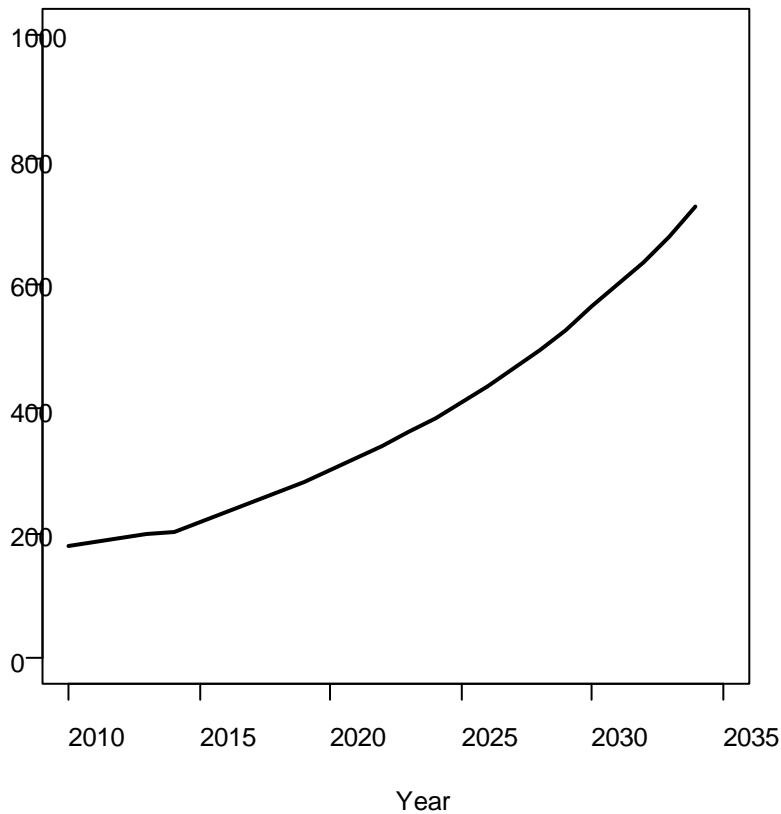
Figure 1. Predicted average population growth rate of the East Caithness Cliffs SPA great black-backed gull with no additional mortality. The black line is the average number of breeding pairs and the red dashed lines are the 95 % confidence intervals, derived from 5000 simulations.

54. This level of baseline population growth is at odds with the reported trend for the East Caithness Cliffs SPA population, which was reported to have declined from 800 pairs in the period 1985-88 to 180 in the period 1998-2002 (Mitchell et al. 2004). With regards to the validity of the model for predicting the potential effects of additional mortality, it is worth considering why the SPA population decreased. Allied to this is the question of whether this is likely to continue, or if the population will undergo a recovery along the lines predicted by the model (i.e. at a rate of approx. 6 % per year).

55. Given the methods employed for counting great black-backed gulls (Mitchell et al. 2004), and the inaccessibility of much of the East Caithness Cliffs SPA breeding sites, it seems plausible that some breeding pairs will have been missed from the total of 180 pairs reported (SNH 2008). Indeed, Mitchell et al. (2004) acknowledge the uncertainty in these estimates, although they do describe the counts presented as being comparable to the previous estimates (collected between 1985-88 for the Seabird Colony Register). Thus, these figures provide an index of population size, rather than accurate counts. The apparent decline in this population has not been reflected elsewhere, with both the British and Irish population as a whole, and the Scottish population, fluctuating around the same levels.

56. Furness (1996) considered that increases in great skua populations may have led to local reductions in great black-backed gull populations due to the former species being competitively dominant in obtaining fisheries discards (at sea). The great skua population on Orkney (the nearest breeding colonies) increased considerably between 1969 and 1985-88 (from 88 pairs to 2000, Mitchell et al. 2004), but since then the population has increased only slightly. Therefore the opposite trends in the two species do not match temporally. However, it does remain possible that the large great skua population has had a negative effect on the great black backed gull one in the region.

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57. If this is the source of the decline in the great black-backed gull population, the next question is whether this will continue (or indeed has already continued since the Seabird 2000 counts). The most recent seabird trend reporting (JNCC 2009) for the UK wide population indicates that the great black backed gull population has remained more or less stable since 2000. The most recent regional breakdown (although not including NE Scotland as no sites are monitored there) reveals small variations in numbers across the sites monitored, but no consistent patterns of increase or decline (Mavor et al. 2008). There is therefore no indication that the UK great black backed gull population is undergoing a decline in numbers.

58. It is difficult to conclude from the wider picture how the East Caithness Cliffs SPA population may change in the future, since the cause of the decline between 1985-88 and 1998-2002 is not known. However, given the national population trend it is reasonable to assume that the population could increase at the rate predicted by the model, at least in the short term.

59. Simulation of the effect of additional mortality was conducted to predict the potential effect on the population of additional mortality. As additional mortality increases the population growth rate declines (Figure 2). The point at which the lower 95 % confidence interval of the population growth rate (i.e. the value of the growth rate exceeded by 95 % of simulated population projections) crossed the line of stable growth (=1) was used as a precautionary level of risk. For a breeding population of 180 pairs this occurred at an additional mortality of 32 individuals (Figure 2). The average population growth rate at this level of mortality was 3.7 % per year.
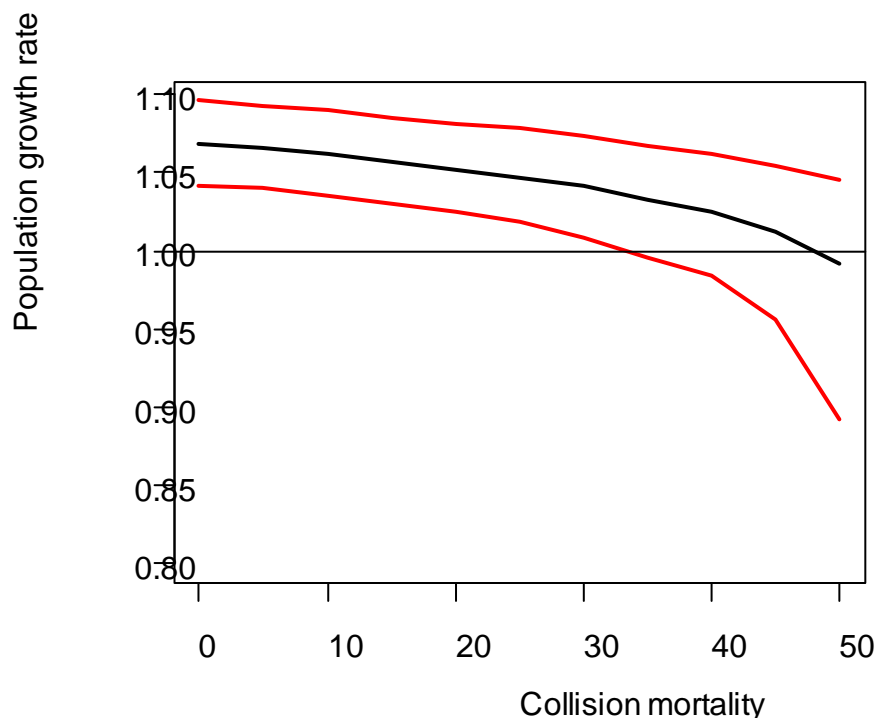

Figure 2. Population growth rate of the East Caithness Cliffs SPA great black-backed gull in relation to additional mortality

60. This indicates that for 95 % of simulated population projections, the predicted level of collision mortality for this population would not trigger a decline.

61. If the initial number of pairs used in the model was increased to 210 (i.e. 30 additional pairs) the point at which the lower 95 % population growth rate line crosses the point of stable growth occurs with an increase in additional mortality of 37 individuals.

62. Thus, with only a small increase in the starting population size used in the model (i.e. one which could plausibly reflect the true population size) the East Caithness Cliffs SPA

population would be considered to be at no risk of population decline as a result of the predicted additional mortality due to collisions with turbines.

63. Overall therefore, given the difficulty of censusing this species, with the consequence that the East Caithness Cliffs population size is very likely to be greater than 180 pairs, and the model prediction that a modest increase in the breeding population is sufficient to prevent population decline in more than 95 % of simulations, the risk to the viability of the great black-backed gull population as a component of the East Caithness Cliffs SPA is not considered to be significant.

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