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Abbreviations and Acronyms

CEH	Centre for Ecology and Hydrology
GAM	Generalised Additive Model
GPS	Global Positioning System
ICOL	Inch Cape Offshore Limited
MS LOT	Marine Scotland Licensing Operations Team
n/a	Not applicable
SD	Standard Deviation
SNCB	Statutory Nature Conservation Bodies
SNH	Scottish Natural Heritage
SPA	Special Protection Area
WTG	Wind Turbine Generator

11D Estimation of the Development Alone and Cumulative Effects from Displacement and Barrier Effects

11D.1 Introduction

1 The Scoping Opinion from Marine Scotland Licencing and Operations Team (MS LOT) identifies six key seabird species on which potential impacts from the Development should be assessed (*Appendix 11A Offshore Ornithology Baseline Survey Report*). Of these six species, the Scoping Opinion advises that displacement and barrier effects should be considered for four – i.e. kittiwake, guillemot, razorbill and puffin.

2 Displacement is defined as 'a reduced number of birds occurring within or immediately adjacent to an offshore wind farm' (Furness et al. 2013) and involves birds present in the air and on the water (SNCBs 2017). Birds that do not intend to utilise a wind farm area but would have previously flown through the area on the way to a feeding, resting or nesting area, and which either stop short or detour around a development, are subject to barrier effects (SNCBs, 2017). For the purposes of assessment, however, it is usually not possible to distinguish between displacement and barrier effects (for example to define where individual birds may have intended to travel to, or beyond an offshore wind farm, even when tracking data are available). Therefore, in this assessment the effects of displacement and barrier effects on the key seabird species are considered together.

3 The overall predicted effect of displacement from an offshore wind farm is a change in the abundance of birds within the wind turbine generator (WTG) array, between the baseline (pre-construction) and construction and/or operational phases of the wind farm (although effects are only likely to be ecologically significant if they extend into the operational phase, given the relatively short-term nature of construction). There are likely to be several main affects which may cause this change in abundance, such as behavioural avoidance of a WTG array, changes in prey abundance, availability or distribution and disturbance from associated anthropogenic activities (e.g. boat traffic, helicopter traffic, presence of maintenance personnel, etc.). These different effects cannot be distinguished on the basis of existing post-construction monitoring data, which are generally designed to detect only whether changes in abundance and distribution occur (as opposed to the causes of any such changes). Furthermore, changes in abundance within a wind farm can also occur as a result of attraction (e.g. from reef effects causing increases in prey and the structures providing roosting sites – Dierschke *et al*., 2016)).

4 An offshore wind farm may represent a barrier to movements so that birds fly around the WTG array, where in the absence of the wind farm, they would have taken a more direct route to their destination, which is assumed be to beyond the wind farm. Thus, there may be an effect of additional energy use to reach a destination. These barrier effects have been suggested to be unimportant for migratory movements where the additional distance needed to fly around the wind farm is trivial compared to the overall distance flown on migration (Masden *et al*., 2009). However, the accumulated effects of a bird flying around one or more wind farms on foraging trips during the breeding season, when seabirds are central place foragers (so constraining their foraging distribution by the need to return to

their nest) has the potential to be important (Masden *et al.*, 2010). In these cases, the additional energetic requirements caused by the wind farm acting as a barrier has the potential to affect adult survival and/or breeding productivity (Searle *et al.*, 2014).

5 This report presents the details of the approach and methods used to estimate the potential displacement and barrier effects for the assessment on the four seabird species identified above, along with the outputs from the resulting calculations. This is undertaken both for the Wind Farm alone and cumulatively with the other three Forth and Tay wind farms (i.e. Neart na Gaoithe, Seagreen Alpha and Seagreen Bravo). In addition, as advised by the Scoping Opinion, comparisons are made between the estimates of displacement and barrier effects used in the assessment with those produced by alternative, individual-based, modelling approaches which simulate the behaviour and energetics of individual birds from Special Protection Area (SPA) breeding colonies - i.e. the Searle *et al*. (2014) and SeabORD[1] models.

11D.2 Estimation of Displacement and Barrier Effects for the Purposes of the Assessment

11D.2.1 The SNCB Matrix Approach

6 In the absence of strong empirical evidence for displacement and barrier effects on seabirds from offshore wind farms, a matrix approach to assessing the impacts has been recommended by MS LOT, following advice from Scottish Natural Heritage (SNH) (and as set out in the Statutory Nature Conservation Bodies (SNCBs) (2017) advice note). The matrix provides a table of the displacement rates, from zero per cent to 100 per cent, against mortality rates, again from zero per cent to 100 per cent. Thus, for a given population-size and any combined value of displacement rate and mortality rate, the matrix provides a prediction of the number of birds that may die as a result of displacement from the wind farm. In their Scoping Opinion, MS LOT provided recommended seasonally specific displacement rates and mortality rates for each of the four relevant species (*Table 1.1*). Although the estimated effects are derived by applying specified displacement rates, the resulting predicted impacts are assumed to encompass both displacement and barrier effects.

7 Following the advice of the Scoping Opinion, impacts from displacement and barrier effects were estimated by applying the specified displacement and mortality rates to the peak seasonal population estimates averaged across the two years of baseline survey (i.e. the mean peak population size). These population estimates included both birds on the water and in flight. The seasonal periods used for each species are as defined in the Scoping Opinion, and as set out below in *Section 11D.2.3*.

8 The Scoping Opinion recommended that displacement was assessed for the Development Area and a two kilometre buffer, whereas the baseline surveys (and the associated analyses of bird densities) were undertaken for the Development Area plus four kilometre buffer

[1] At the time of writing, the SeabORD model and associated documentation was unpublished.

(subsequently referred to as the Survey Area)[2]. Thus, as agreed in the clarifications to the Scoping Opinion (letter of 17 October 2017 from MS LOT to Inch Cape Offshore Limited (ICOL)), population sizes for the two kilometre buffer were estimated by extrapolation from those calculated for the four kilometre buffer (based on the differences in the areas of each buffer – i.e. 280.4 kilometres squared and 128.1 kilometres squared for the four and two kilometre buffer, respectively).

9 Figures showing the distribution of the four seabird species within the Survey Area during each of the surveys which contribute to the mean peak seasonal population estimates used in the displacement assessment are presented in *Annex 11D.1*. Visual inspection of these distributions within the buffer areas gives no indication of any strong or systematic bias which could cause underestimation of population sizes within the two kilometre buffer as a consequence of extrapolating from the four kilometre buffer. Such a bias could occur if densities were lower in the two to four kilometre buffer zone than in the zero to two kilometre zone but, based on the visual inspection, this only appears to occur for the 2011 breeding period peak count for razorbill (*Figure 11D.1.4*), whilst the opposite appears to occur in several of the other peak count surveys (e.g. *Figures 11D.1.1* (for July 2012), *11D.1.5* (for October 2010) and *11D.1.6* (for June 2011)).

Table 11D.1 Displacement and mortality rates used in the matrix assessment, as recommended in the Scoping Opinion.

Species	Breeding season		Non-breeding season	
	Displacement	**Mortality**	**Displacement**	**Mortality**
Kittiwake	30%	2%	Qualitative assessment requested[1]	
Guillemot	60%	1%	60%	1%
Razorbill	60%	1%	60%	1%
Puffin	60%	2%	No assessment requested	

[1]The qualitative assessment for kittiwakes in the non-breeding season is undertaken in relation to the SPA populations with connectivity to the Development Area and two kilometre buffer and is detailed in the *Inch Cape Wind Farm and Offshore Transmission Works Habitats Regulation Appraisal* (ICOL, 2018).

11D.2.2 Species accounts

10 Throughout the species accounts, tables have been colour coded (green for breeding season, amber for non-breeding season, or in the case of the kittiwake non-breeding periods, amber for spring passage and peach for autumn passage) to make the recommended seasons clear. The Scoping Opinion from MS LOT stated that the impacts from displacement should be based on breeding and non-breeding seasons.

[2]The surveys and subsequent analyses to estimate bird densities used a four kilometre buffer on the basis of advice provided by SNH prior to the commencement of the boat-based surveys.

Kittiwake

11 Kittiwakes were generally more abundant in the Development Area and two kilometre buffer during the breeding season than the non-breeding season (*Table 11D.2*). Mean abundance was slightly higher, and peak abundance much higher, in the first breeding season (2011) than the second (2012). Peak abundance occurred in July in the first breeding season and in June in the second.

12 The Scoping Opinion advised that only a qualitative assessment of displacement was required for kittiwakes in the non-breeding period. As such, the peak population size estimates for kittiwake in the non-breeding period are not considered in detail.

Table 11D.2 Estimated abundance of kittiwake in each month of survey. Seasons are colour coded (green = breeding season, peach = autumn passage (non-breeding), amber = spring passage (non-breeding)). Peak abundance for each breeding period shown in bold.

Survey number	Month[1]	Year	Development Area	Buffer (2 km)	Total
1	Sept	2010	26	251	277
2	Oct	2010	1147	1109	2256
3	Dec	2010	15	84	99
4	Jan	2011	196	393	589
5	Feb	2011	83	87	170
6	Mar	2011	104	142	246
7	Apr	2011	296	324	620
8	May	2011	1153	640	1793
9	Jun	2011	1441	1612	3053
10	Jul	2011	**2344**	**2700**	**5044**
11	Aug	2011	561	159	720
12	Sept	2011	1106	911	2017
13	Oct	2011	242	304	546
14	Nov	2011	260	1346	1606
15	Dec	2011	339	301	640
16	Jan	2012	279	625	904
17	Feb	2012	14	335	349
18	Mar	2012	729	354	1083
19	Apr	2012	136	1138	1274
20	May	2012	1503	649	2152
21	Jun	2012	**1894**	**794**	**2688**

Survey number	Month[1]	Year	Development Area	Buffer (2 km)	Total
22	Jul	2012	501	1209	1710
23	Aug	2012	459	460	919
24	Sept	2012	238	300	538
[1]The the kittiwake breeding period was assumed to be mid-April to August (as advised in the Scoping Opinion). The April surveys were allocated to the breeding period, although their inclusion did not affect the seasonal peak estimates.					

13 The breeding period mean peak estimate was 3,866 birds (*Table 11D.3*), which is the value used in the displacement analysis by the matrix approach.

14 In order to better understand the distribution of the kittiwake breeding season abundance data, a frequency distribution was plotted (*Figure 11D.1*). The modal abundance occurred in the lowest category of abundance (620 – 1,120 birds). The mean peak value was much larger than both the mean and the median abundance of kittiwakes across both breeding seasons (*Table 11D.3*), giving an indication of the likely level of precaution provided by the use of this metric in the assessment.

Figure 11D.1 Frequency distribution of kittiwake abundance estimates in the Development Area and two kilometre buffer in the breeding season. The mean peak value is shown as a dark green line in the appropriate abundance bin.

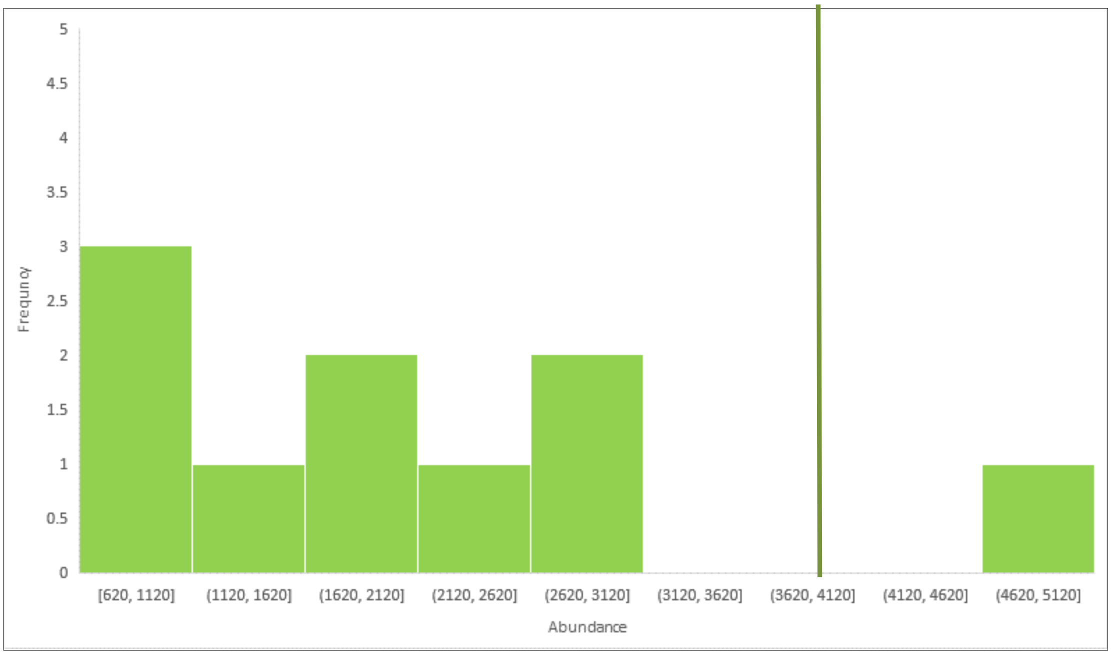


Table 11D.3 Summary statistics for kittiwake abundance in the breeding season.

Period	Mean	SD
First breeding season	2246	1848
Second breeding season	1749	700

All breeding seasons mean	1997	1343
All breeding seasons median	1752	n/a
Mean peak	3866	1666
Counts less than peak mean	9	
Total counts	10	
% less than PM	90.0%	

Guillemot

15 Guillemots were more abundant in the Development Area and two kilometre buffer, during the breeding season than the non-breeding season (*Table 11D.4*). Mean abundance was higher in the second breeding season (2012) than the first (2011), but the peak abundance was higher in the first. Peak abundance occurred in June in the first breeding season, and in July in the second.

16 In the non-breeding season, mean abundance was slightly higher in the second season of study (2011/12) than in the first (2010/11). Peak abundance was also higher in the second non-breeding season than in the first.

17 It should be noted that the final survey occurred in September 2012 and was the only sample in the third non-breeding season. Since this was the only sample in this season it is excluded from the analysis, although its inclusion would not change any of the assessments or conclusions.

Table 11D.4 Estimated abundance of guillemot in each month of survey. Seasons are colour coded (green = breeding season, amber = non-breeding). Seasonal peak abundances shown in bold.

Survey number	Month[1]	Year	Development Area	Buffer (2km)	Total
1	Sept	10	421	546	967
2	Oct	10	835	1116	1951
3	Dec	10	282	758	1040
4	Jan	11	886	977	1863
5	Feb	11	344	599	943
6	Mar	11	**1808**	**1384**	**3192**
7	Apr	11	137	248	385
8	May	11	1466	1210	2676
9	Jun	11	**4545**	**5389**	**9934**
10	Jul	11	2396	2086	4482

Survey number	Month[1]	Year	Development Area	Buffer (2km)	Total
11	Aug	11	769	385	1154
12	Sept	11	**2210**	**2422**	**4632**
13	Oct	11	180	312	492
14	Nov	11	777	1089	1866
15	Dec	11	606	784	1390
16	Jan	12	986	792	1778
17	Feb	12	419	385	804
18	Mar	12	818	557	1375
19	Apr	12	532	726	1258
20	May	12	1270	1391	2661
21	Jun	12	2843	2206	5049
22	Jul	12	**3549**	**2884**	**6433**
23	Aug	12	2883	3285	6168
24	Sept	12	467	940	1407

[1]The guillemot breeding period was assumed to be April to mid-August (as advised by the Scoping Opinion). Both August surveys were included within the breeding period as they occurred in the first half of the month.

18 The breeding season mean peak estimate was 8,184 birds (*Table 11D.5*), which is the value used in the displacement analysis by the matrix approach.

19 In order to better understand the distribution of guillemot breeding season abundance data, a frequency distribution was plotted (*Figure 11D.2*). The distribution was quite flat, with two peaks in the 885 – 1,385 and 2,385 – 2,885 abundance categories. The mean peak value was much larger than both the mean and the median abundance of guillemots across both breeding seasons (*Table 11D.5*), giving an indication of the likely level of precaution provided by the use of this metric in the assessment.

Figure 11D.2 Frequency distribution of guillemot abundance data in the Development Area and two kilometre buffer in the breeding season. The mean peak value is shown as a dark green line in the appropriate abundance bin.

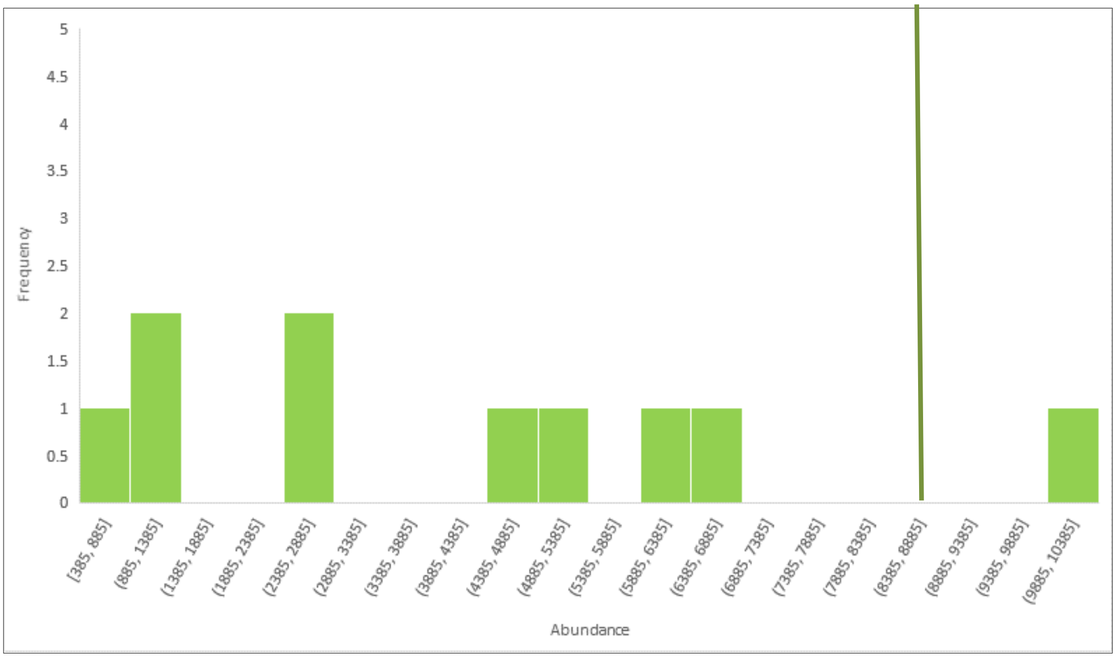


Table 11D.5 Summary statistics of guillemot abundance in the breeding season.

Period	Mean	SD
First breeding season	3726	3808
Second breeding season	4314	2266
All breeding seasons mean	4020	2970
All breeding seasons median	3579	n/a
Mean peak	8184	2476
Counts less than mean peak	9	
Total counts	10	
% less than mean peak	90.0	

20 The non-breeding season mean peak estimate was 3,912 birds (*Table 11D.6*)., which is the value used in the displacement analysis by the matrix approach.

21 The frequency distribution of guillemot non-breeding season abundance data is shown in *Figure 11D.3*. Peak abundances occurred in the lower three categories, between 492 and 1,992 birds. The mean peak value was much larger than both the mean and the median abundance of guillemots across both non-breeding seasons (*Table 11D.6*), giving an indication of the likely level of precaution provided by the use of this metric in the assessment.

Figure 11D.3 Frequency distribution of guillemot abundance data in the Development Area and two kilometre buffer in the non-breeding season. The mean peak value is shown as a dark amber line in the appropriate abundance bin.

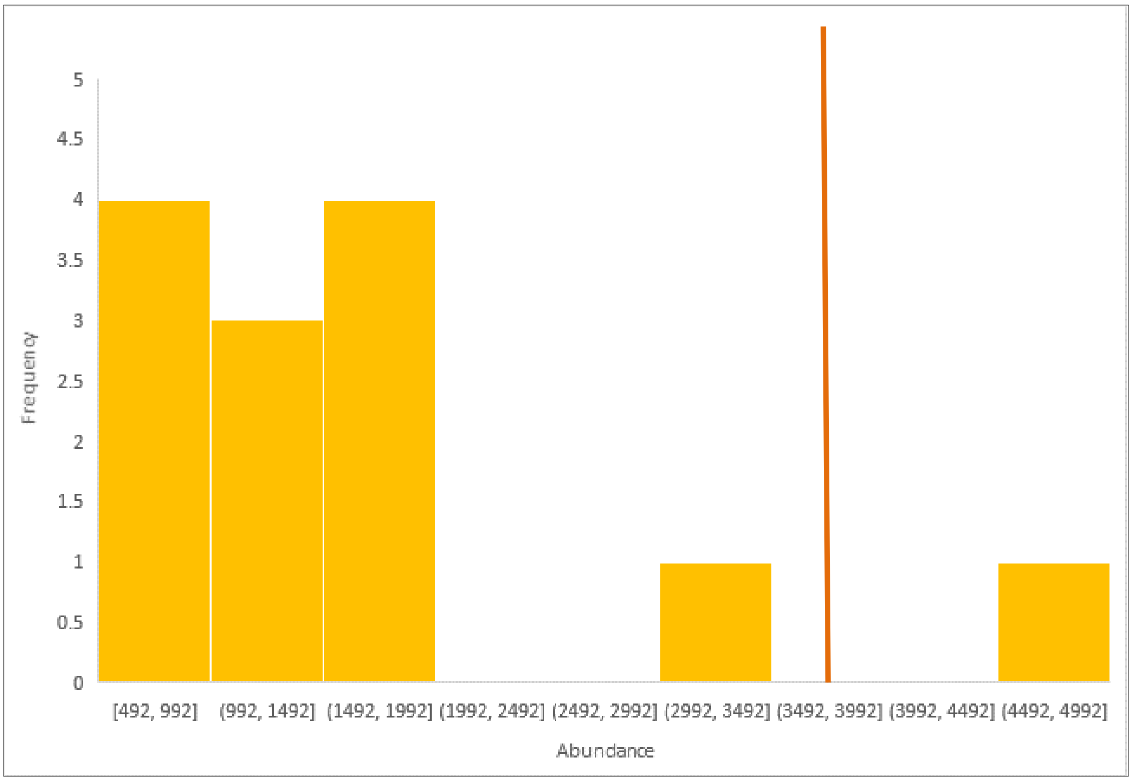


Table 11D.6 Summary statistics of guillemot abundance in the non-breeding season.

Period	Mean	SD
First non-breeding season	1659	878
Second non-breeding season	1762	1358
All non-breeding seasons mean	1715	1116
All non-breeding seasons median	1390	n/a
Mean peak	3912	1018
Counts less than mean peak	12	
Total counts	13	
% less than mean peak	92.3	

Razorbill

22 Razorbill peak abundance in the Development Area and two kilometre buffer occurred in the second non-breeding season (*Table 11D.7*). However, the peak abundance in the first non-breeding season was lower than in both breeding seasons that were sampled. Mean and peak abundance was higher in the second breeding season (2012) than the first (2011). While the difference in the mean abundance was relatively large (almost double in the second breeding season), the peak abundances were similar. Peak abundance occurred in July in both breeding seasons.

23 In the non-breeding season, the mean and peak abundances were higher in the second season of study (2011/12) than in the first (2010/11).

24 It should be noted that the final survey occurred in September 2012 and was the only sample in the third non-breeding season. Since this was the only sample in this season it is excluded it from the analysis, although its inclusion would not change any of the assessments or conclusions.

Table 11D.7 Estimated abundance of razorbill in each month of survey. Seasons are colour coded (green = breeding season, amber = non-breeding). Seasonal peak abundances shown in bold.

Survey number	Month[1]	Year	Development Area	Buffer (2km)	Total
1	Sept	10	321	489	810
2	Oct	10	**1145**	**1673**	**2818**
3	Dec	10	198	375	573
4	Jan	11	528	133	661
5	Feb	11	44	127	171
6	Mar	11	762	471	1233
7	Apr	11	110	106	216
8	May	11	198	161	359
9	Jun	11	367	247	614
10	Jul	11	**2686**	**1916**	**4602**
11	Aug	11	301	206	507
12	Sept	11	**3163**	**3830**	**6993**
13	Oct	11	438	668	1106
14	Nov	11	381	215	596
15	Dec	11	103	267	370
16	Jan	12	118	156	274

Survey number	Month[1]	Year	Development Area	Buffer (2km)	Total
17	Feb	12	73	149	222
18	Mar	12	96	198	294
19	Apr	12	293	492	785
20	May	12	345	293	638
21	Jun	12	205	152	357
22	Jul	12	**2495**	**2245**	**4740**
23	Aug	12	2053	1869	3922
24	Sept	12	1813	1121	2934

[1]The razorbill breeding period was assumed to be April to mid-August (as advised by the Scoping Opinion). Both August surveys were included within the breeding period as they occurred in the first half of the month.

25 The breeding season mean peak value was 4,671 birds (*Table 11D.8*), which is the value used in the displacement analysis by the matrix approach.

26 In order to better understand the distribution of razorbill breeding season abundance data, a frequency distribution was plotted (*Figure 11D.4*). There was a clear peak in the lowest abundance category (216 – 716 birds). The mean peak value was much larger than both the mean and the median abundance of razorbills across both breeding seasons (*Table 11D.8*), giving an indication of the likely level of precaution provided by using this metric in the assessment.

Figure 11D.4 Frequency distribution of razorbill abundance data in the Development Area and two kilometre buffer in the breeding season. The mean peak value is shown as a dark green line in the appropriate abundance bin.

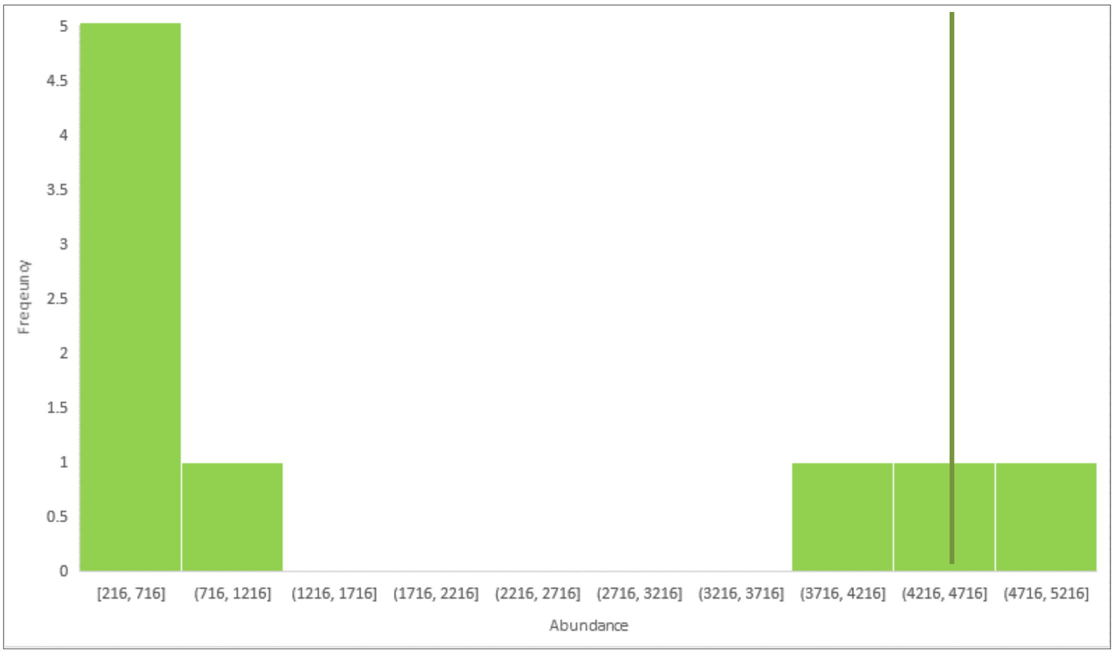


Table 11D.8 Summary statistics of razorbill abundance in the breeding season.

Period	Mean	SD
First breeding season	1260	1874
Second breeding season	2088	2073
All breeding seasons mean	1674	1914
All breeding seasons median	626	n/a
Mean peak	4671	98
Counts less than mean peak	9	
Total counts	10	
% less than PM	90.0	

27 The non-breeding season mean peak value was 4,905 birds (*Table 11D.9*), which is the value used in the displacement analysis by the matrix approach.

28 The frequency distribution of razorbill non-breeding season abundance data is shown in *Figure 11D.5*. Peak abundance occurred in the lowest category, between 171 and 671 birds. The mean peak value was much larger than both the mean and the median abundance of razorbills across both non-breeding seasons (*Table 11D.9*), giving an indication of the likely level of precaution provided by the use of this metric in the assessment.

Figure 11D.5 Frequency distribution of razorbill abundance data in the Development Area and two kilometre buffer in the non-breeding season. The mean peak value is shown as a dark amber line in the appropriate abundance bin.

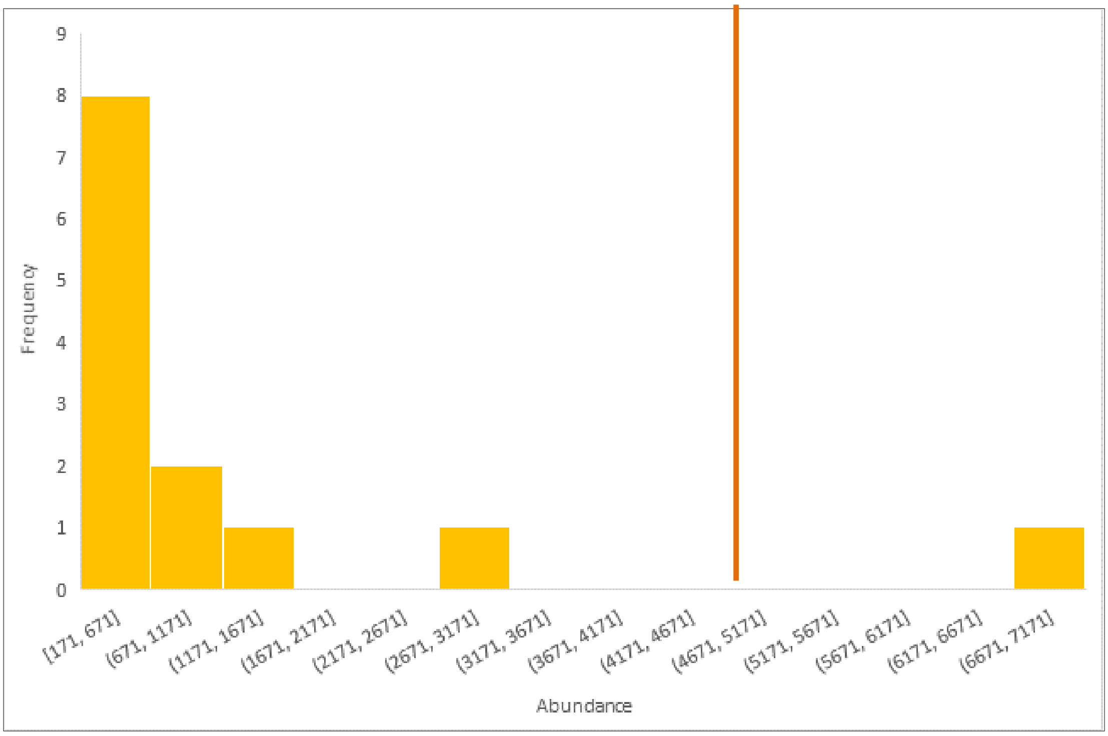


Table 11D.9 Summary statistics of razorbill abundance in the non-breeding season.

Period	Mean	SD
First non-breeding season	1044	935
Second non-breeding season	1408	2482
All non-breeding seasons mean	1240	1865
All non-breeding seasons median	596	n/a
Mean peak	4905	2952
Counts less than mean peak	12	
Total counts	13	
% less than PM	92.3	

__Puffin__

29 Puffins were more abundant in the Development Area and two kilometre buffer during the breeding season than the non-breeding season (*Table 11D.10*). Mean and peak abundances were higher in the second breeding season (2012) than the first (2011). Peak abundance occurred in May in the first breeding season, and in August in the second. The Scoping

Opinion advised that no assessment of displacement and barrier effects was required for puffin in the non-breeding period.

Table 11D.10 Estimated abundance of puffin in each month of survey. Seasons are colour coded (green = breeding season, amber = non-breeding). Peak abundance for each breeding period shown in bold.

Survey number	Month[1]	Year	Development Area	Buffer (2km)	Total
1	Sep	10	138	324	462
2	Oct	10	42	144	186
3	Dec	10	21	30	51
4	Jan	11	0	0	0
5	Feb	11	0	0	0
6	Mar	11	295	108	403
7	Apr	11	155	381	536
8	May	11	**2050**	**1393**	**3443**
9	Jun	11	1224	1063	2287
10	Jul	11	1364	1076	2440
11	Aug	11	1196	769	1965
12	Sept	11	850	1260	2110
13	Oct	11	1939	1326	3265
14	Nov	11	548	315	863
15	Dec	11	169	236	405
16	Jan	12	147	59	206
17	Feb	12	21	20	41
18	Mar	12	274	115	389
19	Apr	12	407	747	1154
20	May	12	2804	1637	4441
21	Jun	12	1358	948	2306
22	Jul	12	1217	960	2177
23	Aug	12	**4152**	**3760**	**7912**
24	Sep	12	1749	1221	2970

[1]The puffin breeding period was assumed to be April to mid-August (as advised by the Scoping Opinion). Both August surveys were included within the breeding period as they occurred in the first half of the month.

30 The breeding period mean peak estimate was 5,678 birds (*Table 11D.11*), which is the value used in the displacement analysis by the matrix method.

31 In order to better understand the distribution of puffin breeding season abundance data, a frequency distribution was plotted (*Figure 11D.6*). There was a clear peak in the abundance category encompassing the range from 2,036 to 2,536 birds. The mean peak value was larger than both the mean and the median abundance of puffins across both breeding season (*Table 11D.11*), giving an indication of the likely level of precaution provided by the use of this metric in the assessment.

Figure 11D.6 Frequency distribution of puffin abundance data in the Development Area and two kilometre buffer in the breeding season. The mean peak value is shown as a dark green line in the appropriate abundance bin.

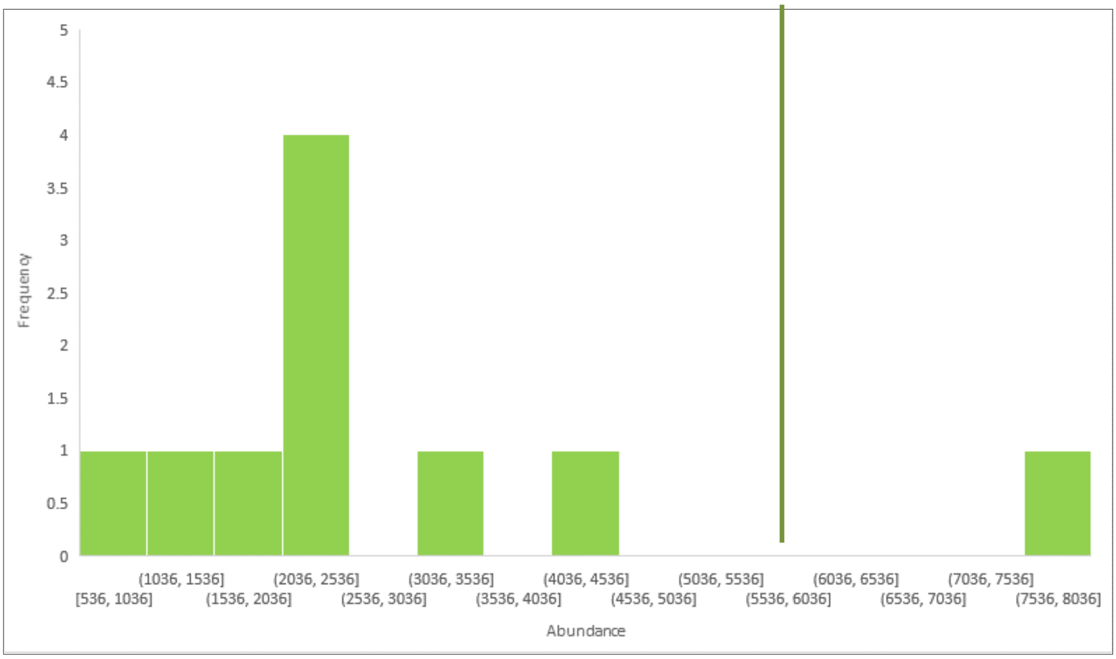


Table 11D.11 Summary statistics of puffin abundance in the breeding season.

	Mean	SD
First breeding season	2134	1050
Second breeding season	3598	2692
All breeding seasons mean	2866	2075
All breeding seasons median	2296	n/a
Mean peak	5678	3160
Counts less than mean peak	9	
Total counts	10	
% less than PM	90.0%	

11D.2.3 Estimated impacts of displacement and barrier effects – Development-alone

32 For each species in each seasonal period requiring assessment, a matrix of displacement against mortality was calculated. These are presented in *Annex 11D.2* and show the total possible range from zero per cent mortality and zero per cent displacement, to 100 per cent mortality and 100 per cent displacement. In each matrix, the recommended displacement and mortality rate is highlighted in green, and interaction between these two is highlighted in dark green. In addition, the estimates obtained by applying a displacement rate 10 per cent lower or higher than that advised in the Scoping Opinion, and the mortality rate one per cent lower or higher than that advised in the Scoping Opinion are shown for each. These are provided to highlight the extent to which estimated mortality varies in relation to variation about the recommended displacement and mortality rates.

33 The seasonal mean peak abundance estimates for each species in each season for which impacts from displacement and barrier effects are to be quantitatively assessed are summarised in *Table 11D.12*. The standard deviation (SD) about the mean value is also shown in each case.

34 The estimated mortalities obtained by applying the recommended rates of displacement and of mortality amongst displaced birds to these abundance estimates are presented in *Table 11D.13*. These are presented in terms of the total estimated mortality of birds in each year, and as the numbers of breeding adults and sub-adults estimated to die. As advised in the Scoping Opinion, the mortality is apportioned to the adult and sub-adult age classes according to the at-sea survey data for kittiwake and the stable age structure derived from the relevant population models for guillemot, razorbill and puffin (*Table 11D.13*, see also *Appendices 11A* and *11E*). The estimated mortality amongst the adult age class is also amended to account for the presence of sabbatical birds, which are assumed to comprise 10 per cent of the adult kittiwakes and seven per cent of the adults of the three auk species.

Table 11D.12 Seasonal mean peak abundance estimates for each species (on the sea and in flight) within the Development Area and two kilometre buffer.

Species	Season	Mean peak (number of individuals)	SD
Kittiwake	Breeding season	3866	1666
Guillemot	Breeding season	8184	2476
	Non-breeding season	3912	1018
Razorbill	Breeding season	4671	98
	Non-breeding season	4905	2952
Puffin	Breeding season	5678	3160

Table 11D.13 The predicted annual mortality from displacement and barrier effects for the Development-alone, by total numbers and as apportioned to age classes.

Species	Season	Mean peak abundance	Number of birds displaced[1]	Total mortality (number of deaths)[1]	Mortality of adults[2]	Mortality of sub-adults[2]
Kittiwake	Breeding	3866	1160	23	19	2
Guillemot	Breeding	8184	4910	49	20	28
	Non-breeding	3912	2347	23	10	13
Razorbill	Breeding	4671	2803	28	13	14
	Non-breeding	4905	2943	29	13	15
Puffin	Breeding	5678	3407	68	24	42

[1]Calculated using rates of displacement and mortality of displaced birds, as recommended in the Scoping Opinion (*Table 11D.1*).
[2]Apportioning of mortality to age classes is based upon the at-sea survey data for kittiwake (giving 93 per cent adults during the breeding period), and the stable age structures of population models for guillemot (43.8 per cent adults), razorbill (49.0 per cent adults) and puffin (38.1 per cent adults). Adult mortality is also reduced by 10 per cent for kittiwake and seven per cent for the three auk species to take account of sabbatical birds (with these percentages as advised in the Scoping Opinion).

35 For kittiwake, the Scoping Opinion advised that only a qualitative assessment of displacement was required for the non-breeding period, and this has been provided in relation to each of the SPA populations with connectivity to the Development Area within the Inch Cape Wind Farm and Offshore Transmission Works Habitats Regulation Appraisal (ICOL, 2018).

11D.2.4 Estimated impacts of displacement and barrier effects - cumulative

36 A quantitative cumulative assessment was carried out for the Development together with the other three Forth and Tay wind farms, as advised in the Scoping Opinion. This was for the breeding period for all four species and also the non-breeding period for guillemot and razorbill. For the other three Forth and Tay wind farms, the additional mortality resulting from displacement and barrier effects was calculated by the SNCB matrix approach using mean peak seasonal abundance estimates for each of the four species provided by the respective developers (*Table 11D.14*).

37 The mean peak abundance estimates for Neart na Gaoithe are derived from their development area and an associated two kilometre buffer, as for the Development. However, the at-sea baseline surveys for the two Seagreen sites encompassed the development areas only, and did not include surrounding buffers. Therefore, the peak seasonal abundances for the two Seagreen sites were adjusted by extrapolating the densities for each site across an assumed two kilometre buffer. The Seagreen sites are contiguous along their longest boundary, so that these assumed buffers did not extend out along the boundary between the two sites. Thus, if the two Seagreen sites are considered

together as a single site, this assumed buffer encompasses the entire site but, when considered as separate sites, each is partially buffered to avoid including areas for which the bird abundance is already incorporated into the estimate for the neighbouring site. Thus, the mean peak abundance estimates for Seagreen Alpha were adjusted on the basis of a calculated area of 197.2 kilometres squared for the site and 300.2 kilometres squared for the site plus partial buffer, whilst the estimates for Seagreen Bravo were adjusted on the basis of a calculated area of 193.7 kilometres squared for the site and 295.1 kilometres squared for the site plus partial buffer.

Table 11.14 Seasonal mean peak abundance estimates for each species (on the sea and in flight) within the Neart na Gaoithe, Seagreen Alpha and Seagreen Beta sites and two kilometre buffers.

Species	Project	Season	Mean peak (number of individuals)[1]	SD[2]
Kittiwake	Neart na Gaoithe	Breeding season	2164	1816
	Seagreen Alpha		2220	-
	Seagreen Bravo		2707	-
Guillemot	Neart na Gaoithe	Breeding season	3263	2028
	Seagreen Alpha		12190	-
	Seagreen Bravo		10778	-
	Neart na Gaoithe	Non-breeding season	7618	3342
	Seagreen Alpha		6131	-
	Seagreen Bravo		6780	-
Razorbill	Neart na Gaoithe	Breeding season	1248	582
	Seagreen Alpha		2768	-
	Seagreen Bravo		993	-
	Neart na Gaoithe	Non-breeding season	3101	1491
	Seagreen Alpha		1253	-
	Seagreen Bravo		1723	-
Puffin	Neart na Gaoithe	Breeding season	6173	2365
	Seagreen Alpha		3704	-
	Seagreen Bravo		5340	-

[1]Based on abundance estimates provided by the respective developers but with the Seagreen values amended to account for an assumed two kilometre buffer (see text for further explanation).
[2]No SD about the mean is provided for the Seagreen sites because of the extrapolation involved in calculating the mean peak abundance estimates.

38 The estimated mortalities obtained by applying the recommended rates of displacement and of mortality amongst displaced birds to the abundance estimates for each of the Forth and Tay wind farms are presented in *Table 11D.15*. These are presented in terms of the total

number of birds estimated to die in each year, and as the numbers of breeding adults and sub-adults estimated to die. As for the Development-alone (*Table 11D.13*), the mortality is apportioned to the adult and sub-adult age classes according to the at-sea survey data for kittiwake and the stable age structure derived from the relevant population models for guillemot, razorbill and puffin (*Table 11D.15*, see also *Appendices 11A* and *11E*). The estimated mortality amongst the adult age class is also amended to account for the presence of sabbatical birds, which are assumed to comprise 10 per cent of the adult kittiwakes and seven per cent of the adults of the three auk species.

Table 11D.15 The predicted annual mortality from displacement and barrier effects for the Inch Cape Wind Farm cumulatively with the other three Forth and Tay wind farms, by total numbers and as apportioned to age classes.

Species	Project	Season	Total mortality (number of deaths)[1]	Mortality of adults[2]	Mortality of sub-adults[2]
Kittiwake	Inch Cape	Breeding	23	19	2
	Neart na Gaoithe		13	11	1
	Seagreen Alpha		13	11	1
	Seagreen Bravo		16	14	1
	Cumulative impact[3]		**66**	**55**	**4**
Guillemot	Inch Cape	Breeding	49	20	28
	Neart na Gaoithe		20	8	11
	Seagreen Alpha		73	30	41
	Seagreen Bravo		65	26	36
	Cumulative impact[3]		**206**	**84**	**116**
	Inch Cape	Non-breeding	23	10	13
	Neart na Gaoithe		46	19	26
	Seagreen Alpha		37	15	21
	Seagreen Bravo		41	17	23
	Cumulative impact[3]		**147**	**60**	**83**
Razorbill	Inch Cape	Breeding	28	13	14
	Neart na Gaoithe		7	3	4
	Seagreen Alpha		17	8	8
	Seagreen Bravo		6	3	3
	Cumulative impact[3]		**58**	**26**	**30**
	Inch Cape	Non-	29	13	15

Species	Project	Season	Total mortality (number of deaths)[1]	Mortality of adults[2]	Mortality of sub-adults[2]
	Neart na Gaoithe	breeding	19	8	9
	Seagreen Alpha		8	3	4
	Seagreen Bravo		10	5	5
	Cumulative impact[3]		**66**	**30**	**34**
Puffin	Inch Cape	Breeding	68	24	42
	Neart na Gaoithe		74	26	46
	Seagreen Alpha		44	16	28
	Seagreen Bravo		64	23	40
	Cumulative impact[3]		**251**	**89**	**155**

[1]Calculated using rates of displacement and mortality of displaced birds, as recommended in the Scoping Opinion (*Table 11D.1*).
[2]Apportioning of mortality to age classes is based upon the at-sea survey data for kittiwake (giving 93 per cent adults during the breeding period at each site, except Seagreen Bravo where 95 per cent were estimated to be adults), and the stable age structures of population models for guillemot (43.8 per cent adults), razorbill (49.0 per cent adults) and puffin (38.1 per cent adults). Adult mortality is also reduced by 10 per cent for kittiwake and seven per cent for the three auk species to take account of sabbatical birds (with these percentages as advised in the Scoping Opinion).
[3]Totals for each site are rounded to the nearest integer and so may differ from the cumulative totals.

11D.3 Comparisons with estimates of displacement and barrier effects from individual-based modelling approaches

11D.3.1 Individual-based Modelling Approaches

39 The Scoping Opinion (and subsequent clarifications – Chapter 11, Table 11.1[3]) advised that estimates of displacement and barrier effects as generated by individual-based modelling approaches should be used to provide context to the estimates produced by the SNCB matrix approach. In this regard, the Scoping Opinion specifically identified the existing estimates from the work of Searle *et al.* (2014), undertaken in relation to the Forth and Tay wind farms, and the SeabORD model, which is a prototype tool that was in the course of being developed at the time the Scoping Opinion was published. To inform this element of the assessment, the Centre for Ecology and Hydrology (CEH) were commissioned by ICOL to run the SeabORD model in relation to the Wind Farm alone and in-combination with the other three Forth and Tay wind farms. This was undertaken before publication of the SeabORD model, and at the time of writing the model remains unpublished.

40 These individual-based modelling approaches simulate the behaviour and energetics of individual birds from breeding seabird populations under baseline conditions (i.e. with no

[3] At the time of writing, the correspondence relating the clarifications of the Scoping Opinion is available at: http://www.gov.scot/Topics/marine/Licensing/marine/scoping/ICOLRevised-2017/OrnithologyQ-092017 [Accessed 15/05/18]

wind farm present) and compare the resulting demographic estimates to model runs undertaken in scenarios which have the wind farm(s) of interest present (so that birds undertaking foraging trips from the colony have the potential to incur energetic costs from barrier effects and of increased intra-specific competition for food if they are displaced). In both the SeabORD and Searle *et al.* (2014) models, these effects are estimated in terms of changes to adult and chick mortality, with the available outputs relating to the individual SPA populations that are of interest to the assessment. The estimated mortality to adult birds relates only to the breeding period.

41 Both the SeabORD and Searle *et al.* (2014) models rely upon predictions of the distribution of seabird prey resources and of foraging birds. Both of these aspects are determined by the availability of Global Position System (GPS) tracking data from breeding birds associated with the colonies of interest. For the Searle *et al.* (2014) model, these predictions derived from relatively small numbers of tracked birds (i.e. 33 to 53 for five of the SPA populations relevant to the current assessment, and fewer than 19 for the remaining five SPA populations of relevance, including one population with no tracking data available). Consequently, the Searle *et al.* (2014) model considered scenarios of both heterogeneous prey distribution (as determined using the GPS tracking data) and uniform (or homogeneous) prey distribution across the entire Forth and Tay region for each SPA population that was eventually modelled, so giving two estimates of effects for each of these populations.

42 Further GPS tracking data have become available for some of the SPA populations of interest since the Searle *et al.* (2014) work, so that the SeabORD modelling is based upon a larger sample of such data (e.g. Wakefield *et al.*, 2017). Nevertheless, uniform prey distributions have had to be assumed by SeabORD for both puffin and razorbill due to the fact that GPS tracking data for these species are available from one SPA population only.

43 As well as the predictions of the distribution of prey resources and of foraging birds, both modelling approaches are underpinned by a range of other assumptions and predictions (e.g. on the relationships between adult body mass and survival), each of which have associated uncertainties. Full details of the Searle *et al.* (2014) modelling approach can be obtained from the published report to Marine Scotland Science, but at the time of writing full details of the SeabORD modelling approach are not yet available.

11D.3.2 Estimating Displacement and Barrier Effects Using SeabORD

44 Details of the SeabORD modelling undertaken to inform the current assessment are provided in *Annex 11D.3*. In summary, the modelling was based upon 10 matched paired model runs (i.e. with and without the wind farm(s) of interest present) for each SPA population and wind farm scenario (i.e. Development-alone or in-combination), with the percentage of birds within each population assumed to be susceptible to displacement being equivalent to the species-specific displacement rates advised by the Scoping Opinion (i.e. 30 per cent for kittiwake and 60 per cent for the three auk species). All ‘displacement susceptible’ birds were also assumed susceptible to barrier effects. As advised in the clarifications to the Scoping Opinion, a two kilometre buffer was assumed for the

Development Area and each of the other three Forth and Tay wind farms for the purposes of this modelling[4], whilst birds displaced from foraging within the wind farms (and associated buffers) were assumed to re-locate to forage in areas that were within five kilometres of the wind farm from which they had been displaced. All modelling used the "perimeter" method to calculate flight paths around the wind farms (there being two options for this calculation within the SeabORD model) and had the seed number set at 19873.

45 Except for the guillemot SPA populations, the modelling was undertaken on the basis of sampling 100 per cent of the simulated population. For guillemot, modelling was based upon sampling 50 per cent of the Forth Islands SPA population and 10 per cent of the other three SPA populations with connectivity to the Development Area and two kilometre buffer (see above). The presence of birds from a range of the colonies with connectivity to the Development Area and two kilometre buffer (including non-SPA colonies) was incorporated into the modelling, although estimates of effects were generated only for the SPA populations of interest.

46 At the time of commissioning the SeabORD modelling used to inform the current assessment, MS LOT were unable to provide advice on the number and range of prey levels that should be encompassed by the modelling, and advised that this should be determined via discussion with CEH[5]. Following discussion between ICOL and CEH, moderate prey levels were assumed for each SPA population (with values as in *Annex 11D.4*).

47 The effects on adult and chick mortality predicted by SeabORD are expressed as percentage point changes (i.e. the number of deaths expressed as a percentage of the source population size). For the Development-alone, the effects on adult mortality range from a decrease of 0.005 per cent for the St Abb's Head to Fast Castle SPA guillemot population to an increase of 0.59 per cent for the Forth Islands SPA puffin population, whilst for chick mortality they range from a decrease of 0.003 per cent for the St Abb's Head to Fast Castle SPA kittiwake population to an increase of 1.1 per cent for the Forth Islands SPA kittiwake population (*Table 11D.16*). In terms of the predicted in-combination effects, for adult mortality these range from a decrease of 0.004 per cent for the Buchan Ness to Collieston Coast SPA guillemot population to an increase of 1.63 per cent for the Forth Islands SPA puffin population, whilst for chick mortality they range from a decrease of 0.02 per cent for the St Abb's Head to Fast Castle SPA guillemot population to an increase of 5.89 per cent for the Forth Islands SPA kittiwake population (*Table 11D.17*).

48 For all species, the predicted effects on both adult and chick mortality are invariably greatest for the Forth Islands SPA populations, with the effects on other SPA populations often orders of magnitude lower. This applies to both the Development-alone and in-combination scenarios (*Tables 11D.16* and *11D.17*). Such differences might be expected in relation to the St Abb's Head to Fast Castle SPA and the Buchan Ness to Collieston Coast SPA due to the considerably greater distances of these two SPAs from the Development Area and the other Forth and Tay wind farms. However, such marked differences in the magnitude of the

[4] Letter of 29 September 2017 from MS LOT to ICOL.
[5] Letter of 3 November 2017 from MS LOT to ICOL.

predicted effects between the Forth Islands SPA populations and the Fowlsheugh SPA populations are more surprising, given that the Fowlsheugh SPA is a similar distance from the Development Area (at 49 kilometres, compared to 40 kilometres for the Forth Islands SPA[6]) and is closer to both of the Seagreen wind farm sites (at 46 kilometres, compared to 68 kilometres for the Forth Islands SPA[6] above, respectively), albeit that it is considerably further from the Neart na Gaoithe wind farm site (at 73 kilometres, compared to 21 kilometres for the Forth Islands SPA[6]).

49 The estimates of uncertainty associated with the predicted effects on adult and chick mortality are expressed as 95 per cent prediction intervals (*Annex 11D.3*). These invariably encompass a wide range of values, with the lower and upper interval values differing from the mean by more than 100 per cent of the mean for 33 of the 40 estimates and with the intervals spanning zero for 30 of the 40 estimates (*Tables 11D.16* and *11D.17*). The calculated 95 per cent intervals only account for some of the known sources of variability and uncertainty in estimating the effects on adult and chick mortality, with several known and potentially important sources of uncertainty unaccounted for (e.g. uncertainty in the adult body mass and survival relationship – see *Annex 11D.3*).

Table 11D.16 The predicted percentage point changes in adult and chick mortality as a result of displacement and barrier effects for different SPA populations from the Development-alone, as estimated by the SeabORD and Searle *et al.* (2014) models

SPA	Species	SeabORD		Searle *et al.* (2014)[1]		
		Adult mortality (95 % prediction interval)	Chick mortality (95 % prediction interval)	Prey distribution[2]	Adult mortality	Chick mortality
Forth Islands	Kittiwake	0.20 (0.03 – 0.37)	1.10 (-0.54 – 2.73)	Uniform	0.31	-0.44
				Heterogeneous	0.47	-0.14
	Guillemot	0.22 (0.08 – 0.36)	0.62 (-0.71 – 1.94)	Uniform	-	-
				Heterogeneous	-	-
	Razorbill	0.24 (-0.04 – 0.52	0.65 (-0.60 – 1.90)	Uniform	0.09	0.07
				Heterogeneous	0.11	-0.17
	Puffin	0.59 (0.24 – 0.94)	0.39 (-0.77 – 1.56)	Uniform	1.44	1.73
				Heterogeneous	0.13	-0.31
Fowlsheugh	Kittiwake	0.005 (-0.01 – 0.02)	0.08 (-0.37 – 0.53)	Uniform	0.15	-0.06
				Heterogeneous	0.21	0.03

[6]Taken as the distance to the Isle of May, which holds the largest numbers of kittiwake, guillemot and razorbill amongst the different colonies that comprise the Forth Islands SPA.

SPA	Species	SeabORD		Searle *et al.* (2014)[1]		
		Adult mortality (95 % prediction interval)	Chick mortality (95 % prediction interval)	Prey distribution[2]	Adult mortality	Chick mortality
	Guillemot	0.007 (-0.07 – 0.08)	0.05 (-0.07 – 0.16)	Uniform	-	-
				Heterogeneous	-	-
	Razorbill	0.14 (-0.03 – 0.32)	0.40 (-1.25 – 2.04)	Uniform	-	-
				Heterogeneous	-	-
St Abb's Head to Fast Castle	Kittiwake	0.002 (-0.02 – 0.02)	-0.003 (-0.04 – 0.04)	Uniform	0.00	-13.57
				Heterogeneous	-0.03	-0.94
	Guillemot	-0.005 (-0.02 – 0.01)	0.00 (-0.04 – 0.04)	Uniform	-	-
				Heterogeneous	-	-
Buchan Ness to Collieston Coast	Guillemot	0.00 (0.00 – 0.00)	0.00 (0.00 – 0.00)	Uniform	-	-
				Heterogeneous	-	-

[1]Outputs from Searle *et al.* (2014) were not available for some SPA populations either because of a lack of interaction of the simulated SPA birds with the wind farm (i.e. guillemots from both St Abb's Head to Fast Castle SPA and Buchan Ness to Collieston Coast SPA and razorbills from Fowlsheugh SPA) or else the effects could not be estimated reliably.
[2]Modelling in Searle *et al.* (2014) was undertaken for both uniform and heterogeneous prey distributions for each SPA population, whereas SeabORD used only heterogeneous prey distributions for kittiwake and guillemot and only uniform prey distributions for razorbill and puffin (due to insufficient tracking data for the latter two species – see *11D.3.1*).

Table 11D.17 The predicted percentage point changes in adult and chick mortality as a result of displacement and barrier effects for different SPA populations from the Development in-combination with the other three Forth and Tay wind farms, as estimated by the SeabORD and Searle et al. (2014) models

SPA	Species	SeabORD		Searle et al. (2014)1		
		Adult mortality (95 % prediction interval)	Chick mortality (95 % prediction interval)	Prey distribution2	Adult mortality	Chick mortality
Forth Islands	Kittiwake	0.84 (-0.19 – 1.87)	5.89 (-0.19 – 11.97)	Uniform	1.97	2.14
				Heterogeneous	1.82	1.18
	Guillemot	1.42 (0.27 – 2.57)	5.62 (-4.31 – 15.55)	Uniform	-	-
				Heterogeneous	-	-
	Razorbill	0.59 (0.22 –	1.87 (-1.76 –	Uniform	0.82	-1.99

SPA	Species	SeabORD		Searle et al. (2014)1		
		Adult mortality (95 % prediction interval)	Chick mortality (95 % prediction interval)	Prey distribution2	Adult mortality	Chick mortality
		0.96)	5.49)	Heterogeneous	0.24	2.93
	Puffin	1.63 (0.74 – 2.51)	1.34 (-2.48 – 5.15)	Uniform	3.32	4.87
				Heterogeneous	-0.04	1.56
Fowlsheugh	Kittiwake	0.10 (0.02 – 0.18)	0.49 (-0.63 – 1.61)	Uniform	0.48	1.67
				Heterogeneous	0.44	-
	Guillemot	0.14 (-0.005 – 0.29)	1.01 (-0.66 – 2.67)	Uniform	-	-
				Heterogeneous	-	-
	Razorbill	0.27 (-0.03 – 0.57)	0.52 (-1.23 – 2.27)	Uniform	-	-
				Heterogeneous	-	-
St Abb's Head to Fast Castle	Kittiwake	0.04 (-0.05 – 0.12)	0.14 (-0.10 – 0.38)	Uniform	0.18	-
				Heterogeneous	0.22	-
	Guillemot	0.02 (-0.08 – 0.12)	0.02 (-0.12 – 0.17)	Uniform	-	-
				Heterogeneous	-	-
Buchan Ness to Collieston Coast	Guillemot	-0.004 (-0.04 – 0.03)	0.09 (-0.12 – 0.29)	Uniform	-	-
				Heterogeneous	-	-

[1]Outputs from Searle et al. (2014) were not available for some SPA populations either because of a lack of interaction of the simulated SPA birds with the wind farm (i.e. guillemots from both St Abb's Head to Fast Castle SPA and Buchan Ness to Collieston Coast SPA and razorbills from Fowlsheugh SPA) or else the effects could not be estimated reliably.

[2]Modelling in Searle et al. (2014) was undertaken for both uniform and heterogeneous prey distributions for each SPA population, whereas SeabORD used only heterogeneous prey distributions for kittiwake and guillemot and only uniform prey distributions for razorbill and puffin (due to insufficient tracking data for the latter two species – see *11D.3.1*).

11D.3.3 Comparisons of the Predicted Effects from SeabORD and Searle *et al.* (2014)

50 Predictions of adult and chick mortality resulting from displacement and barrier effects as calculated by the earlier Searle *et al.* (2014) model are also presented in *Tables 11D.16* and *11D.17* for comparison with the SeabORD outputs. These predicted effects are available for a subset of the SPA populations of interest only. This is because there was a lack of interaction of the simulated SPA birds with the Forth and Tay wind farms in some instances (leading to no predicted effects), whilst in others the effects could not be estimated reliably. The Searle *et al.* (2014) modelling also assumed a one kilometre buffer for the Development Area and each of the other Forth and Tay wind farms (in contrast to the two kilometre buffer used in the current SeabORD modelling), whilst for kittiwake it was assumed that 40 per cent

of birds were susceptible to displacement and barrier effects (as opposed to 30 per cent for the current SeabORD modelling).

51 For most SPA populations, the adult mortality effects predicted by the Searle *et al.* (2014) model using the different assumed prey distributions tended to show reasonable agreement. However, the Forth Islands SPA puffin population was a notable exception in this regard, with the effects for the homogenous prey distribution being orders of magnitude greater than those predicted using the heterogeneous prey distribution for both the Development-alone and in-combination scenarios. The predicted chick mortalities tended to show greater differences according to the assumed prey distribution, with this again relatively marked for the Forth Islands SPA puffin population (*Tables 11D.16* and *11D.17*).

52 The effects predicted by the SeabORD and Searle *et al.* (2014) models show varying degrees of agreement. Thus, approximately only half of the estimated effects from Searle *et al.* (2014) lie within the 95 per cent prediction intervals of the corresponding SeabORD estimate, despite these encompassing a wide range of values. This level of correspondence is irrespective of the prey distribution assumed in the Searle *et al.* (2014) modelling.

53 Across the different SPA populations, the predicted adult mortalities from SeabORD are highly correlated with those from the Searle *et al.* (2014) model when uniform prey distributions are assumed (r = 0.97[7] when considering the Development-alone and in-combination estimates together, although the correlation is apparent for both the Development-alone and in-combination estimates when considered in isolation). This correlation is not reliant solely upon the razorbill and puffin SPA populations modelled using uniform prey distributions by SeabORD (i.e. the estimates for the kittiwake and guillemot SPA populations are also correlated, despite the fact that the SeabORD model uses GPS tracking data to predict prey distributions for those species – *Annex 11D.3*). However, despite this high correlation, the effects predicted by the Searle *et al.* (2014) model are, on average, more than twice as large as those predicted by SeabORD[8].

54 In contrast to the above, there is little evidence of close correlation between the adult mortalities predicted by SeabORD and those predicted by the Searle *et al.* (2014) model when based upon heterogeneous prey distributions (r = 0.14), or between the chick mortalities predicted by SeabORD and those predicted by the Searle *et al.* (2014) model (r = 0.31 and r = 0.49 for the Searle *et al.* (2014) estimates based upon uniform and heterogeneous prey distributions, respectively). The chick mortalities predicted by SeabORD tended to be greater than those predicted by the Searle *et al.* (2014) model, irrespective of the prey distribution assumed by the latter (*Tables 11D.16* and *11D.17*).

55 Overall, it is difficult to discern clear consistencies in the effects predicted by the two individual-based modelling approaches used to examine displacement and barrier effects.

[7] The r-value is the Pearson correlation coefficient, for which values can range from 0 (no correlation) to 1 (perfect correlation). The statistical significance associated with r = 0.97 for a sample size of 10 is $P < 0.001$ (indicating that the likelihood of this level of correlation occurring by chance is less than one in a thousand).
[8]Based upon the regression of the SeabORD estimates against the Searle *et al.* (2014) estimates, which gives the equation y = 0.47x + 0.02.

This is perhaps unsurprising given that some of the assumptions applied to the two approaches differ (i.e. in terms of the buffer distances and, for kittiwakes, the proportion of displacement sensitive birds), as will the underpinning predicted distributions of prey resources and foraging birds for those species where the sample of GPS tracking data has increased in the intervening period between the development of the two models. Furthermore, there are likely to be differences in the structures and functioning of the underlying modelling systems themselves.

56 The fact that the adult mortalities predicted by SeabORD correlate strongly with those derived from the Searle *et al.* (2014) model when based upon uniform, but not heterogeneous, prey distributions seems surprising and could indicate that outputs are particularly sensitive to changes in the underlying GPS tracking data. At the same time, the very marked differences in the predicted effects on the Forth Islands SPA puffin population according to the model used and, for Searle *et al.* (2014), the assumptions concerning prey distribution suggest considerable uncertainty surrounds the predictions for some SPA populations at least.

11D.3.4 Considering the SNCB Matrix Estimates in Relation to the Predicted Effects from SeabORD

57 Direct comparisons of the predicted effects from the SeabORD model with those derived by applying the SNCB matrix approach are limited because of the differences in the outputs produced by each of these approaches. Thus, predictions from the SeabORD model are restricted to the breeding period and concerned with effects on adult and chick mortality, whereas the matrix (as used in the current assessment) has considered both the breeding and (for guillemot and razorbill) non-breeding periods and has estimated effects in terms of the mortality to adult and sub-adult birds (but not to chicks). Therefore, direct comparisons are limited to the predicted adult mortality during the breeding period (*Table 11D.18*).

58 To enable these comparisons to be made for each SPA population, the matrix estimates of adult mortality during the breeding period (as presented in *Table 11D.15* above) were apportioned to the respective SPA populations and expressed as percentage point changes in mortality. This was undertaken by first applying the apportionment estimates presented in *Table 11B.3* of *Appendix 11B: Apportioning Effects to SPA Colonies During the Breeding and Non-breeding Seasons* (and which are calculated for each SPA population in relation to the Inch Cape Wind Farm and each of the other three Forth and Tay wind farms). The number of adult deaths per annum attributed to each SPA population was then expressed as a percentage of the number of individual adult birds estimated in each population (*Table 11D.19*).

59 In terms of predicted adult mortality during the breeding period, the estimates produced by the matrix lie within the 95 per cent prediction intervals of the SeabORD estimates for six and seven of the 10 SPA populations for the Development-alone and in-combination scenarios, respectively (*Table 11D.18*). The estimates from the SeabORD model are invariably greater than those from the matrix for the Forth Islands SPA populations (for both the Development-alone and in-combination scenarios), and for the in-combination scenario for the Fowlsheugh SPA populations. For most other SPA populations, the matrix produced

higher estimates of breeding period adult mortality, although the estimated effects by either method were small in several of these instances. There was limited evidence of correlation between the estimates produced by the matrix and the SeabORD model (r = 0.40 and r = 0.42 for the Development-alone and in-combination estimates, respectively).

Table 11D.18 Predicted adult mortality amongst SPA populations during the breeding period, as estimated by the SNCB Matrix and the SeabORD model for the Development-alone and in-combination with the other three Forth and Tay wind farms

SPA	Species	Percentage point changes in adult mortality for the Development-alone		Percentage point changes in adult mortality for in-combination	
		Matrix estimate	SeabORD estimate (95 % prediction interval)	Matrix estimate	SeabORD estimate (95 % prediction interval)
Forth Islands	Kittiwake	0.044	0.20 (0.03 – 0.37)	0.147	0.84 (-0.19 – 1.87)
	Guillemot	0.018	0.22 (0.08 – 0.36)	0.056	1.42 (0.27 – 2.57)
	Razorbill	0.052	0.24 (-0.04 – 0.52)	0.089	0.59 (0.22 – 0.96)
	Puffin	0.024	0.59 (0.24 – 0.94)	0.086	1.63 (0.74 – 2.51)
Fowlsheugh	Kittiwake	0.029	0.005 (-0.01 – 0.02)	0.082	0.10 (0.02 – 0.18)
	Guillemot	0.010	0.007 (-0.07 – 0.08)	0.054	0.14 (-0.005 – 0.29)
	Razorbill	0.040	0.14 (-0.03 – 0.32)	0.103	0.27 (-0.03 – 0.57)
St Abb's Head to Fast Castle	Kittiwake	0.016	0.002 (-0.02 – 0.02)	0.038	0.04 (-0.05 – 0.12)
	Guillemot	0.006	-0.005 (-0.02 – 0.01)	0.025	0.02 (-0.08 – 0.12)

SPA	Species	Percentage point changes in adult mortality for the Development-alone		Percentage point changes in adult mortality for in-combination	
		Matrix estimate	SeabORD estimate (95 % prediction interval)	Matrix estimate	SeabORD estimate (95 % prediction interval)
Buchan Ness to Collieston Coast	Guillemot	0.001	0.00 (0.00 – 0.00)	0.008	-0.004 (-0.04 – 0.03)

60 The largest differences in breeding period adult mortality between the matrix and SeabORD estimates were associated with the Forth Islands SPA populations, for which three of the four Development-alone and two of the four in-combination SeabORD estimates were an order of magnitude (or more) greater than the corresponding matrix estimate (*Table 11D.18*). These differences were most marked for guillemot and puffin, with the SeabORD estimates for guillemot being 12 and 25 times greater than the matrix estimates for Development-alone and in-combination, respectively, whilst for puffin they were 26 and 19 times greater for Development-alone and in-combination, respectively. As noted above, the effects predicted by the SeabORD model were particularly high for the Forth Islands SPA populations, whilst the effects predicted on the Forth Islands SPA puffin population by the SeabORD and Searle *et al.* (2014) modelling show high variability.

61 The comparisons between the SeabORD and matrix estimates can be used to derive the displacement and mortality rates that are required to match the SeabORD estimates of adult mortality, given the mean peak abundance estimates and associated apportioning used to estimate the number of birds from each SPA population that occur within the Development Area and two kilometre buffer (as well as the development areas and associated buffers for the other three Forth and Tay wind farms). Thus, if the displacement rate advised in the Scoping Opinion is assumed, the mortality rate of displaced guillemots from the Forth Islands SPA would have to be 12 per cent to match the estimates produced by SeabORD for the Development-alone, and 26 per cent to match the in-combination estimates. Similarly, mortality rates of displaced puffins from the Forth Islands SPA would have to be 49 per cent to match the Development-alone estimates from SeabORD, and 38 per cent to match the in-combination estimates from SeabORD (noting that the matrix approach has used a mortality rate amongst displaced birds of one per cent for guillemots and two per cent for puffin, as advised in the Scoping Opinion). Likewise, in these examples, a displacement rate of over 100 per cent would be required to match the SeabORD estimates of mortality if it is assumed that the mortality rate amongst displaced birds is as advised by the Scoping Opinion.

62 The above examples highlight the most extreme differences between the adult mortality estimates produced by the matrix and the SeabORD model. However, for several other SPA populations, the mortality rates amongst displaced birds would have to be three to nine times greater than advised in the Scoping Opinion if the displacement rates advised in the Scoping Opinion are applied, whilst the displacement rates would have to exceed 100 per

cent if the mortality rates amongst displaced birds are as advised in the Scoping Opinion (*Table 11D.18*).

63 These comparisons suggest that the SeabORD estimates of adult mortality during the breeding period may be unrealistically high for some SPA populations. An alternative explanation is that the assumptions on which the matrix approach is based fail to take sufficient account of barrier effects (which may be less dependent on the estimates of the number of birds using the wind farm sites) or the turn-over of SPA birds on the wind farm sites. However, the matrix approach incorporates the estimates of birds in flight within the Development Area and two kilometre buffer, as well as those recorded on the water (so accounting for birds potentially exposed to barrier effects), and uses the mean peak count of birds on the wind farm sites (plus their associated buffers), which (as detailed above) incorporates a relatively high degree of precaution (given the extent to which it exceeds the mean and median estimates of abundance). Therefore, these explanations seem unlikely to be sufficient in themselves to account for high mortality estimates produced by the SeabORD model.

64 As a further sense check on how the SeabORD estimates of adult mortality relate to those produced by the matrix, the percentage of each SPA population that is required to occur on the wind farm sites and their buffers to account for the mortality estimated by SeabORD was calculated. This calculation was based upon applying the rates of displacement and of mortality amongst displaced birds advised by the Scoping Opinion (and used in the matrix approach) to the SeabORD mortality estimates (*Table 11D.19*). The comparison indicates that approximately 30 to 50 per cent of each of the Forth Islands SPA populations have to occur within the Development Area and two kilometre buffer to account for the estimated mortality, whilst these percentages approach or greatly exceed 100 per cent when all of the Forth and Tay wind farms are considered (*Table 11D.19*). A relatively high percentage occurrence of the Fowlsheugh SPA populations is also indicated when all Forth and Tay wind farms are considered.

65 Therefore, the comparison with the SPA population sizes again suggests that the SeabORD model has produced unrealistically high estimates of mortality for some SPA populations, particularly those from the Forth Islands SPA. This conclusion depends upon the assumption that the rates of displacement and of mortality amongst displaced birds advised in the Scoping Opinion are reasonable and are not gross underestimates. Whilst there is a lack of supporting data to confirm this assumption, it remains the case that these rates have been determined on the basis of careful consideration of what is plausible from the biological perspective, with there being broad agreement on these rates by the range of expertise on which the Scoping Opinion relied.

Table 11D.19 The percentage of each SPA population estimated to be using the Development Area and two kilometre buffer (upper rows) and the combined Forth and Tay wind farm sites and buffers (lower rows) on the basis of relating the SeabORD estimates of adult mortality to the rates of displacement and mortality amongst displaced birds that are advised by the Scoping Opinion

SPA	Species	SPA population size (number of individuals)	Estimated adult mortality		Estimated number of birds using the site(s)[1]	Percentage of SPA population estimated to use the site(s) (%)
			Percentage point change	Number of deaths		
Forth Islands	Kittiwake	9,326	0.20	19	3,109	33
			0.84	78	13,056	140
	Guillemot	38,573	0.22	85	14,143	37
			1.42	548	91,289	237
	Razorbill	7,792	0.24	19	3,117	40
			0.59	46	7,662	98
	Puffin	90,010	0.59	531	44,255	49
			1.63	1,467	122,264	136
Fowlsheugh	Kittiwake	19,310	0.005	1	161	1
			0.10	19	3,218	17
	Guillemot	74,379	0.007	5	868	1
			0.14	104	17,355	23
	Razorbill	9,950	0.14	14	2,322	23
			0.27	27	4,478	45
St Abb's Head to Fast Castle	Kittiwake	6,668	0.002	0	22	<1
			0.04	3	445	7
	Guillemot	48,516	-0.005	-2	N/A	N/A

SPA	Species	SPA population size (number of individuals)	Estimated adult mortality		Estimated number of birds using the site(s)[1]	Percentage of SPA population estimated to use the site(s) (%)
			Percentage point change	Number of deaths		
			0.02	10	1,617	3
Buchan Ness to Collieston Coast	Guillemot	45,067	0.00	0	0	0
			-0.004	-2	N/A	N/A

[1]Calculated by multiplying the estimated number of deaths by the breeding period rates of displacement and mortality amongst displaced birds advised for each species by the Scoping Opinion (i.e. 60 per cent displacement for guillemot, razorbill and puffin and 30 per cent displacement for kittiwakes, and one per cent mortality for guillemot and razorbill and two per cent mortality for puffin and kittiwake).

11D.3.5 Conclusions

66 The above comparisons suggest considerable variability in the predicted effects from the individual-based modelling approaches and, potentially, considerable sensitivity in the outputs according to certain assumptions on which the modelling is based (notably in terms of prey distributions). The extrapolations from the adult mortality estimates produced by the SeabORD model suggest that for some populations (particularly from the Forth Islands SPA) unrealistically high rates of displacement and/or of mortality amongst displaced birds are required for these estimates to match the population sizes (as determined by the mean peak counts) recorded on the Development Area and two kilometre buffer, and on the other Forth and Tay wind farm sites. Similarly, extrapolations based on the advised rates of displacement and of mortality amongst displaced birds suggest that the use of the Development Area and two kilometre buffer and the other Forth and Tay wind farms would have to be unrealistically high amongst some SPA populations to match the adult mortality predicted by SeabORD.

67 This suggests that the level of knowledge and understanding of the biology underpinning the effects of displacement and barrier effects on breeding seabird populations may be insufficient at the current time to enable reliable prediction using sophisticated individual-based modelling approaches (albeit that such approaches have considerable potential to advance the understanding of these effects). As such, the matrix approach may remain a more suitable method for estimating impacts from displacement and barrier effects at the current time, given its greater reliance on qualitative (and expert) consideration of what is likely to be biologically plausible and its dependence on bird abundance estimates from the actual sites of interest.

References

Dierschke V., Furness R.W. and Garthe S. (2016). Seabirds and offshore wind farms in European waters: avoidance and attraction. *Biological Conservation*, 202:59–68.

Furness, R.W., Wade, H.M. and Masden, E.A. (2013) Assessing vulnerability of marine bird populations to offshore wind farms. *Journal of Environmental Management*, 119:.56-66.

ICOL (2018). *Inch Cape Wind Farm and Offshore Transmission Works Habitats Regulation Appraisal*

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

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

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

SNCB (2017). Joint SNCB Interim Displacement Advice Note. Advice on how to present assessment information on the extent and potential consequences of seabird displacement from Offshore Wind farm (OWF) developments. Available at: http://jncc.defra.gov.uk/pdf/Joint_SNCB_Interim_Displacement_AdviceNote_2017.pdf [Accessed 15/05/18]

Wakefield, E.D., Owen, E., Baer, J., Daunt, F., Dodd, L.S., Green, J.A., Guildford, T., Mavor, R., Miller, P.I., Newell, M., Newton, S.F., Robertson, G., Shoji, A., Soanes, L.M., Votier, S., Wanless, S. and Bolton, M. (2017) Breeding density, fine scale telemetry and large-scale modelling reveal the regional distribution of a four-species seabird assemblage. *Ecological Applications*, 27: 2074–2091.

Annex 11D.1: Distribution and abundance of seabird species during surveys that contribute to the calculation of the mean peak counts

Figure 11D.1.1 Kittiwake distribution during surveys with the peak count in each breeding period

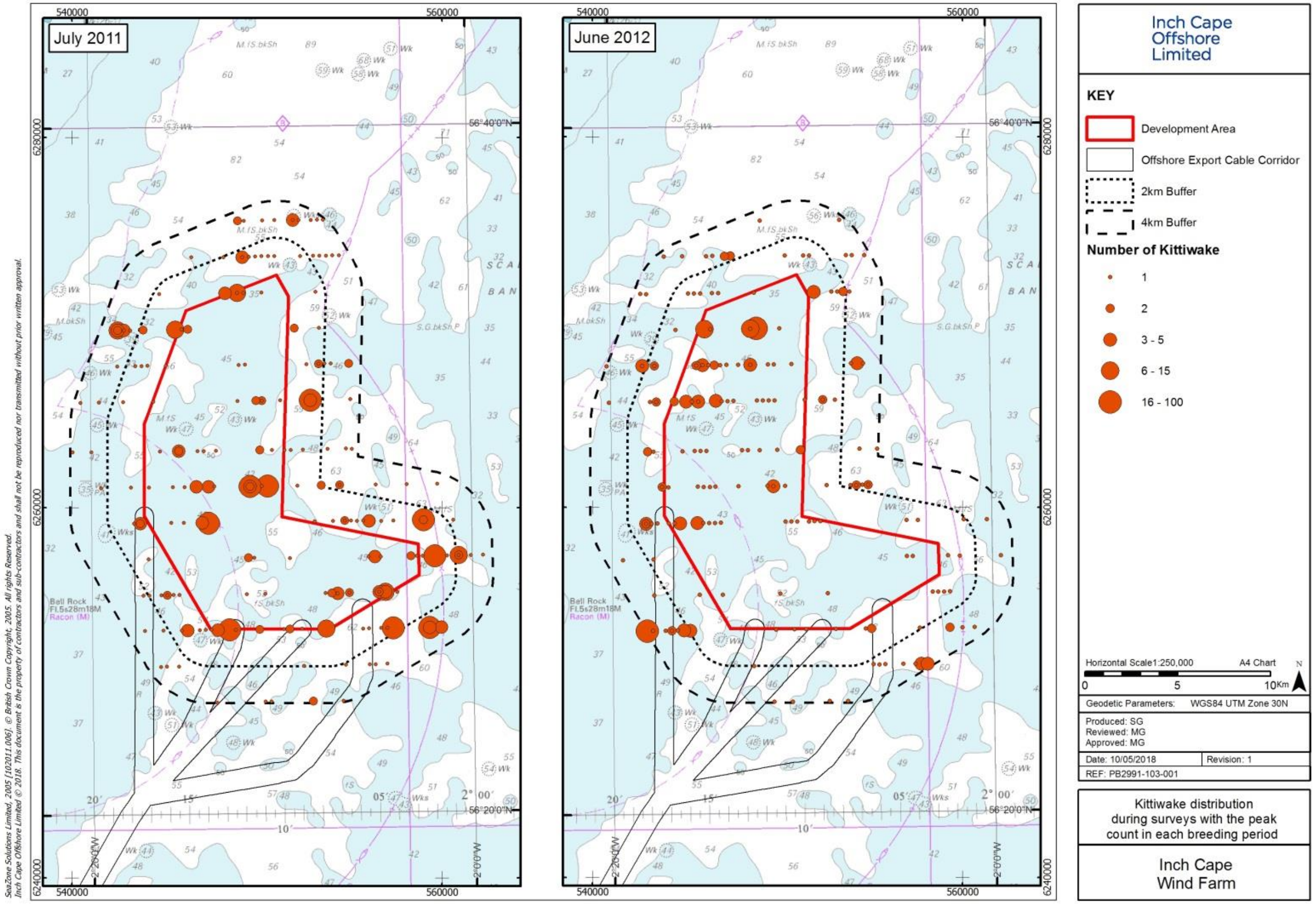


Figure 11D.1.2 Guillemot distribution during surveys with the peak count in each breeding period

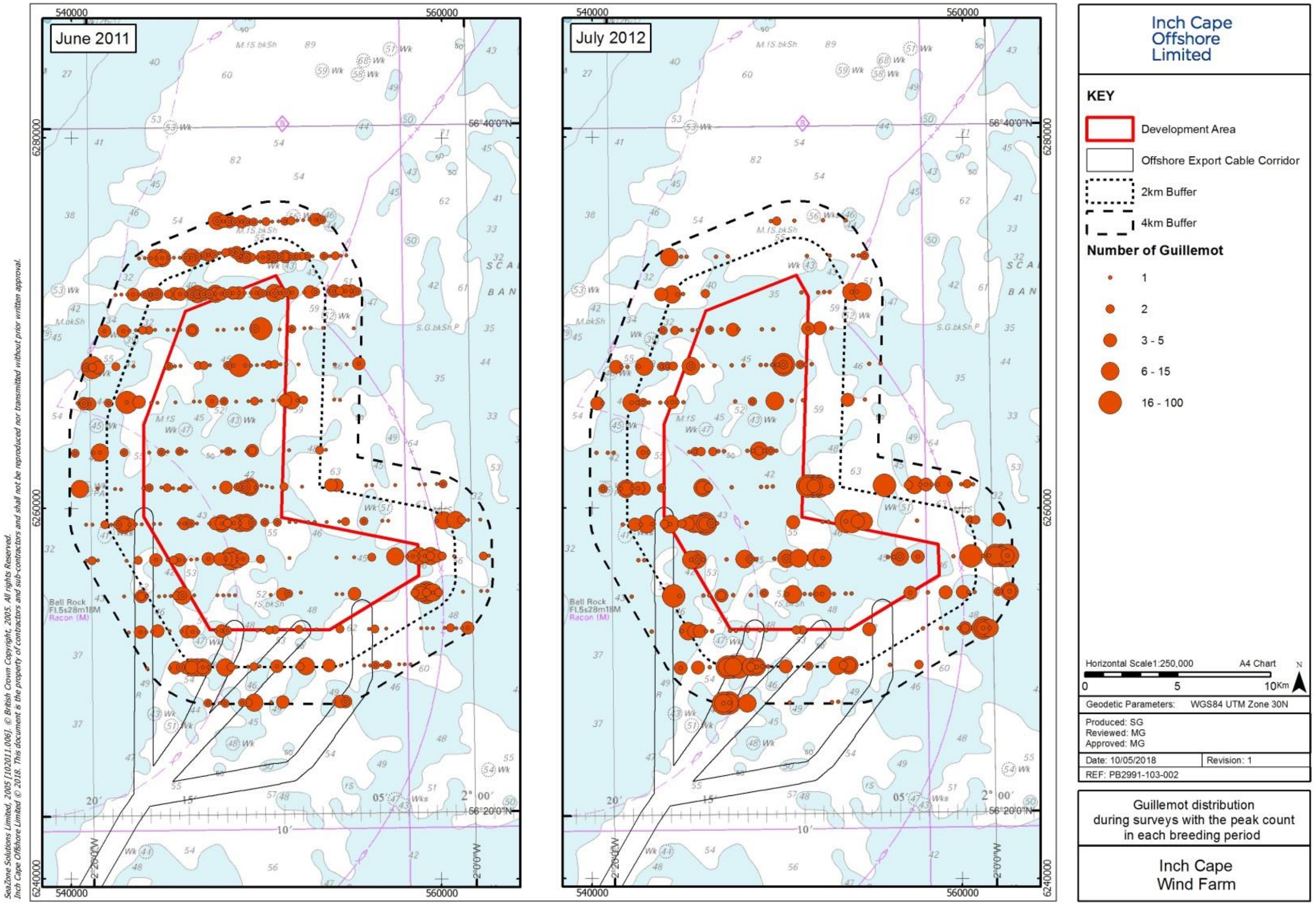


Figure 11D.1.3 Guillemot distribution during surveys with the peak count in each non-breeding period

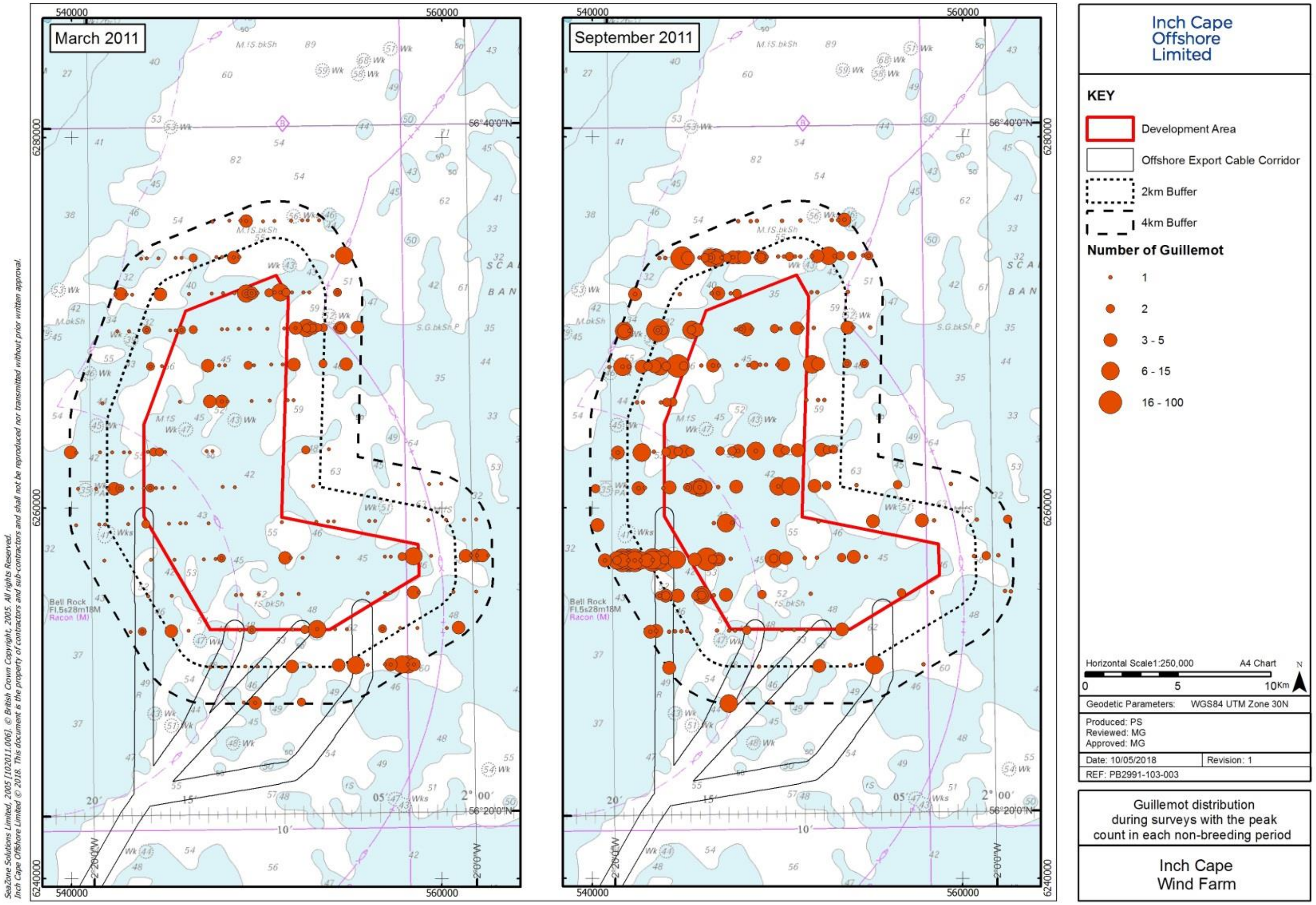


Figure 11D.1.4 Razorbill distribution during surveys with the peak count in each breeding period

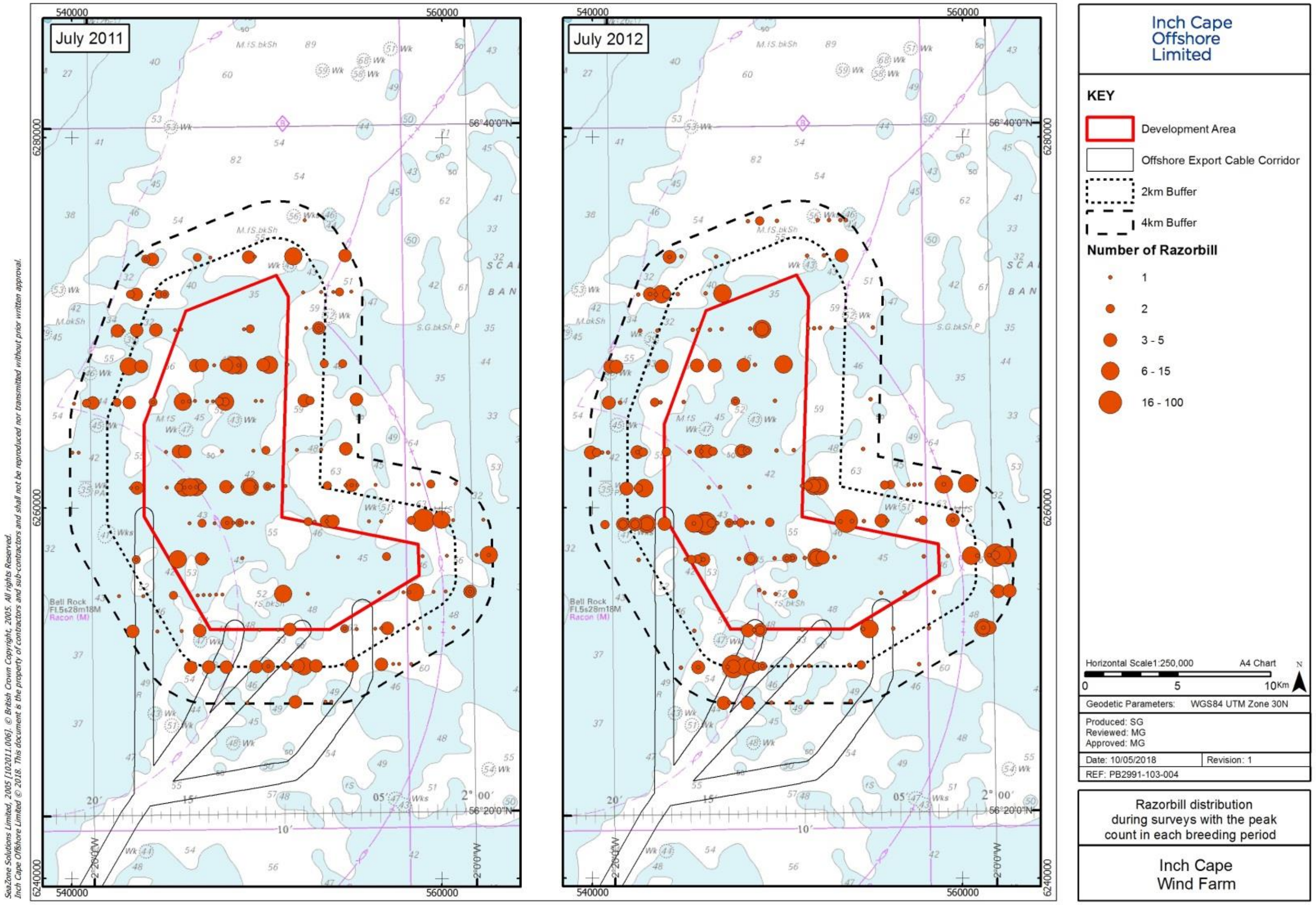


Figure 11D.1.5 Razorbill distribution during surveys with the peak count in each non-breeding period

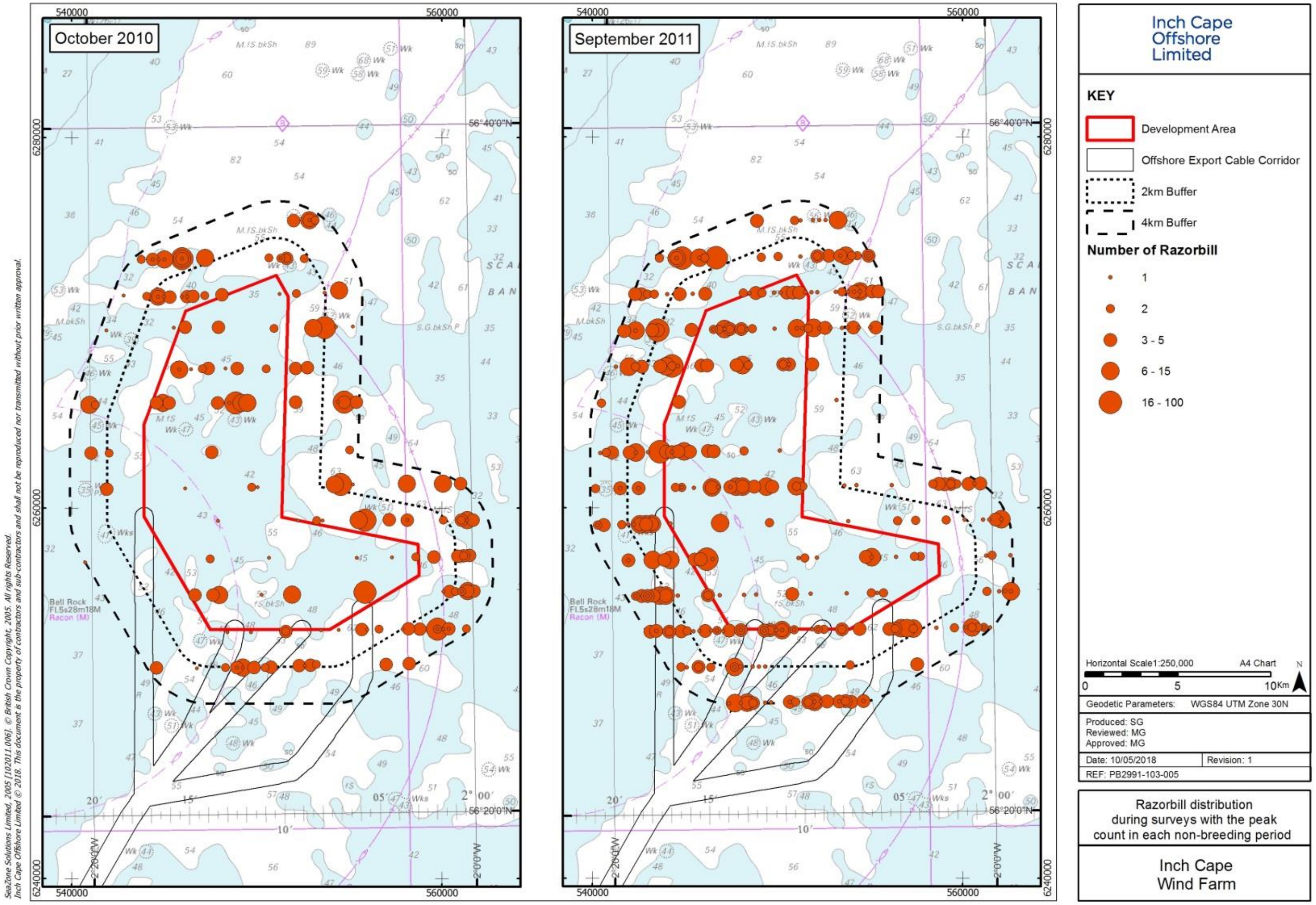
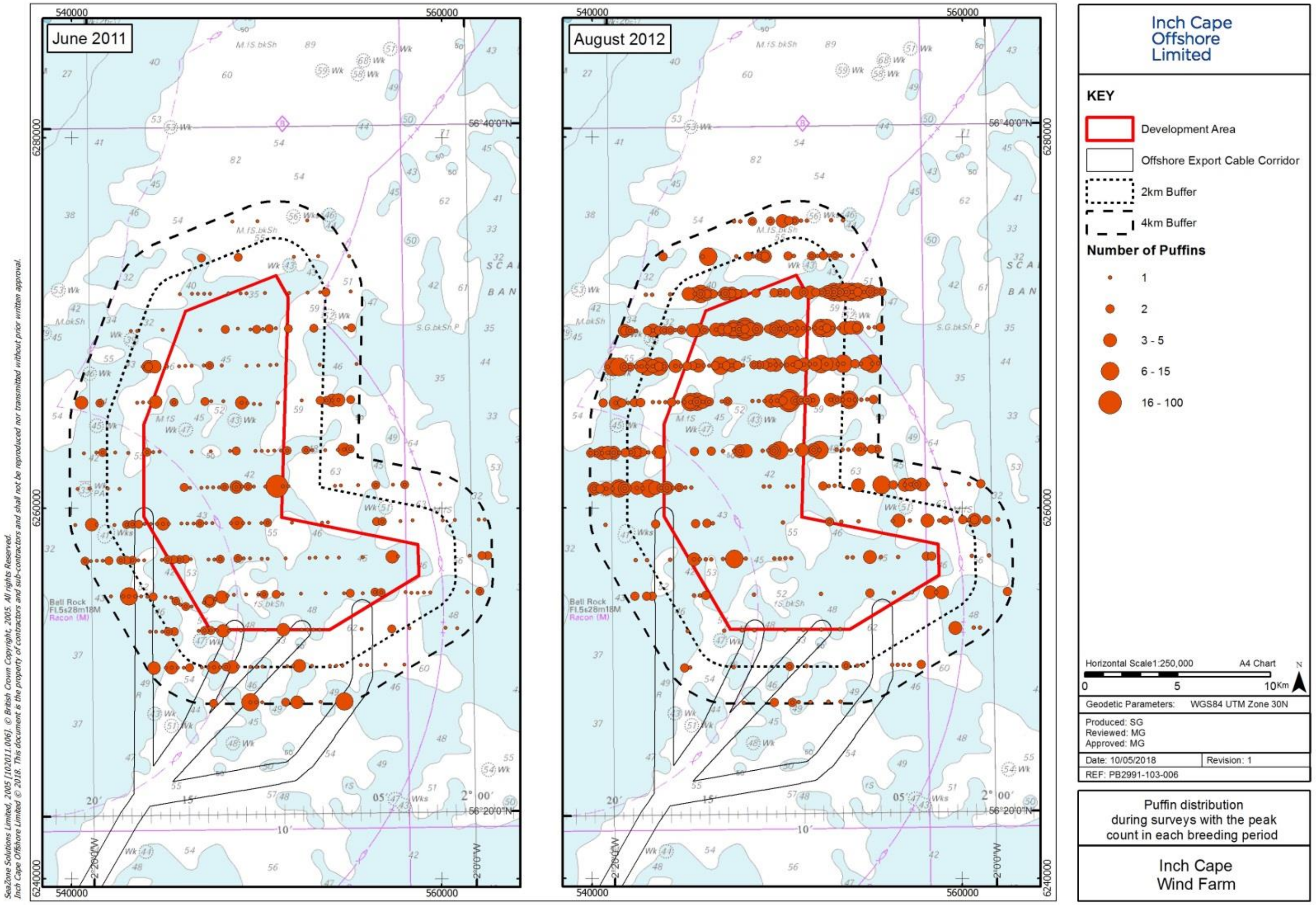


Figure 11D.1.6 Puffin distribution during surveys with the peak count in each breeding period



Annex 11D.2: Displacement matrices

Development-alone

Table 11D.2.1 Displacement matrix for kittiwake in the breeding season. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	4	8	12	15	19	23	27	31	35	39
	2%	0	8	15	**23**	31	39	46	54	62	70	77
	3%	0	12	23	35	46	58	70	81	93	104	116
	4%	0	15	31	46	62	77	93	108	124	139	155
	5%	0	19	39	58	77	97	116	135	155	174	193
	10%	0	39	77	116	155	193	232	271	309	348	387
	15%	0	58	116	174	232	290	348	406	464	522	580
	20%	0	77	155	232	309	387	464	541	619	696	773
	30%	0	116	232	348	464	580	696	812	928	1044	1160
	40%	0	155	309	464	619	773	928	1082	1237	1392	1546
	50%	0	193	387	580	773	966	1160	1353	1546	1740	1933
	60%	0	232	464	696	928	1160	1392	1624	1856	2088	2320
	70%	0	271	541	812	1082	1353	1624	1894	2165	2436	2706
	80%	0	309	619	928	1237	1546	1856	2165	2474	2784	3093
	90%	0	348	696	1044	1392	1740	2088	2436	2784	3131	3479
	100%	0	387	773	1160	1546	1933	2320	2706	3093	3479	3866

Table 11D.2.2 Displacement matrix for guillemot in the breeding season. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	8	16	25	33	41	**49**	57	65	74	82
	2%	0	16	33	49	65	82	98	115	131	147	164
	3%	0	25	49	74	98	123	147	172	196	221	246
	4%	0	33	65	98	131	164	196	229	262	295	327
	5%	0	41	82	123	164	205	246	286	327	368	409
	10%	0	82	164	246	327	409	491	573	655	737	818
	15%	0	123	246	368	491	614	737	859	982	1105	1228
	20%	0	164	327	491	655	818	982	1146	1309	1473	1637
	30%	0	246	491	737	982	1228	1473	1719	1964	2210	2455
	40%	0	327	655	982	1309	1637	1964	2291	2619	2946	3273
	50%	0	409	818	1228	1637	2046	2455	2864	3273	3683	4092
	60%	0	491	982	1473	1964	2455	2946	3437	3928	4419	4910
	70%	0	573	1146	1719	2291	2864	3437	4010	4583	5156	5728
	80%	0	655	1309	1964	2619	3273	3928	4583	5237	5892	6547
	90%	0	737	1473	2210	2946	3683	4419	5156	5892	6629	7365
	100%	0	818	1637	2455	3273	4092	4910	5728	6547	7365	8184

Table 11D.2.3 Displacement matrix for razorbill in the breeding season. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	5	9	14	19	23	**28**	33	37	42	47
	2%	0	9	19	28	37	47	56	65	75	84	93
	3%	0	14	28	42	56	70	84	98	112	126	140
	4%	0	19	37	56	75	93	112	131	149	168	187
	5%	0	23	47	70	93	117	140	163	187	210	234
	10%	0	47	93	140	187	234	280	327	374	420	467
	15%	0	70	140	210	280	350	420	490	560	631	701
	20%	0	93	187	280	374	467	560	654	747	841	934
	30%	0	140	280	420	560	701	841	981	1121	1261	1401
	40%	0	187	374	560	747	934	1121	1308	1495	1681	1868
	50%	0	234	467	701	934	1168	1401	1635	1868	2102	2335
	60%	0	280	560	841	1121	1401	1681	1962	2242	2522	2802
	70%	0	327	654	981	1308	1635	1962	2289	2616	2943	3270
	80%	0	374	747	1121	1495	1868	2242	2616	2989	3363	3737
	90%	0	420	841	1261	1681	2102	2522	2943	3363	3783	4204
	100%	0	467	934	1401	1868	2335	2802	3270	3737	4204	4671

Table 11D.2.4 Displacement matrix for puffin in the breeding season. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	6	11	17	23	28	34	40	45	51	57
	2%	0	11	23	34	45	57	**68**	79	91	102	114
	3%	0	17	34	51	68	85	102	119	136	153	170
	4%	0	23	45	68	91	114	136	159	182	204	227
	5%	0	28	57	85	114	142	170	199	227	256	284
	10%	0	57	114	170	227	284	341	397	454	511	568
	15%	0	85	170	256	341	426	511	596	681	767	852
	20%	0	114	227	341	454	568	681	795	908	1022	1136
	30%	0	170	341	511	681	852	1022	1192	1363	1533	1703
	40%	0	227	454	681	908	1136	1363	1590	1817	2044	2271
	50%	0	284	568	852	1136	1419	1703	1987	2271	2555	2839
	60%	0	341	681	1022	1363	1703	2044	2385	2725	3066	3407
	70%	0	397	795	1192	1590	1987	2385	2782	3180	3577	3974
	80%	0	454	908	1363	1817	2271	2725	3180	3634	4088	4542
	90%	0	511	1022	1533	2044	2555	3066	3577	4088	4599	5110
	100%	0	568	1136	1703	2271	2839	3407	3974	4542	5110	5678

Table 11D.2.5 Displacement matrix for guillemot in the non-breeding season. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	4	8	12	16	20	**23**	27	31	35	39
	2%	0	8	16	23	31	39	47	55	63	70	78
	3%	0	12	23	35	47	59	70	82	94	106	117
	4%	0	16	31	47	63	78	94	110	125	141	156
	5%	0	20	39	59	78	98	117	137	156	176	196
	10%	0	39	78	117	156	196	235	274	313	352	391
	15%	0	59	117	176	235	293	352	411	469	528	587
	20%	0	78	156	235	313	391	469	548	626	704	782
	30%	0	117	235	352	469	587	704	822	939	1056	1174
	40%	0	156	313	469	626	782	939	1095	1252	1408	1565
	50%	0	196	391	587	782	978	1174	1369	1565	1760	1956
	60%	0	235	469	704	939	1174	1408	1643	1878	2112	2347
	70%	0	274	548	822	1095	1369	1643	1917	2191	2465	2738
	80%	0	313	626	939	1252	1565	1878	2191	2504	2817	3130
	90%	0	352	704	1056	1408	1760	2112	2465	2817	3169	3521
	100%	0	391	782	1174	1565	1956	2347	2738	3130	3521	3912

Table 11D.2.6 Displacement matrix for razorbill in the non-breeding season. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	5	10	15	20	25	**29**	34	39	44	49
	2%	0	10	20	29	39	49	59	69	78	88	98
	3%	0	15	29	44	59	74	88	103	118	132	147
	4%	0	20	39	59	78	98	118	137	157	177	196
	5%	0	25	49	74	98	123	147	172	196	221	245
	10%	0	49	98	147	196	245	294	343	392	441	491
	15%	0	74	147	221	294	368	441	515	589	662	736
	20%	0	98	196	294	392	491	589	687	785	883	981
	30%	0	147	294	441	589	736	883	1030	1177	1324	1472
	40%	0	196	392	589	785	981	1177	1374	1570	1766	1962
	50%	0	245	491	736	981	1226	1472	1717	1962	2207	2453
	60%	0	294	589	883	1177	1472	1766	2060	2355	2649	2943
	70%	0	343	687	1030	1374	1717	2060	2404	2747	3090	3434
	80%	0	392	785	1177	1570	1962	2355	2747	3139	3532	3924
	90%	0	441	883	1324	1766	2207	2649	3090	3532	3973	4415
	100%	0	491	981	1472	1962	2453	2943	3434	3924	4415	4905

Cumulative

Table 11D.2.7 Displacement matrix for kittiwake in the breeding season at Neart na Gaoithe. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	2	4	6	9	11	13	15	17	19	22
	2%	0	4	9	13	17	22	26	30	35	39	43
	3%	0	6	13	19	26	32	39	45	52	58	65
	4%	0	9	17	26	35	43	52	61	69	78	87
	5%	0	11	22	32	43	54	65	76	87	97	108
	10%	0	22	43	65	87	108	130	151	173	195	216
	15%	0	32	65	97	130	162	195	227	260	292	325
	20%	0	43	87	130	173	216	260	303	346	390	433
	30%	0	65	130	195	260	325	390	454	519	584	649
	40%	0	87	173	260	346	433	519	606	692	779	866
	50%	0	108	216	325	433	541	649	757	866	974	1082
	60%	0	130	260	390	519	649	779	909	1039	1169	1298
	70%	0	151	303	454	606	757	909	1060	1212	1363	1515
	80%	0	173	346	519	692	866	1039	1212	1385	1558	1731
	90%	0	195	390	584	779	974	1169	1363	1558	1753	1948
	100%	0	216	433	649	866	1082	1298	1515	1731	1948	2164

Table 11D.2.8 Displacement matrix for kittiwake in the breeding season at Seagreen Alpha. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	2	4	7	9	11	13	16	18	20	22
	2%	0	4	9	13	18	22	27	31	36	40	44
	3%	0	7	13	20	27	33	40	47	53	60	67
	4%	0	9	18	27	36	44	53	62	71	80	89
	5%	0	11	22	33	44	55	67	78	89	100	111
	10%	0	22	44	67	89	111	133	155	178	200	222
	15%	0	33	67	100	133	166	200	233	266	300	333
	20%	0	44	89	133	178	222	266	311	355	400	444
	30%	0	67	133	200	266	333	400	466	533	599	666
	40%	0	89	178	266	355	444	533	622	710	799	888
	50%	0	111	222	333	444	555	666	777	888	999	1110
	60%	0	133	266	400	533	666	799	932	1066	1199	1332
	70%	0	155	311	466	622	777	932	1088	1243	1399	1554
	80%	0	178	355	533	710	888	1066	1243	1421	1598	1776
	90%	0	200	400	599	799	999	1199	1399	1598	1798	1998
	100%	0	222	444	666	888	1110	1332	1554	1776	1998	2220

Table 11D.2.9 Displacement matrix for kittiwake in the breeding season at Seagreen Bravo. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	3	5	8	11	14	16	19	22	24	27
	2%	0	5	11	16	22	27	32	38	43	49	54
	3%	0	8	16	24	32	41	49	57	65	73	81
	4%	0	11	22	32	43	54	65	76	87	97	108
	5%	0	14	27	41	54	68	81	95	108	122	135
	10%	0	27	54	81	108	135	162	190	217	244	271
	15%	0	41	81	122	162	203	244	284	325	366	406
	20%	0	54	108	162	217	271	325	379	433	487	541
	30%	0	81	162	244	325	406	487	569	650	731	812
	40%	0	108	217	325	433	541	650	758	866	975	1083
	50%	0	135	271	406	541	677	812	948	1083	1218	1354
	60%	0	162	325	487	650	812	975	1137	1300	1462	1624
	70%	0	190	379	569	758	948	1137	1327	1516	1706	1895
	80%	0	217	433	650	866	1083	1300	1516	1733	1949	2166
	90%	0	244	487	731	975	1218	1462	1706	1949	2193	2437
	100%	0	271	541	812	1083	1354	1624	1895	2166	2437	2707

Table 11D.2.10 Displacement matrix for guillemot in the breeding season at Neart na Gaoithe. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	3	7	10	13	16	20	23	26	29	33
	2%	0	7	13	20	26	33	39	46	52	59	65
	3%	0	10	20	29	39	49	59	69	78	88	98
	4%	0	13	26	39	52	65	78	91	104	117	131
	5%	0	16	33	49	65	82	98	114	131	147	163
	10%	0	33	65	98	131	163	196	228	261	294	326
	15%	0	49	98	147	196	245	294	343	392	440	489
	20%	0	65	131	196	261	326	392	457	522	587	653
	30%	0	98	196	294	392	489	587	685	783	881	979
	40%	0	131	261	392	522	653	783	914	1044	1175	1305
	50%	0	163	326	489	653	816	979	1142	1305	1468	1631
	60%	0	196	392	587	783	979	1175	1370	1566	1762	1958
	70%	0	228	457	685	914	1142	1370	1599	1827	2055	2284
	80%	0	261	522	783	1044	1305	1566	1827	2088	2349	2610
	90%	0	294	587	881	1175	1468	1762	2055	2349	2643	2936
	100%	0	326	653	979	1305	1631	1958	2284	2610	2936	3263

Table 11D.2.11 Displacement matrix for guillemot in the breeding season at Seagreen Alpha. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

	DISPLACEMENT										
MORTALITY	0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
0%	0	0	0	0	0	0	0	0	0	0	0
1%	0	12	24	37	49	61	**73**	85	98	110	122
2%	0	24	49	73	98	122	146	171	195	219	244
3%	0	37	73	110	146	183	219	256	293	329	366
4%	0	49	98	146	195	244	293	341	390	439	488
5%	0	61	122	183	244	305	366	427	488	549	609
10%	0	122	244	366	488	609	731	853	975	1097	1219
15%	0	183	366	549	731	914	1097	1280	1463	1646	1828
20%	0	244	488	731	975	1219	1463	1707	1950	2194	2438
30%	0	366	731	1097	1463	1828	2194	2560	2926	3291	3657
40%	0	488	975	1463	1950	2438	2926	3413	3901	4388	4876
50%	0	609	1219	1828	2438	3047	3657	4266	4876	5485	6095
60%	0	731	1463	2194	2926	3657	4388	5120	5851	6582	7314
70%	0	853	1707	2560	3413	4266	5120	5973	6826	7680	8533
80%	0	975	1950	2926	3901	4876	5851	6826	7801	8777	9752
90%	0	1097	2194	3291	4388	5485	6582	7680	8777	9874	10971
100%	0	1219	2438	3657	4876	6095	7314	8533	9752	10971	12190

Table 11D.2.12 Displacement matrix for guillemot in the breeding season at Seagreen Bravo. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

	DISPLACEMENT										
MORTALITY	0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
0%	0	0	0	0	0	0	0	0	0	0	0
1%	0	11	22	32	43	54	**65**	75	86	97	108
2%	0	22	43	65	86	108	129	151	172	194	216
3%	0	32	65	97	129	162	194	226	259	291	323
4%	0	43	86	129	172	216	259	302	345	388	431
5%	0	54	108	162	216	269	323	377	431	485	539
10%	0	108	216	323	431	539	647	754	862	970	1078
15%	0	162	323	485	647	808	970	1132	1293	1455	1617
20%	0	216	431	647	862	1078	1293	1509	1724	1940	2156
30%	0	323	647	970	1293	1617	1940	2263	2587	2910	3233
40%	0	431	862	1293	1724	2156	2587	3018	3449	3880	4311
50%	0	539	1078	1617	2156	2695	3233	3772	4311	4850	5389
60%	0	647	1293	1940	2587	3233	3880	4527	5173	5820	6467
70%	0	754	1509	2263	3018	3772	4527	5281	6036	6790	7545
80%	0	862	1724	2587	3449	4311	5173	6036	6898	7760	8622
90%	0	970	1940	2910	3880	4850	5820	6790	7760	8730	9700
100%	0	1078	2156	3233	4311	5389	6467	7545	8622	9700	10778

Table 11D.2.13 Displacement matrix for guillemot in the non-breeding season at Neart na Gaoithe. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	8	15	23	30	38	**46**	53	61	69	76
	2%	0	15	30	46	61	76	91	107	122	137	152
	3%	0	23	46	69	91	114	137	160	183	206	229
	4%	0	30	61	91	122	152	183	213	244	274	305
	5%	0	38	76	114	152	190	229	267	305	343	381
	10%	0	76	152	229	305	381	457	533	609	686	762
	15%	0	114	229	343	457	571	686	800	914	1028	1143
	20%	0	152	305	457	609	762	914	1067	1219	1371	1524
	30%	0	229	457	686	914	1143	1371	1600	1828	2057	2286
	40%	0	305	609	914	1219	1524	1828	2133	2438	2743	3047
	50%	0	381	762	1143	1524	1905	2286	2666	3047	3428	3809
	60%	0	457	914	1371	1828	2286	2743	3200	3657	4114	4571
	70%	0	533	1067	1600	2133	2666	3200	3733	4266	4800	5333
	80%	0	609	1219	1828	2438	3047	3657	4266	4876	5485	6095
	90%	0	686	1371	2057	2743	3428	4114	4800	5485	6171	6857
	100%	0	762	1524	2286	3047	3809	4571	5333	6095	6857	7618

Table 11D.2.14 Displacement matrix for guillemot in the non-breeding season at Seagreen Alpha. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	6	12	18	25	31	**37**	43	49	55	61
	2%	0	12	25	37	49	61	74	86	98	110	123
	3%	0	18	37	55	74	92	110	129	147	166	184
	4%	0	25	49	74	98	123	147	172	196	221	245
	5%	0	31	61	92	123	153	184	215	245	276	307
	10%	0	61	123	184	245	307	368	429	491	552	613
	15%	0	92	184	276	368	460	552	644	736	828	920
	20%	0	123	245	368	491	613	736	858	981	1104	1226
	30%	0	184	368	552	736	920	1104	1288	1472	1655	1839
	40%	0	245	491	736	981	1226	1472	1717	1962	2207	2453
	50%	0	307	613	920	1226	1533	1839	2146	2453	2759	3066
	60%	0	368	736	1104	1472	1839	2207	2575	2943	3311	3679
	70%	0	429	858	1288	1717	2146	2575	3004	3434	3863	4292
	80%	0	491	981	1472	1962	2453	2943	3434	3924	4415	4905
	90%	0	552	1104	1655	2207	2759	3311	3863	4415	4966	5518
	100%	0	613	1226	1839	2453	3066	3679	4292	4905	5518	6131

Table 11D.2.15 Displacement matrix for guillemot in the non-breeding season at Seagreen Bravo. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	7	14	20	27	34	41	47	54	61	68
	2%	0	14	27	41	54	68	81	95	108	122	136
	3%	0	20	41	61	81	102	122	142	163	183	203
	4%	0	27	54	81	108	136	163	190	217	244	271
	5%	0	34	68	102	136	170	203	237	271	305	339
	10%	0	68	136	203	271	339	407	475	542	610	678
	15%	0	102	203	305	407	509	610	712	814	915	1017
	20%	0	136	271	407	542	678	814	949	1085	1220	1356
	30%	0	203	407	610	814	1017	1220	1424	1627	1831	2034
	40%	0	271	542	814	1085	1356	1627	1898	2170	2441	2712
	50%	0	339	678	1017	1356	1695	2034	2373	2712	3051	3390
	60%	0	407	814	1220	1627	2034	2441	2848	3254	3661	4068
	70%	0	475	949	1424	1898	2373	2848	3322	3797	4271	4746
	80%	0	542	1085	1627	2170	2712	3254	3797	4339	4882	5424
	90%	0	610	1220	1831	2441	3051	3661	4271	4882	5492	6102
	100%	0	678	1356	2034	2712	3390	4068	4746	5424	6102	6780

Table 11D. 2.16 Displacement matrix for razorbill in the breeding season at Neart na Gaoithe. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	1	2	4	5	6	7	9	10	11	12
	2%	0	2	5	7	10	12	15	17	20	22	25
	3%	0	4	7	11	15	19	22	26	30	34	37
	4%	0	5	10	15	20	25	30	35	40	45	50
	5%	0	6	12	19	25	31	37	44	50	56	62
	10%	0	12	25	37	50	62	75	87	100	112	125
	15%	0	19	37	56	75	94	112	131	150	168	187
	20%	0	25	50	75	100	125	150	175	200	225	250
	30%	0	37	75	112	150	187	225	262	300	337	374
	40%	0	50	100	150	200	250	300	349	399	449	499
	50%	0	62	125	187	250	312	374	437	499	562	624
	60%	0	75	150	225	300	374	449	524	599	674	749
	70%	0	87	175	262	349	437	524	612	699	786	874
	80%	0	100	200	300	399	499	599	699	799	899	998
	90%	0	112	225	337	449	562	674	786	899	1011	1123
	100%	0	125	250	374	499	624	749	874	998	1123	1248

Table 11D.2.17 Displacement matrix for razorbill in the breeding season at Seagreen Alpha. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	3	6	8	11	14	17	19	22	25	28
	2%	0	6	11	17	22	28	33	39	44	50	55
	3%	0	8	17	25	33	42	50	58	66	75	83
	4%	0	11	22	33	44	55	66	78	89	100	111
	5%	0	14	28	42	55	69	83	97	111	125	138
	10%	0	28	55	83	111	138	166	194	221	249	277
	15%	0	42	83	125	166	208	249	291	332	374	415
	20%	0	55	111	166	221	277	332	388	443	498	554
	30%	0	83	166	249	332	415	498	581	664	747	830
	40%	0	111	221	332	443	554	664	775	886	996	1107
	50%	0	138	277	415	554	692	830	969	1107	1246	1384
	60%	0	166	332	498	664	830	996	1163	1329	1495	1661
	70%	0	194	388	581	775	969	1163	1356	1550	1744	1938
	80%	0	221	443	664	886	1107	1329	1550	1772	1993	2214
	90%	0	249	498	747	996	1246	1495	1744	1993	2242	2491
	100%	0	277	554	830	1107	1384	1661	1938	2214	2491	2768

Table 11D.2.18 Displacement matrix for razorbill in the breeding season at Seagreen Bravo. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	1	2	3	4	5	6	7	8	9	10
	2%	0	2	4	6	8	10	12	14	16	18	20
	3%	0	3	6	9	12	15	18	21	24	27	30
	4%	0	4	8	12	16	20	24	28	32	36	40
	5%	0	5	10	15	20	25	30	35	40	45	50
	10%	0	10	20	30	40	50	60	70	79	89	99
	15%	0	15	30	45	60	75	89	104	119	134	149
	20%	0	20	40	60	79	99	119	139	159	179	199
	30%	0	30	60	89	119	149	179	209	238	268	298
	40%	0	40	79	119	159	199	238	278	318	358	397
	50%	0	50	99	149	199	248	298	348	397	447	497
	60%	0	60	119	179	238	298	358	417	477	536	596
	70%	0	70	139	209	278	348	417	487	556	626	695
	80%	0	79	159	238	318	397	477	556	636	715	795
	90%	0	89	179	268	358	447	536	626	715	805	894
	100%	0	99	199	298	397	497	596	695	795	894	993

Table 11D.2.19 Displacement matrix for razorbill in the non-breeding season at Neart na Gaoithe. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	3	6	9	12	16	19	22	25	28	31
	2%	0	6	12	19	25	31	37	43	50	56	62
	3%	0	9	19	28	37	47	56	65	74	84	93
	4%	0	12	25	37	50	62	74	87	99	112	124
	5%	0	16	31	47	62	78	93	109	124	140	155
	10%	0	31	62	93	124	155	186	217	248	279	310
	15%	0	47	93	140	186	233	279	326	372	419	465
	20%	0	62	124	186	248	310	372	434	496	558	620
	30%	0	93	186	279	372	465	558	651	744	837	930
	40%	0	124	248	372	496	620	744	868	992	1116	1240
	50%	0	155	310	465	620	775	930	1085	1240	1395	1550
	60%	0	186	372	558	744	930	1116	1302	1488	1674	1860
	70%	0	217	434	651	868	1085	1302	1519	1736	1953	2170
	80%	0	248	496	744	992	1240	1488	1736	1984	2232	2481
	90%	0	279	558	837	1116	1395	1674	1953	2232	2512	2791
	100%	0	310	620	930	1240	1550	1860	2170	2481	2791	3101

Table 11D.2.20 Displacement matrix for razorbill in the non-breeding season at Seagreen Alpha. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	1	3	4	5	6	8	9	10	11	13
	2%	0	3	5	8	10	13	15	18	20	23	25
	3%	0	4	8	11	15	19	23	26	30	34	38
	4%	0	5	10	15	20	25	30	35	40	45	50
	5%	0	6	13	19	25	31	38	44	50	56	63
	10%	0	13	25	38	50	63	75	88	100	113	125
	15%	0	19	38	56	75	94	113	132	150	169	188
	20%	0	25	50	75	100	125	150	175	200	226	251
	30%	0	38	75	113	150	188	226	263	301	338	376
	40%	0	50	100	150	200	251	301	351	401	451	501
	50%	0	63	125	188	251	313	376	439	501	564	627
	60%	0	75	150	226	301	376	451	526	601	677	752
	70%	0	88	175	263	351	439	526	614	702	789	877
	80%	0	100	200	301	401	501	601	702	802	902	1002
	90%	0	113	226	338	451	564	677	789	902	1015	1128
	100%	0	125	251	376	501	627	752	877	1002	1128	1253

Table 11D.2.21 Displacement matrix for razorbill in the non-breeding season at Seagreen Bravo. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	2	3	5	7	9	10	12	14	16	17
	2%	0	3	7	10	14	17	21	24	28	31	34
	3%	0	5	10	16	21	26	31	36	41	47	52
	4%	0	7	14	21	28	34	41	48	55	62	69
	5%	0	9	17	26	34	43	52	60	69	78	86
	10%	0	17	34	52	69	86	103	121	138	155	172
	15%	0	26	52	78	103	129	155	181	207	233	258
	20%	0	34	69	103	138	172	207	241	276	310	345
	30%	0	52	103	155	207	258	310	362	414	465	517
	40%	0	69	138	207	276	345	414	482	551	620	689
	50%	0	86	172	258	345	431	517	603	689	775	862
	60%	0	103	207	310	414	517	620	724	827	931	1034
	70%	0	121	241	362	482	603	724	844	965	1086	1206
	80%	0	138	276	414	551	689	827	965	1103	1241	1379
	90%	0	155	310	465	620	775	931	1086	1241	1396	1551
	100%	0	172	345	517	689	862	1034	1206	1379	1551	1723

Table 11D.2.22 Displacement matrix for puffin in the breeding season at Neart na Gaoithe. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	6	12	19	25	31	37	43	49	56	62
	2%	0	12	25	37	49	62	74	86	99	111	123
	3%	0	19	37	56	74	93	111	130	148	167	185
	4%	0	25	49	74	99	123	148	173	198	222	247
	5%	0	31	62	93	123	154	185	216	247	278	309
	10%	0	62	123	185	247	309	370	432	494	556	617
	15%	0	93	185	278	370	463	556	648	741	833	926
	20%	0	123	247	370	494	617	741	864	988	1111	1235
	30%	0	185	370	556	741	926	1111	1296	1481	1667	1852
	40%	0	247	494	741	988	1235	1481	1728	1975	2222	2469
	50%	0	309	617	926	1235	1543	1852	2160	2469	2778	3086
	60%	0	370	741	1111	1481	1852	2222	2593	2963	3333	3704
	70%	0	432	864	1296	1728	2160	2593	3025	3457	3889	4321
	80%	0	494	988	1481	1975	2469	2963	3457	3951	4444	4938
	90%	0	556	1111	1667	2222	2778	3333	3889	4444	5000	5555
	100%	0	617	1235	1852	2469	3086	3704	4321	4938	5555	6173

Table 11D.2.23 Displacement matrix for puffin in the breeding season at Seagreen Alpha. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	4	7	11	15	19	22	26	30	33	37
	2%	0	7	15	22	30	37	44	52	59	67	74
	3%	0	11	22	33	44	56	67	78	89	100	111
	4%	0	15	30	44	59	74	89	104	119	133	148
	5%	0	19	37	56	74	93	111	130	148	167	185
	10%	0	37	74	111	148	185	222	259	296	333	370
	15%	0	56	111	167	222	278	333	389	445	500	556
	20%	0	74	148	222	296	370	445	519	593	667	741
	30%	0	111	222	333	445	556	667	778	889	1000	1111
	40%	0	148	296	445	593	741	889	1037	1185	1334	1482
	50%	0	185	370	556	741	926	1111	1297	1482	1667	1852
	60%	0	222	445	667	889	1111	1334	1556	1778	2000	2223
	70%	0	259	519	778	1037	1297	1556	1815	2074	2334	2593
	80%	0	296	593	889	1185	1482	1778	2074	2371	2667	2964
	90%	0	333	667	1000	1334	1667	2000	2334	2667	3001	3334
	100%	0	370	741	1111	1482	1852	2223	2593	2964	3334	3704

Table 11D.2.24 Displacement matrix for puffin in the breeding season at Seagreen Bravo. Based on mean peak abundance. Recommended displacement rate and mortality rate is shown in dark green.

		DISPLACEMENT										
		0%	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
MORTALITY	0%	0	0	0	0	0	0	0	0	0	0	0
	1%	0	5	11	16	21	27	32	37	43	48	53
	2%	0	11	21	32	43	53	64	75	85	96	107
	3%	0	16	32	48	64	80	96	112	128	144	160
	4%	0	21	43	64	85	107	128	150	171	192	214
	5%	0	27	53	80	107	134	160	187	214	240	267
	10%	0	53	107	160	214	267	320	374	427	481	534
	15%	0	80	160	240	320	401	481	561	641	721	801
	20%	0	107	214	320	427	534	641	748	854	961	1068
	30%	0	160	320	481	641	801	961	1121	1282	1442	1602
	40%	0	214	427	641	854	1068	1282	1495	1709	1922	2136
	50%	0	267	534	801	1068	1335	1602	1869	2136	2403	2670
	60%	0	320	641	961	1282	1602	1922	2243	2563	2884	3204
	70%	0	374	748	1121	1495	1869	2243	2617	2991	3364	3738
	80%	0	427	854	1282	1709	2136	2563	2991	3418	3845	4272
	90%	0	481	961	1442	1922	2403	2884	3364	3845	4326	4806
	100%	0	534	1068	1602	2136	2670	3204	3738	4272	4806	5340

Annex 11D.3: Details of Methods and Results of the SeabORD Modelling Undertaken in Relation to the Inch Cape Wind Farm (based on information provided to ICOL by CEH)

METHODS

Individual based simulation model (SeabORD)

The prototype tool (SeabORD) was used to estimate the impact of the Inch Cape Wind Farm alone, and in combination with other wind farms in the Forth and Tay Region (i.e. Neart na Gaoithe, Seagreen Alpha and Seagreen Bravo), on four species of seabirds (kittiwake, razorbill, puffin and guillemot) during the breeding season. This tool simulates individual behaviour and energetics in 'baseline' scenarios with no wind farm present, and compares resulting population demographic estimates to model runs with the wind farm present. Specifically, the model simulates changes in seabird behaviour and energetics arising from displacement and barrier effects. Final model metrics are produced for additional adult and chick mortality (expressed as percentage points) arising as a result of the wind farm(s), assuming moderate environmental conditions.

At the time of writing, the model has not been published and a full description of the tool and underlying methodology are not available.

SeabORD Metrics

For each species, 10 matched paired model runs were used to calculate a single metric assessing additional adult and chick mortality as a result of displacement and barrier effects from the wind farm(s) of interest. Initial model baseline runs are first used to identify the range of median prey values within the model that result in 'moderate' conditions (based on empirical data for adult mass loss over the chick-rearing period).

Once the lower and upper bound for moderate conditions have been established, 10 paired runs are executed over this range using stratified random sampling to produce 10 estimates for each model metric, capturing variation over the 'moderate' prey range. These estimates are then combined to produce single metrics for additional adult and chick mortality for each breeding colony of interest (**P1**).

Metric **P1** calculates the population-level impact (in terms of the change in adult and chick mortality of the wind farm(s):

(mortality with wind farm(s) present - mortality in baseline) / (population size)

More specifically:

$$P1 = 100 * \frac{\text{(Total number of birds simulated to die when the wind farm(s) present} - \text{Total number of birds simulated to die when the wind farms absent)}}{\text{Total population size}}$$

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

The main population level metric (**P1**) is calculated for both adults and chicks (with the formulae being identical in all cases: “birds” is simply replaced with either “adults” or “chicks”).

Prediction intervals

Outputs are generated for each model run for any particular output e.g. the change in adult mortality that results from including the wind farm(s). For each metric the following is calculated:

the mean of this value across runs, m (to provide a “best estimate” for this quantity); and

the SD across runs, s, to capture the uncertainty associated with natural stochastic variation.

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

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

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

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

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

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

Model input for bird densities and prey availability

For each species, available Global Positioning System (GPS) tracking data was compiled from each of the breeding colonies of interest (Searle *et al.* 2014; Wakefield *et al.* 2017). Birds in flight were removed from GPS tracking data by applying a speed threshold for each species, leaving GPS location points for birds foraging and resting on the sea surface. An inhomogenous Poisson point process model (Generalised Additive Model (GAM); sensu Wakefield *et al*. 2017) was then applied to estimate the spatial density of bird locations around each colony. Prey availability was estimated from a GAM model of bird GPS locations assuming that once the accessibility (distance from source colony) and competition (distance from next nearest colony) effects are accounted for, the remaining spatial distribution in the intensity of usage is due to prey availability.

Foraging ranges were set for each species based on distances travelled from the source colony in the GPS tracking data (with the maximum assumed distances being as follows - kittiwake: 300 km, guillemot: 200 km, razorbill: 105 km, puffin: 132 km).

Uniform prey availability had to be used for some razorbills and puffins due to GPS tracking data only being available from one colony for each of these species. This prevented reliable use of the GAM to estimate spatial variation in prey availability around all colonies of interest.

RESULTS

Details are presented below of the results from the model runs for each species, along with the model parameters in each case. For each species, the results are presented only for those SPA colonies that were identified as having connectivity to the Development Area and two kilometre buffer during the breeding period (*Appendix 11B*).

Kittiwake

The following model parameters were set for each of the 10 paired runs:

- 100% of total population
- Colonies included: Forth Islands SPA, Fowlsheugh SPA, St Abb's Head to Fast Castle SPA, Buchan Ness to Collieston Coast SPA, Farne Islands SPA, Angus coast colonies
- Probability of displacement: 30%[9]
- Probability of barrier effect: 100% (all displacement susceptible birds are also barrier-susceptible)[9]
- Buffer around wind farm Development Area[10]: 2km
- Buffer size (i.e. area around the wind farm into which birds are displaced to forage): 5km
- Mapped bird densities and prey availability derived from most recent GPS tracking data with birds in flight removed (using speed threshold)

Table 11D.3.1. Effect sizes (percentage points change in additional mortality) for the Inch Cape Wind Farm alone based on 10 paired runs covering the range of 'moderate' prey conditions for breeding birds based on baseline model output.

	Forth Islands	Fowlsheugh	St Abb's Head to Fast Castle
Adult mortality			
Mean	0.20	0.005	0.002
95% prediction interval	(0.03, 0.37)	(-0.01, 0.02)	(-0.02, 0.02)
Chick mortality			
Mean	1.10	0.08	-0.003
95% prediction interval	(-0.54, 2.73)	(-0.37, 0.53)	(-0.04, 0.04)

[9] Probability of displacement for all species is based upon the displacement rates advised for that species in the Scoping Opinion, with the same assumptions made in relation to barrier effects.
[10] Based upon the advice in the Scoping Opinion.

Table 11D.3.2. Effect sizes (percentage points change in additional mortality) for the Inch Cape Wind Farm combined with the other Forth and Tay wind farms based on 10 paired runs covering the range of 'moderate' prey conditions for breeding birds based on baseline model output.

	Forth Islands	Fowlsheugh	St Abb's Head to Fast Castle
Adult mortality			
Mean	0.84	0.10	0.04
95% prediction interval	(-0.19, 1.87)	(0.02, 0.18)	(-0.05, 0.12)
Chick mortality			
Mean	5.89	0.49	0.14
95% prediction interval	(-0.19, 11.97)	(-0.63, 1.61)	(-0.10, 0.38)

Razorbill

The following model parameters were set for each of the 10 paired runs:

- 100% of total population
- Colonies included: Forth Islands SPA, Fowlsheugh SPA, St Abb's Head to Fast Castle SPA, Farne Islands SPA, Angus coast colonies
- Probability of displacement: 60%[9]
- Probability of barrier effect: 100% (all displacement susceptible birds are also barrier-susceptible)[9]
- Buffer around wind farm Development Area[10]: 2km
- Buffer size: 5km
- Mapped bird densities derived from most recent GPS tracking data with birds in flight removed (using speed threshold) and uniform prey availability (since tracking data were only available from one colony).

Table 11D.3.3. Effect sizes (percentage points change in additional mortality) for the Inch Cape Wind Farm alone based on 10 paired runs covering the range of 'moderate' prey conditions for breeding birds based on baseline model output.

	Forth Islands	Fowlsheugh
Adult mortality		
Mean	0.24	0.14
95% prediction interval	(-0.04, 0.52)	(-0.03, 0.32)
Chick mortality		
Mean	0.65	0.40
95% prediction interval	(-0.60, 1.90)	(-1.25, 2.04)

Table 11D.3.4. Effect sizes (percentage points change in additional mortality) for the Inch Cape Wind Farm combined with the other Forth and Tay wind farms based on 10 paired runs covering the range of 'moderate' prey conditions for breeding birds based on baseline model output.

	Forth Islands	Fowlsheugh
Adult mortality		
Mean	0.59	0.27
95% prediction interval	(0.22, 0.96)	(-0.03, 0.57)
Chick mortality		
Mean	1.87	0.52
95% prediction interval	(-1.76, 5.49)	(-1.23, 2.27)

Puffin

The following model parameters were set for each of the 10 paired runs:

- 100% of total population
- Colonies included: Forth Islands SPA
- Probability of displacement: 60%[9]
- Probability of barrier effect: 100% (all displacement susceptible birds are also barrier-susceptible)[9]
- Buffer around wind farm Development Area[10]: 2km
- Buffer size: 5km
- Mapped bird densities derived from most recent GPS tracking data with birds in flight removed (using speed threshold) and uniform prey availability (since tracking data were only available from one colony).

Table 11D.3.5. Effect sizes (percentage points change in additional mortality) for the Inch Cape Wind Farm alone based on 10 paired runs covering the range of 'moderate' prey conditions for breeding birds based on baseline model output.

	Forth Islands
Adult mortality	
Mean	0.59
95% prediction interval	(0.24, 0.94)
Chick mortality	
Mean	0.39
95% prediction interval	(-0.77, 1.56)

Table 11D.3.6. Effect sizes (percentage points change in additional mortality) for the Inch Cape Wind Farm combined with the other Forth and Tay wind farms based on 10 paired runs covering the range of 'moderate' prey conditions for breeding birds based on baseline model output.

	Forth Islands
Adult mortality	
Mean	1.63
95% prediction interval	(0.74, 2.51)
Chick mortality	
Mean	1.34
95% prediction interval	(-2.48, 5.15)

Common guillemot

The following model parameters were set for each of the 10 paired runs:

- 10% of total population
- Colonies included: Forth Islands SPA, Fowlsheugh SPA, St Abb's Head to Fast Castle SPA, Buchan Ness to Collieston Coast SPA, Farne Islands SPA, Angus coast colonies
- Probability of displacement: 60%[9]
- Probability of barrier effect: 100% (all displacement susceptible birds are also barrier-susceptible)[9]
- Buffer around wind farm Development Area[10]: 2km
- Buffer size: 5km
- Mapped bird densities and prey availability derived from most recent GPS tracking data with birds in flight removed (using speed threshold)

Results are first presented here for Fowlsheugh SPA, St Abb's Head to Fast Castle SPA and Buchan Ness to Collieston Coast SPA based on 10% of the total population. Forth Islands results are then shown based on runs using 50% of the total population.

Table 11D.3.7. Effect sizes (percentage points change in additional mortality) for the Inch Cape Wind Farm alone based on 10 paired runs covering the range of 'moderate' prey conditions for breeding birds based on baseline model output.

	Fowlsheugh	St Abb's Head to Fast Castle	Buchan Ness to Collieston Coast
Adult mortality			
Mean	0.007	-0.005	0.00
95% prediction interval	(-0.07, 0.08)	(-0.02, 0.01)	0.00
Chick mortality			
Mean	0.05	0.00	0.00
95% prediction interval	(-0.07, 0.16)	(-0.04, 0.04)	0.00

Table 11D.3.8. Effect sizes (percentage points change in additional mortality) for the Inch Cape Wind Farm combined with the other Forth and Tay wind farms based on 10 paired runs covering the range of 'moderate' prey conditions for breeding birds based on baseline model output.

	Fowlsheugh	St Abb's Head to Fast Castle	Buchan Ness to Collieston Coast
Adult mortality			
Mean	0.14	0.02	-0.004
95% prediction interval	(-0.005, 0.29)	(-0.08, 0.12)	(-0.04, 0.03)
Chick mortality			
Mean	1.01	0.02	0.09
95% prediction interval	(-0.66, 2.67)	(-0.12, 0.17)	(-0.12, 0.29)

For Forth Islands SPA guillemots, the following model parameters were set for each of the 10 paired runs:

- 50% of total population
- Colonies included: Forth Islands SPA, Fowlsheugh SPA, St Abb's Head to Fast Castle SPA, Buchan Ness to Collieston Coast SPA, Farne Islands SPA, Angus coast colonies
- Probability of displacement: 60%[9]
- Probability of barrier effect: 100% (all displacement susceptible birds are also barrier-susceptible)[9]
- Buffer around wind farm Development Area[10]: 2km
- Buffer size: 5km
- Mapped bird densities and prey availability derived from most recent GPS tracking data with birds in flight removed (using speed threshold)

Table 11D.3.9. Effect sizes (percentage points change in additional mortality) for the Inch Cape Wind Farm alone based on 10 paired runs covering the range of 'moderate' prey conditions for breeding birds based on baseline model output.

	Forth Islands
Adult mortality	
Mean	0.22
95% prediction interval	(0.08, 0.36)
Chick mortality	
Mean	0.62
95% prediction interval	(-0.71, 1.94)

Table 11D.3.10. Effect sizes (percentage points change in additional mortality) for the Inch Cape Wind Farm combined with the other Forth and Tay wind farms based on 10 paired runs covering the range of 'moderate' conditions for breeding birds based on baseline model output.

	Forth Islands
Adult mortality	
Mean	1.42
95% prediction interval	(0.27, 2.57)
Chick mortality	
Mean	5.62
95% prediction interval	(-4.31, 15.55)

Annex References

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

Wakefield, E.D., Owen, E., Baer, J., Daunt, F., Dodd, L.S., Green, J.A., Guildford, T., Mavor, R., Miller, P.I., Newell, M., Newton, S.F., Robertson, G., Shoji, A., Soanes, L.M., Votier, S., Wanless, S. and Bolton, M. (2017) Breeding density, fine scale telemetry and large-scale modelling reveal the regional distribution of a four-species seabird assemblage. *Ecol Appl*, 27: 2074–2091.

Annex 11D.4: Details of barrier type, prey values and seed number used in the different SeabORD models run

Species	Colony	ORDs	Barrier Type	Prey regional median values, g, (lower, upper)	Seed
kittiwake	Forth I	Inch Cape	Perimeter	89-106	19873
kittiwake	Fowlsheugh	Inch Cape	Perimeter	86-106	19873
kittiwake	St Abbs	Inch Cape	Perimeter	87-102	19873
kittiwake	Buchan Ness	Inch Cape	Perimeter	75-86	19873
kittiwake	Forth I	All ORDs	Perimeter	89-106	19873
kittiwake	Fowlsheugh	All ORDs	Perimeter	86-106	19873
kittiwake	St Abbs	All ORDs	Perimeter	87-102	19873
kittiwake	Buchan Ness	All ORDs	Perimeter	75-86	19873
razorbill	Forth I	Inch Cape	Perimeter	196-231	19873
razorbill	Fowlsheugh	Inch Cape	Perimeter	214-257	19873
razorbill	St Abbs	Inch Cape	Perimeter	211-249	19873
razorbill	Forth I	All ORDs	Perimeter	196-231	19873
razorbill	Fowlsheugh	All ORDs	Perimeter	214-257	19873
razorbill	St Abbs	All ORDs	Perimeter	211-249	19873
puffin	Forth I	Inch Cape	Perimeter	191-231	19873
puffin	Forth I	All ORDs	Perimeter	191-231	19873
guillemot	Fowlsheugh	Inch Cape	Perimeter	211-243	19873
guillemot	St Abbs	Inch Cape	Perimeter	219-262	19873
guillemot	Buchan Ness	Inch Cape	Perimeter	165-192	19873
guillemot	Forth I	Inch Cape	Perimeter	227-270	19873
guillemot	Fowlsheugh	All ORDs	Perimeter	211-243	19873
guillemot	St Abbs	All ORDs	Perimeter	219-262	19873
guillemot	Buchan Ness	All ORDs	Perimeter	165-192	19873
guillemot	Forth I	All ORDs	Perimeter	227-270	19873