Beatrice Offshore Windfarm

Environmental Statement Addendum

Annex 6A Bottlenose Dolphin and

Harbour Seal Population Modelling for

Beatrice Offshore Windfarm

Beatrice Offshore Wind Farm Environmental Statement Addendum

Annex 6A: Bottlenose Dolphin and Harbour Seal Population Modelling for Beatrice Offshore Wind Farm

Prepared for Beatrice Offshore Wind Farm Ltd

Professor Paul Thompson, University of Aberdeen

21st May 2013

Bottlenose Dolphin and Harbour Seal Population Modelling for Beatrice Offshore Wind Farm

Background

Following comments on the Beatrice Offshore Wind Farm Environmental Statement (ES) from the regulators and their advisors, additional population modelling was required to assess the long term effects of the Beatrice Offshore Wind Farm development on harbour seals and bottlenose dolphins.

Since submitting the Beatrice Offshore Wind Farm ES, methods for carrying out such assessments on these same populations were developed further within the Moray Offshore Round 3 Zone. Following these developments, additional information was required by Beatrice Offshore Wind Farm Ltd (BOWL) to:

1) Model potential effects of Beatrice Offshore Wind Farm construction upon the bottlenose dolphin population (for comparison with cumulative modelling presented in the Moray Offshore Round 3 Zone ES).
2) Model effects on the harbour seal population using the most recent version of the seal assessment framework and, again, presenting outputs for the Beatrice Offshore Wind Farm construction alone as well as for cumulative effects with Moray Offshore Round 3 Zone.
3) For both species, to explore the effect of varying dose response curves (upper, best and lower fit) for the prediction of the proportion of animals excluded from the area
4) For harbour seals, to explore the effect of changing the animal's probability of survival resulting from Permanent Threshold Shift (PTS) and to explore the effect of changing assumptions about the carrying capacity for the Moray Firth harbour seal population.

Bottlenose Dolphin Population Modelling

The approach used to model construction effects was as for the Moray Offshore Round 3 Zone ES, using VORTEX to model the viability of the East coast bottlenose dolphin population using a Population Viability Analysis (PVA) model previously published in Thompson et al. (2000).

This model was based upon best available demographic and life history values, adjusted to produce, on average, a population that was stable or very slightly increasing. Although detailed demographic and life-history data are not available for this population, this is in line with the latest Site Condition Monitoring report for the Moray Firth Special Area of Conservation (SAC) which highlights that there is a >99% chance that the population is either stable or increasing (Cheney et al. 2012). Input parameters for the baseline scenario are presented in Annex I.

This baseline scenario was run 1000x to provide a distribution of final population sizes after 25 years. Different construction scenarios were then compared with this baseline by running each scenario 100x and presenting the population trajectories and a histogram of final population sizes. Details of the different construction scenarios are given in Table 1.

The number of animals displaced was estimated using the approach developed within the Moray Firth Seal Assessment Framework, as summarised in Annex II, and described in detail within the MORL ES. Noise modelling for BOWL construction was carried out as part of the MORL ES to assess cumulative impacts, and these outputs have been used here to model the impacts of BOWL construction alone. The number of animals with PTS was estimated using SAFESIMM, again, as part of the MORL modelling. In no cases was exposure sufficient to predict that a single individual would experience PTS, so effects of PTS were not modelled for dolphins (see Table 2).

To model the impacts of displacement, it was assumed that displacement would result in a reduction in reproduction (modelled as a harvest of calves), proportional to the proportion of the population that were displaced in each construction year (see Table 2). Calculations were based on there being an average of 4 female and 4 male calves produced in each year from a stable population of 196 bottlenose dolphins. i.e. If 100% of the population were displaced, all 8 calves were harvested the next year. This impact was always worst case, rounding up numbers of calves and always taking more females if there was an odd number of calves. Details of the number of animals harvested in each scenario are presented in Table 3.

The outputs for different construction scenarios are presented in Figure 1 & Table 4. Key differences between the approach used here and that used for the original Beatrice Offshore Wind Farm ES are highlighted in Annex II, an annotated version of the document supplied to BOWL on 29/11/11.

Harbour Seal Population Mmodelling

The approach used to model construction effects was as for the Moray Offshore Round 3 Zone ES, using the modelling framework developed within the Moray Firth Seal Assessment Framework. A description of this process has now been prepared for publication and is included as Annex III.

As for the dolphins, noise modelling for Beatrice Offshore Wind Farm construction was carried out as part of the Moray Offshore Round 3 Zone ES to assess cumulative effects, and these outputs have been used here to model effects of Beatrice Offshore Wind Farm construction alone. The number of animals displaced was estimated using the Seal Assessment Framework, and numbers affected by PTS were estimated using SAFESIMM as this gave the most precautionary value when modelling long term population effects.

The outputs for different construction scenarios are presented in Figure 2. Key differences between the approach used here and that used for the Original Beatrice Offshore Wind

Farm ES are highlighted in Annex II, an annotated version of the document supplied to BOWL on 29/11/11.

References

Cheney, B., Corkrey, R., Quick, N.J., Janik, V.M., Islas-Villanueva, V., Hammond, P.S. & Thompson, P.M. (2012) *Site Condition Monitoring of bottlenose dolphins within the Moray Firth Special Area of Conservation: 2008-2010.* Scottish Natural Heritage Commissioned Report No. 512

Thompson, P.M., Wilson, B., Grellier, K. & Hammond, P.S. (2000) Combining power analysis and population viability analysis to compare traditional and precautionary approaches to the conservation of coastal cetaceans. Conservation Biology, 14(5): 1253-1263.

Table 1. Construction Scenarios

Construction scenario	Description	Corresponding model combination in Table 2
BOWL 1	One piling vessels at location A:Construction starts in 2014 for 3 years	Combination No. 9 for 3 years
BOWL 2	Two piling vessels at location A & B: Construction starts in 2014 for 2 years	Combination No. 8 for 2 years
CUMULATIVE A	BOWL A&B for two years (2014&2015) followed immediately by MORL 1&5 for three years (2016-2018)	Combination No. 8 for 2 years Combination No. 5 for 3 years
CUMULATIVE B	BOWL A for three years (2014-2016) overlapped with MORL 1 five years (2016-2020)	Combination No. 9 for 2 years Combination No. 10 for 1 year Combination No. 4 for 3 years
CUMULATIVE C	BOWL A&B plus MORL 1, 2, 3, 4, 5 & 6 for two years (2016-2017)	Combination No. 11 for 2 years

Table 2. Estimated Numbers of Bottlenose Dolphins Predicted to be Displaced or Having PTS from Piling at Different Locations. N.B. The Combinations of Locations were Modelled to Match the Scenarios in Table 1.

| Bottlenose Dolphin							
Scenario	High		Best fit		Low		PTS
	n	%	n	%	n	%	n
4 (MORL 1)	31	15.7	17	8.9	0	0.2	0.06
5 (MORL 1 & 5)	33	16.8	19	9.7	1	0.3	0.07
6 (MORL 1-6)	36	18.5	21	11.0	1	0.4	0.12
8 (BOWL A+B)	33	17.0	20	10.3	1	0.5	0.11
9 (BOWL A)	32	16.3	19	9.6	1	0.4	0.07
10 (BOWL A + MORL 1)	35	17.8	21	10.7	1	0.4	0.1
11 (BOWL A+B) + (MORL 1-6)	82	41.8	67	34.2	7	3.6	0.23

Table 3. Numbers of Bottlenose Dolphin Calves Harvested in Each of the Combinations Listed in Table 2.

No.	Number of calves harvested					
	High		Best Fit		Low	
	Est	Used	Est	Used	Est	Used
4	1.259	2	0.709	1	0.018	0
5	1.347	2	0.774	1	0.022	0
6	1.478	2	0.879	1	0.031	0
8	1.359	2	0.822	1	0.037	0
9	1.306	2	0.77	1	0.028	0
10	1.421	2	0.854	1	0.034	0
11	3.344	4	2.736	3	0.288	1

Table 4. Median Final Population Size for Each of the Modelled Scenarios. Frequency Histograms for the Full Distribution of Final Population Sizes are Presented in Figure 1.
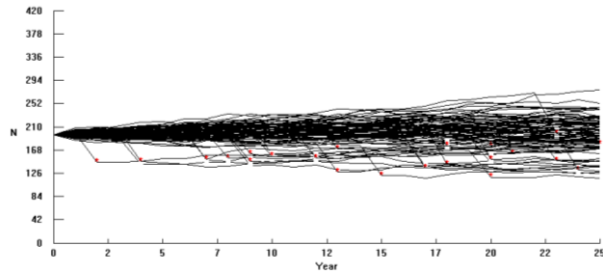
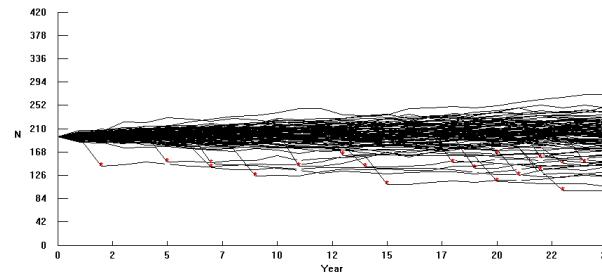|	High	Best fit	Low
BOWL Scenario 1	198	201	198
BOWL Scenario 2	194	197	207
Cumulative Scenario A	193	204	199
Cumulative Scenario B	186	191	201
Cumulative Scenario C	193	195	203.5

Figure 1 – Outputs of Bottlenose Dolphin Population Modelling
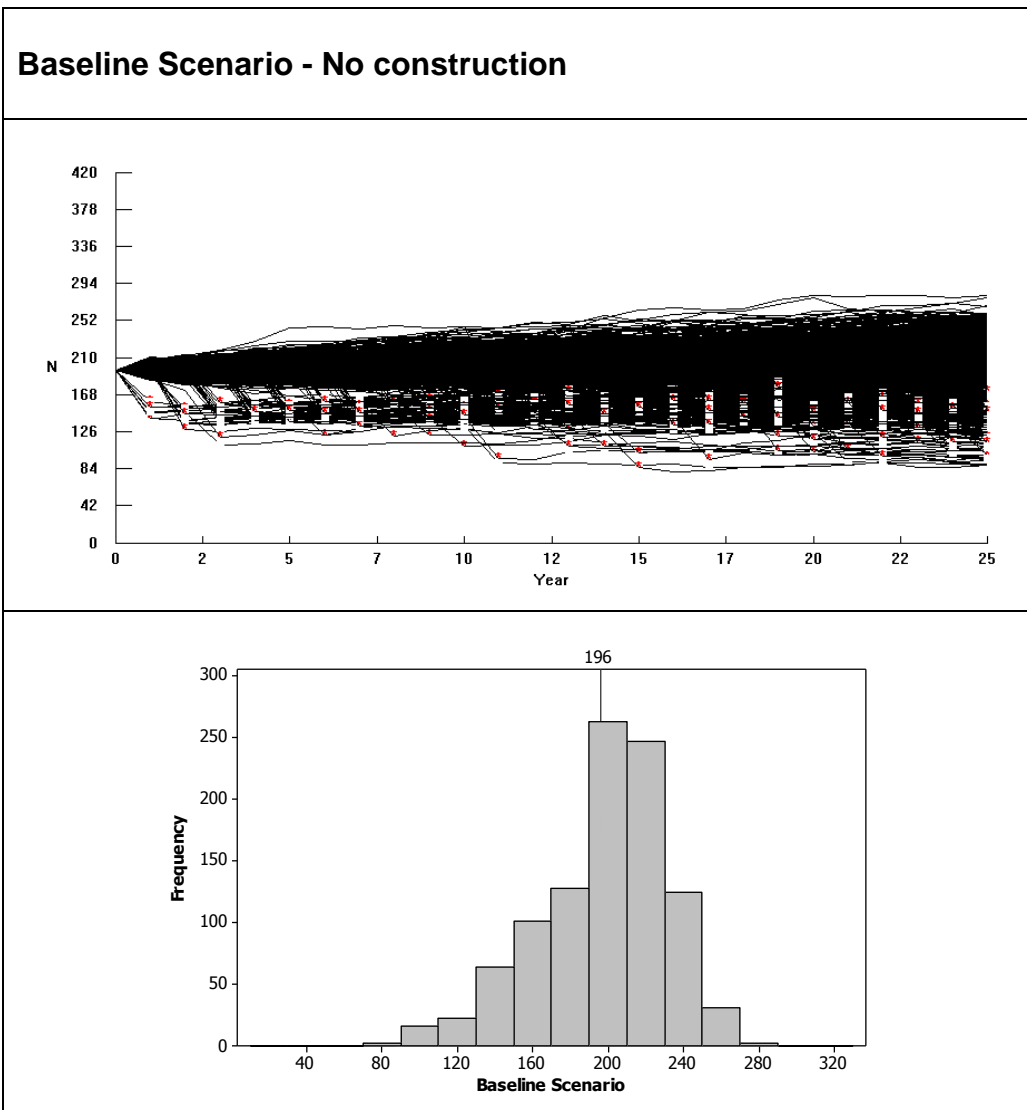
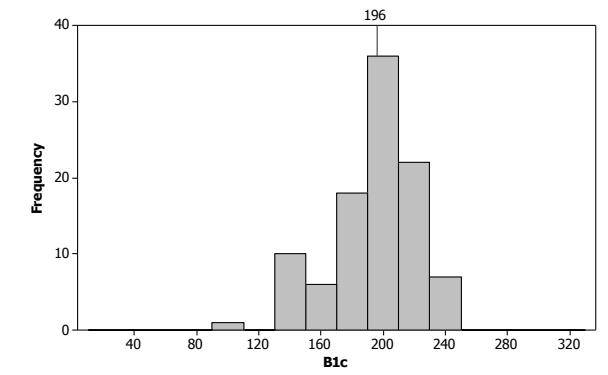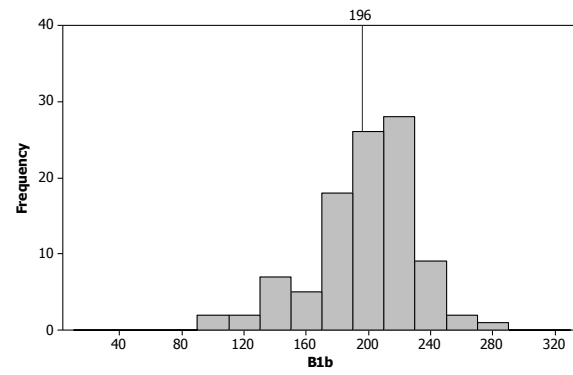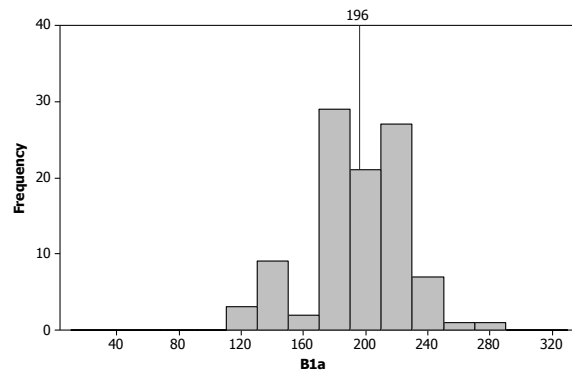BOWL Scenario 1 - BOWL A for three years (2014-2016)

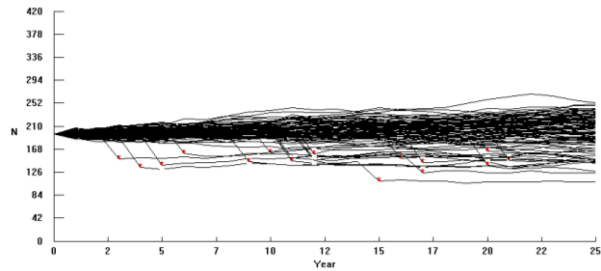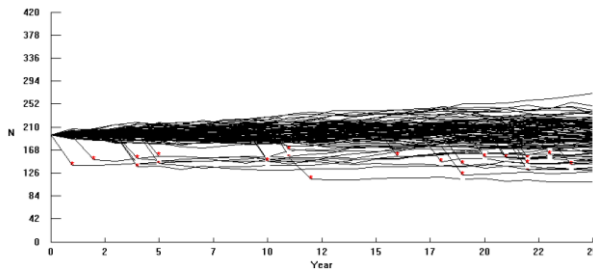| Lower	Best	Upper
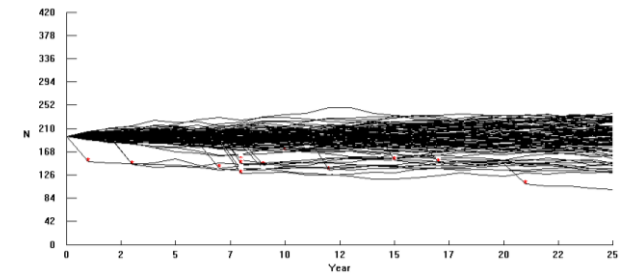
BOWL Scenario 2 - BOWL A&B for two years (2014 & 2015)

| Lower | Best | Upper |
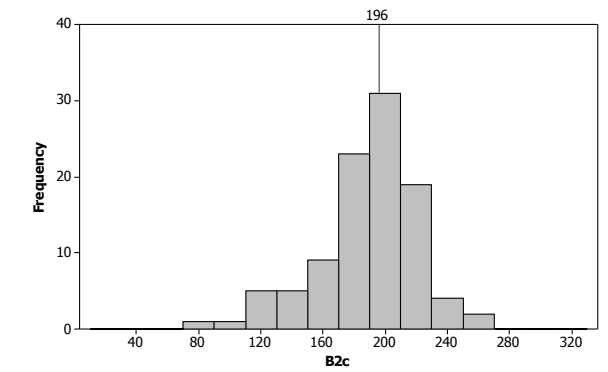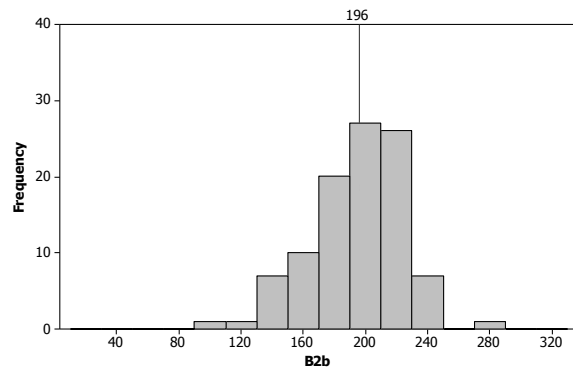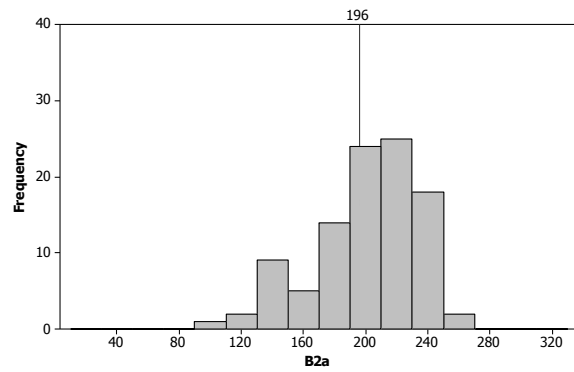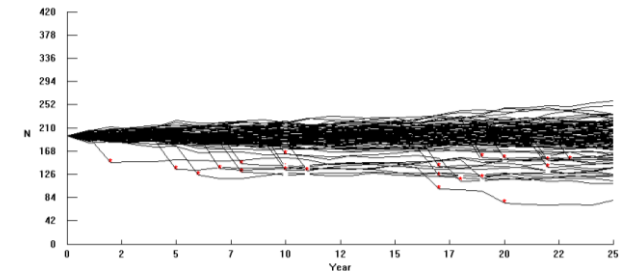
CUMULATIVE Scenario A - BOWL A&B (2014 & 2015) followed immediately by MORL 1&5 (2016-2018)

Lower	Best	Upper
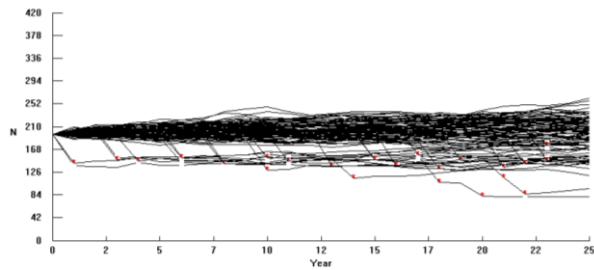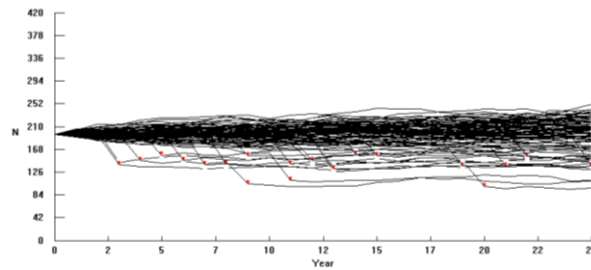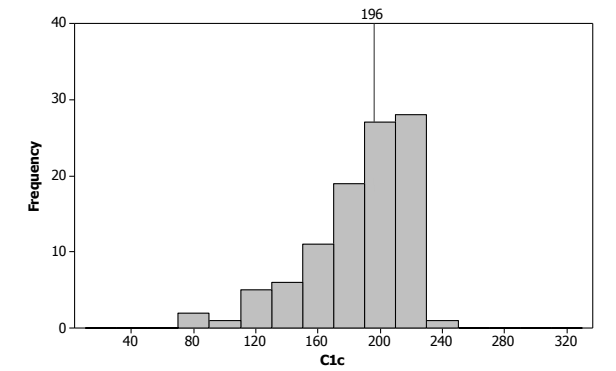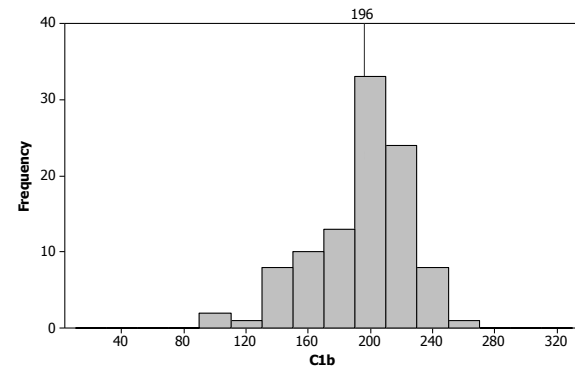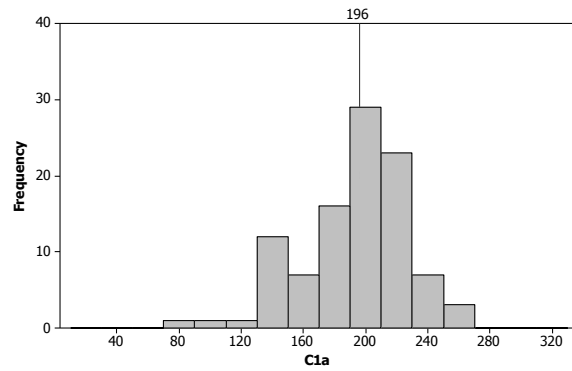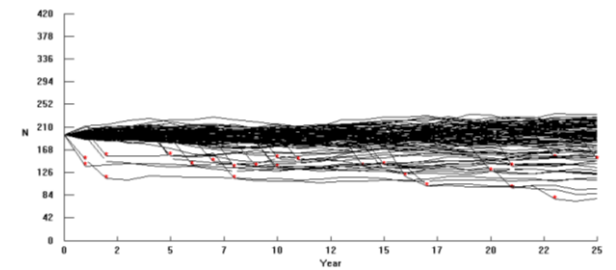
CUMULATIVE Scenario B - BOWL A for three years (2014-2016) + MORL 1 five years (2016-2020)

| Lower | Best | Upper |
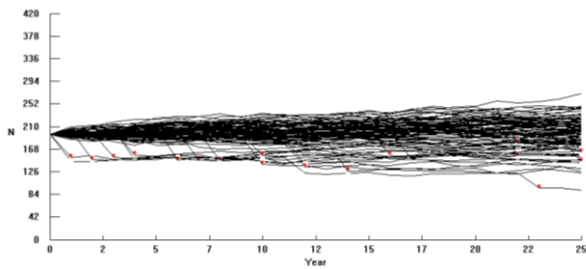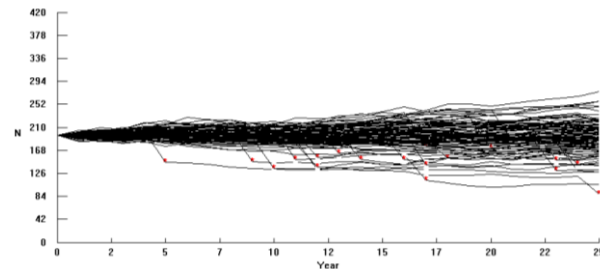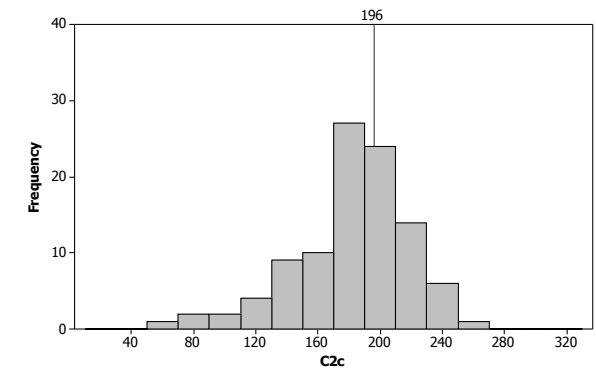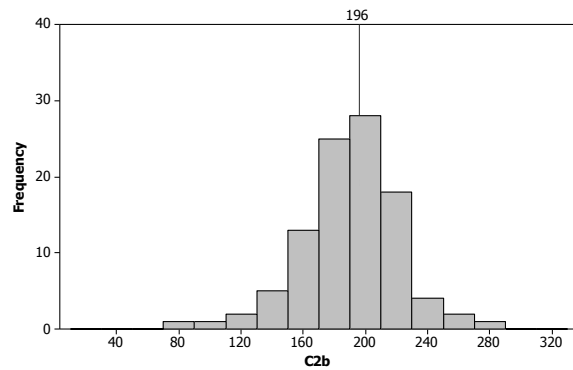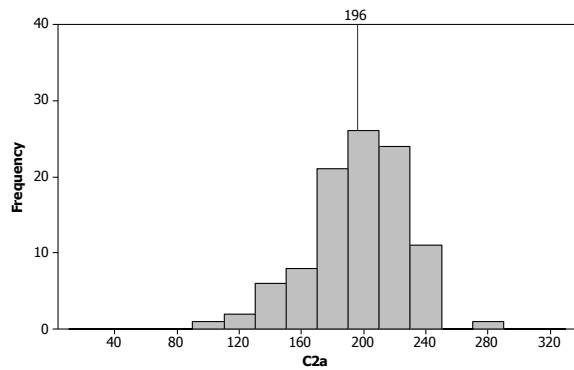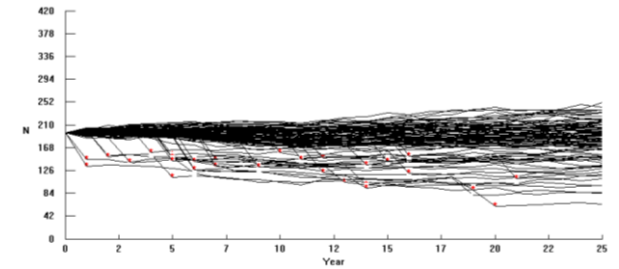
CUMULATIVE Scenario C - BOWL A&B for two years and MORL 1, 2, 3, 4, 5 & 6 for two years (2016-2017)
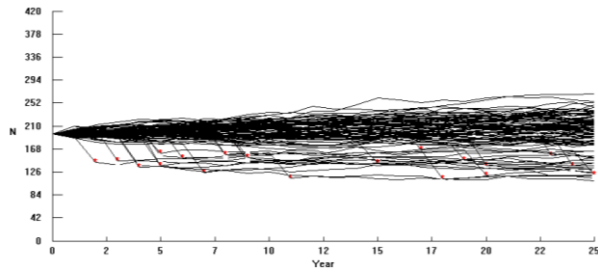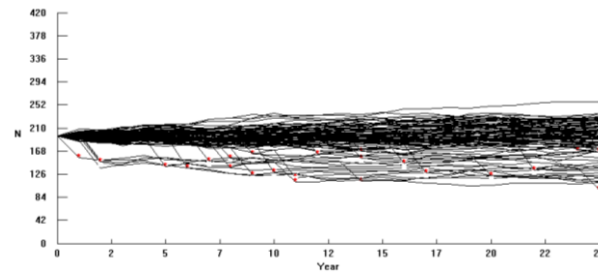
Lower	Best	Upper
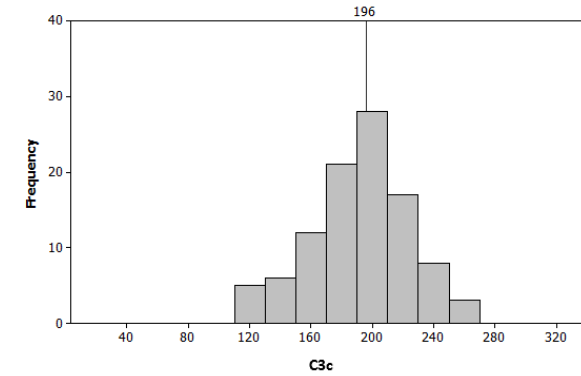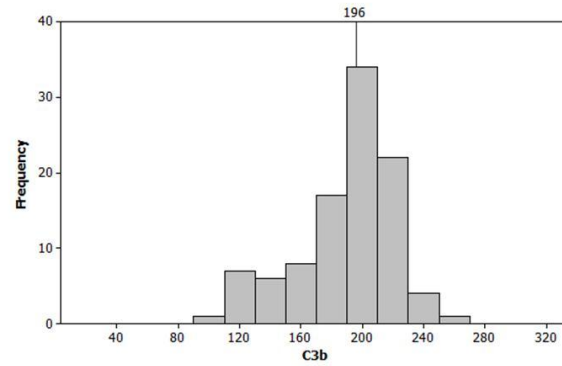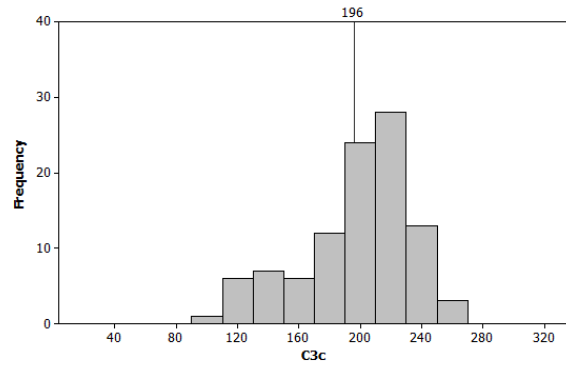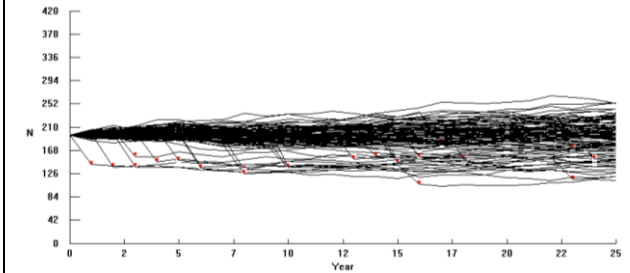
Figure 2 – Outputs of harbour seal modelling

Scenario BOWL 1

BOWL A for three years (2014-2016)

	Carrying Capacity (K) = 1000	Carrying Capacity (K) = 2000
Upper		
Best		
Lower		
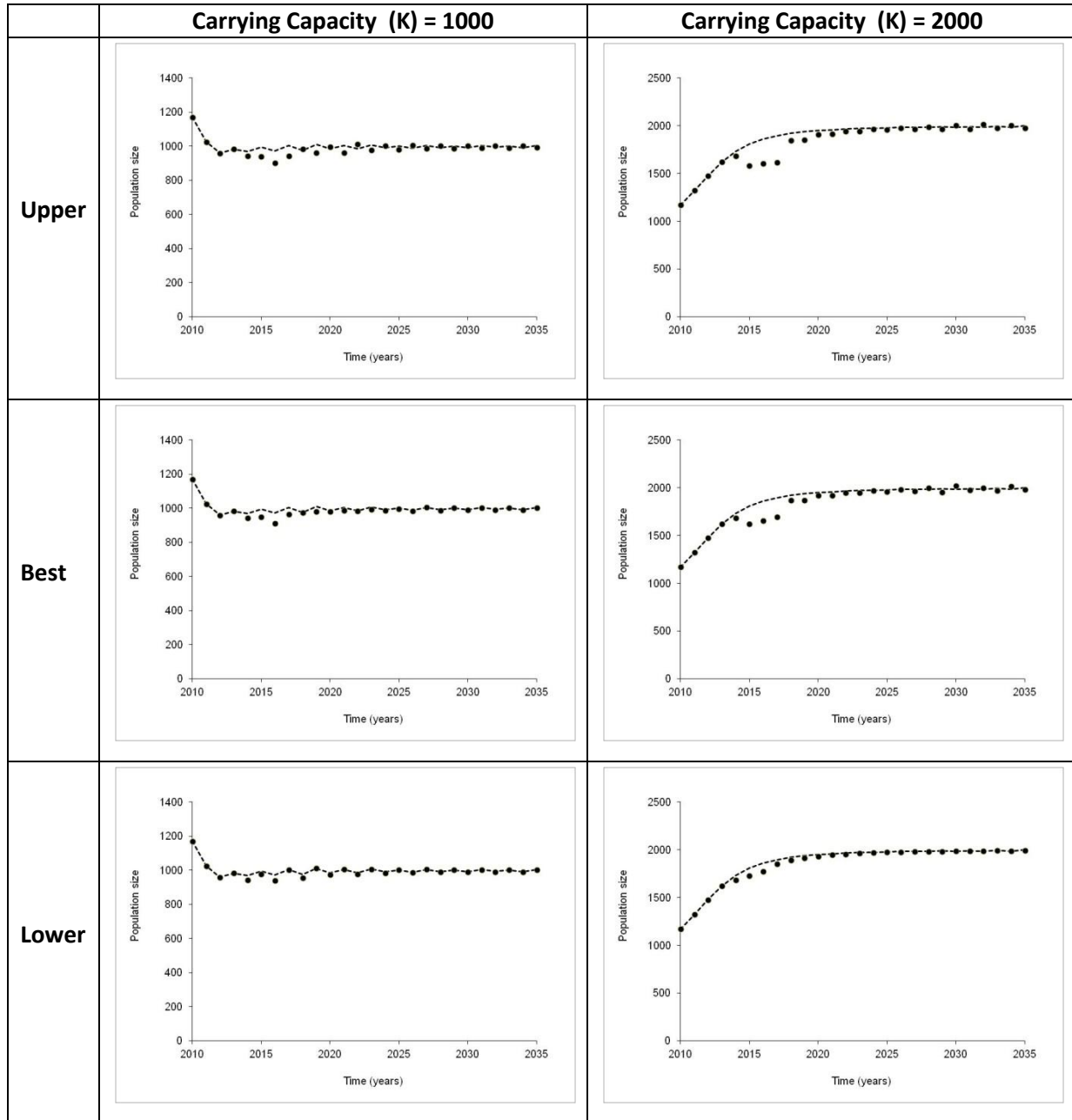
Scenario BOWL 2

BOWL A&B for two years (2014 & 2015)

	Carrying Capacity (K) = 1000	Carrying Capacity (K) = 2000
Upper		
Best		
Lower		
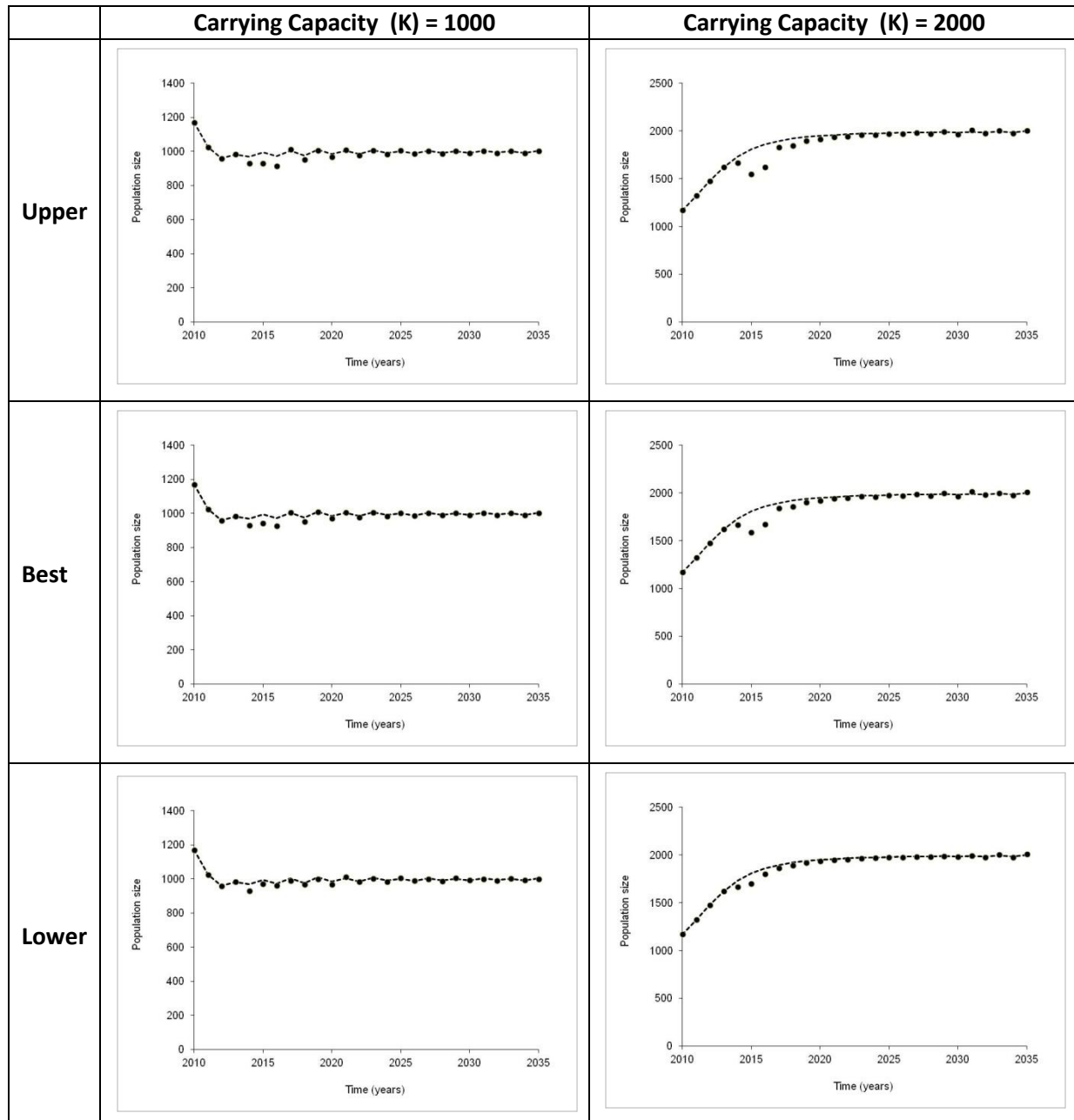
Effect of changing assumption about extent of PTS related mortality

Scenarios based on best estimate for behavioural displacement, K=2000

	BOWL Scenario 1	BOWL Scenario 2
30%		
20%		
10%		
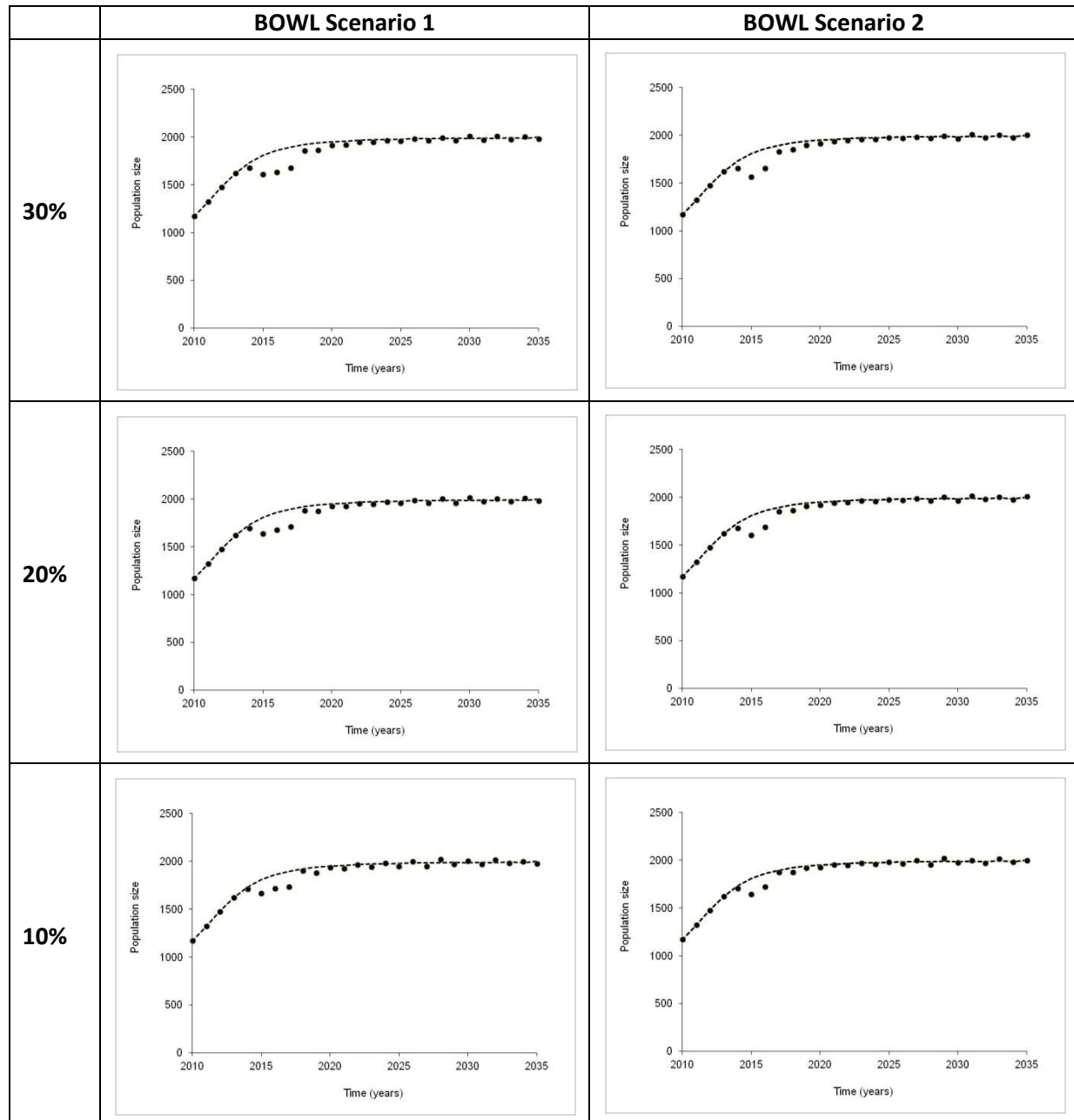
Scenario CUMULATIVE A

BOWL A&B for two years (2014 & 2015) followed immediately by MORL 1&5 for three years (2016-2018)

	Carrying Capacity (K) = 1000	Carrying Capacity (K) = 2000
Upper		
Best		
Lower		

CUMULATIVE B

BOWL A for three years (2014-2016) overlapped with MORL 1 five years (2016-2020)

	Carrying Capacity (K) = 1000	Carrying Capacity (K) = 2000
Upper		
Best		
Lower		
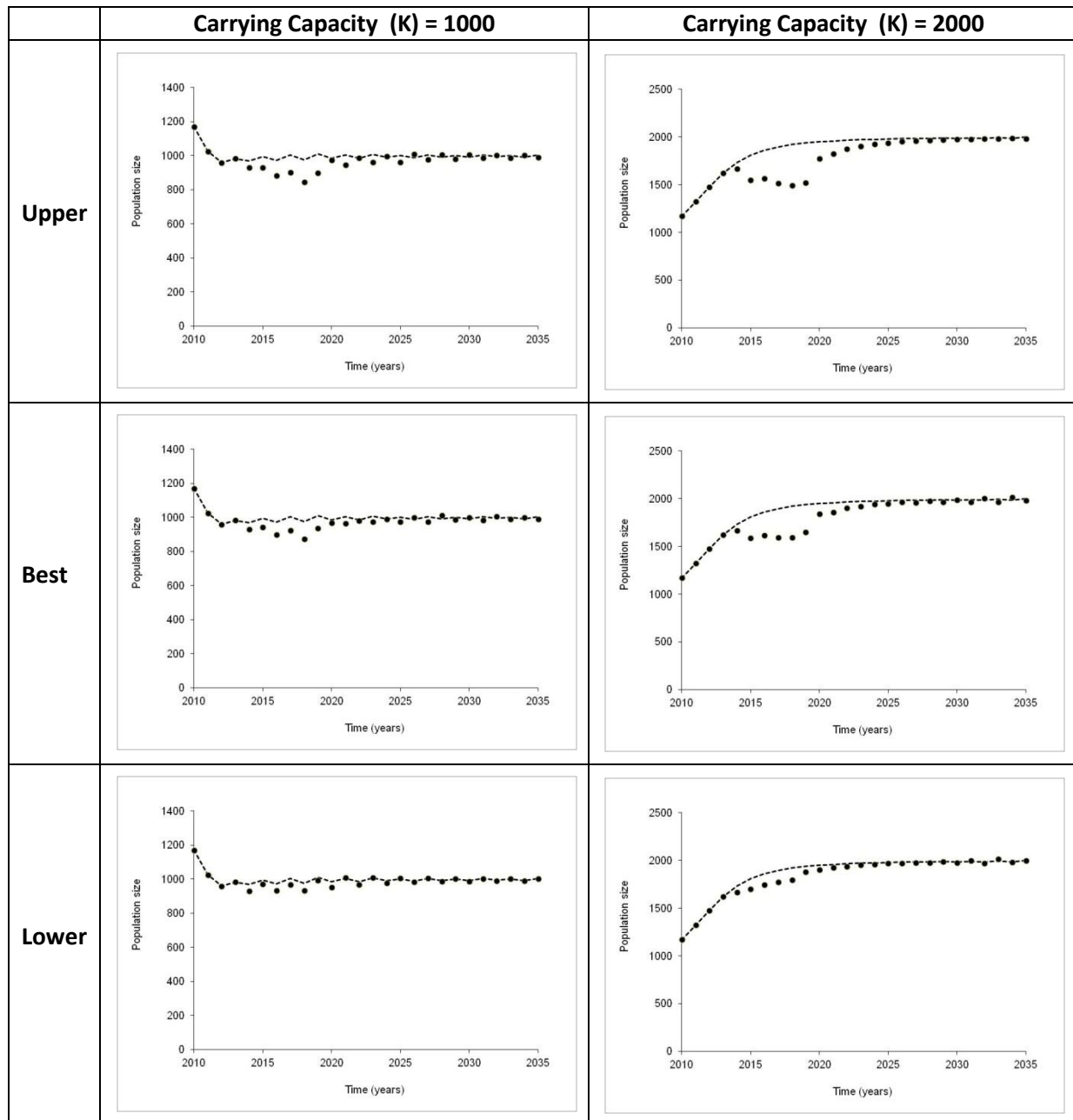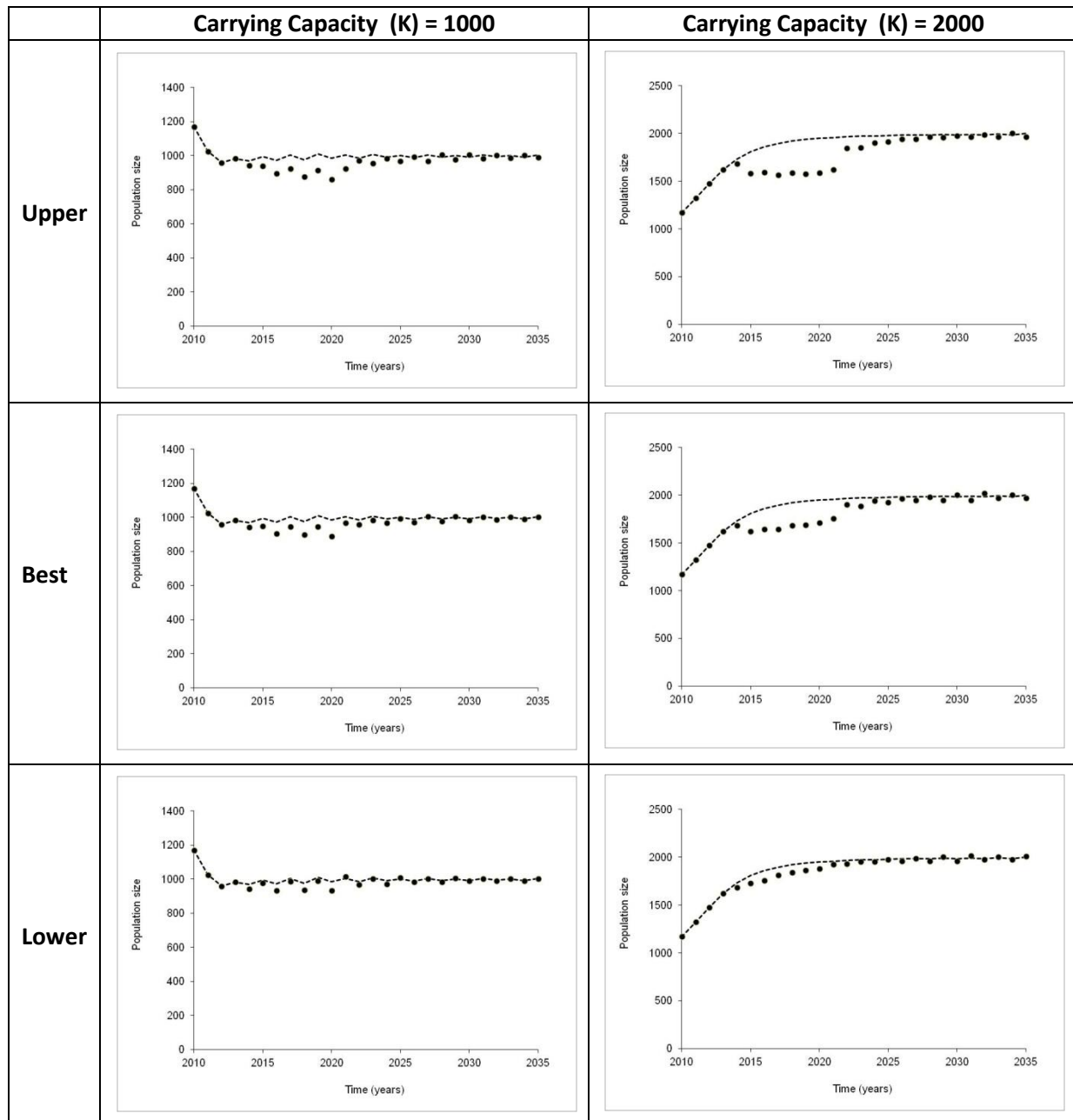
CUMULATIVE C

BOWL A&B and MORL 1, 2, 3, 4, 5 & 6 for two years (2016-2017).

Nb. As in the MORL ES, SAFESIMM estimates from combination 6 and 8 in Table 2 were added together, which represents a degree of double counting and additional conservatism.

	Carrying Capacity (K) = 1000	Carrying Capacity (K) = 2000
Upper		
Best		
Lower		
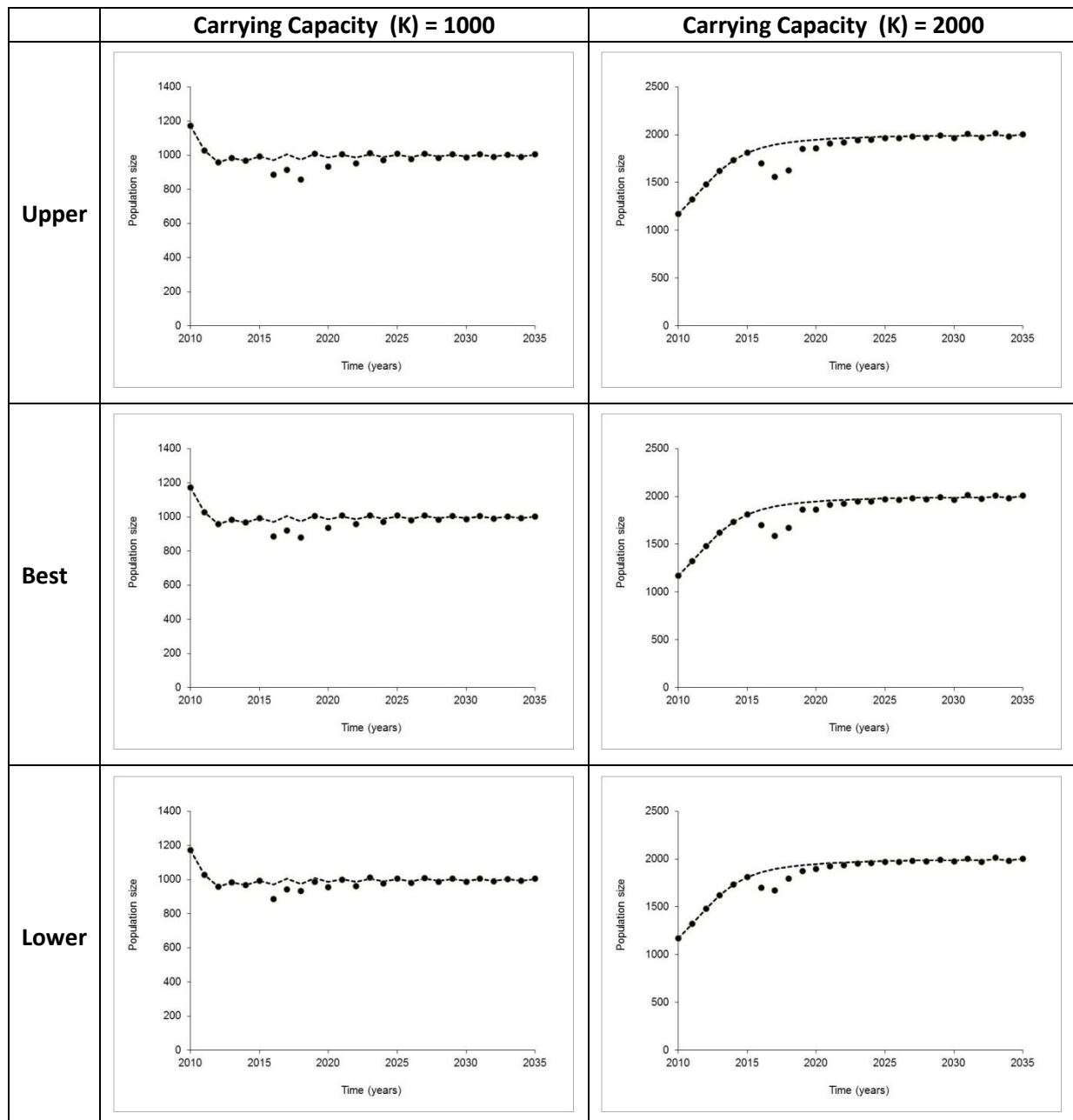
Annex I. Vortex Input Parameters for the Baseline Bottlenose Dolphin Population Model.

```
VORTEX 9.99 -- simulation of population dynamics

Baseline_A


  1 population(s) simulated for 25 years, 1000 iterations
  Each simulation year is 365 days duration.

  Extinction is defined as no animals of one or both sexes.

  Inbreeding depression modeled with 3.14000 lethal equivalents per individual,
    comprised of 1.57000 recessive lethal alleles,
    and 1.57000 lethal equivalents not subject to removal by selection.

  EV in mortality will be concordant among age-sex classes
    but independent from EV in reproduction.

  First age of reproduction for females: 10   for males: 11
  Maximum breeding age (senescence): 50
  Sex ratio at birth (percent males): 50


Population 1: Population 1

  Polygynous mating;
    % of adult males in the breeding pool = 75

  % adult females breeding = 14.4
   EV in % adult females breeding: SD = 2.44

  Distribution of number of separately sired broods produced by a female in a year
...
      0.00 percent of females produce 0 broods (litters, clutches) in an average
year
    100.00 percent of females produce 1 broods (litters, clutches) in an average
year

   Of those females producing progeny, ...
   Mean number of progeny per breeding female per year = 1
   SD in number of progeny = 0

   % mortality of females between ages 0 and 1 = 20
    EV in % mortality: SD = 7
   % mortality of females between ages 1 and 2 = 2.85
    EV in % mortality: SD = .5
   % mortality of females between ages 2 and 3 = 2.85
    EV in % mortality: SD = .5
   % mortality of females between ages 3 and 4 = 2.85
    EV in % mortality: SD = .5
   % mortality of females between ages 4 and 5 = 2.85
    EV in % mortality: SD = .5
   % mortality of females between ages 5 and 6 = 2.85
    EV in % mortality: SD = .5
   % mortality of females between ages 6 and 7 = 2.85
    EV in % mortality: SD = .5
   % mortality of females between ages 7 and 8 = 2.85
    EV in % mortality: SD = .5
   % mortality of females between ages 8 and 9 = 2.85
    EV in % mortality: SD = .5
   % mortality of females between ages 9 and 10 = 2.85
    EV in % mortality: SD = .5
   % mortality of adult females (10<=age<=50) = 2.85
    EV in % mortality: SD = .5
   % mortality of males between ages 0 and 1 = 20
    EV in % mortality: SD = 7
```

```
 % mortality of males between ages 1 and 2 = 2.85
  EV in % mortality: SD = .5
 % mortality of males between ages 2 and 3 = 2.85
  EV in % mortality: SD = .5
 % mortality of males between ages 3 and 4 = 2.85
  EV in % mortality: SD = .5
 % mortality of males between ages 4 and 5 = 2.85
  EV in % mortality: SD = .5
 % mortality of males between ages 5 and 6 = 2.85
  EV in % mortality: SD = .5
 % mortality of males between ages 6 and 7 = 2.85
  EV in % mortality: SD = .5
 % mortality of males between ages 7 and 8 = 2.85
  EV in % mortality: SD = .5
 % mortality of males between ages 8 and 9 = 2.85
  EV in % mortality: SD = .5
 % mortality of males between ages 9 and 10 = 2.85
  EV in % mortality: SD = 0.5
 % mortality of males between ages 10 and 11 = 2.85
  EV in % mortality: SD = .5
 % mortality of adult males (11<=age<=50) = 2.85
  EV in % mortality: SD = .5


  EVs may be adjusted to closest values possible for binomial distribution.

  Catastrophe type 1: 1
    Frequency (as a percent): 1
    Multiplicative effect on reproduction = 0.5
    Multiplicative effect on survival = .75

  Initial size of Population 1:      196
    (set to reflect stable age distribution)
 Age 1     2     3     4     5     6     7     8     9    10    11    12    13
14    15    16    17    18    19    20    21    22    23    24    25    26    27
28    29    30    31    32    33    34    35    36    37    38    39    40    41
42    43    44    45    46    47    48    49    50    Total
      4     4     3     4     3     3     4     3     3     3     2     3     3
2     3     2     2     3     2     2     2     2     2     2     1     2     2
2     1     2     1     2     1     1     2     1     1     1     1     2     1
1     1     1     1     1     0     1     1     1     98    Males
      4     4     3     4     3     3     4     3     3     3     2     3     3
2     3     2     2     3     2     2     2     2     2     2     1     2     2
2     1     2     1     2     1     1     2     1     1     1     1     2     1
1     1     1     1     1     0     1     1     1      98   Females

  Carrying capacity = 400
  EV in Carrying capacity = 10
```

Annex II. Outline of Changes in the Assessment Process in this Document Compared with the Original BOWL ES.

MORL BOWL Marine Mammal Noise Assessments

Revised outputs, October 2012

This document is based on the document supplied to BOWL on 29/11/11 that outlined the approach used to assess impacts of construction noise on marine mammals in the original ES.

Several methodological developments were made prior to the application of this approach in the MORL ES. In the description below, all such changes are described in UPPERCASE text. Points made in lower case remain unchanged.

1. *Harbour Seal Model.*

- provides comparison with a baseline trend based upon best estimates of Loch Fleet demographic rates
- current PBR take is included in baseline model and all pile-driving scenarios
- uses the predicted at-sea distribution based upon telemetry data, but assumes that 25% of animals are hauled out when assessing SEL and PTS.
- ESTIMATES OF THE NUMBER OF ANIMALS WITH PTS ARE BASED ON THE SAFESIMM MODEL, AS DESCRIBED IN DETAIL IN THE MORL ES. WE NOW ALSO USE THE 186 DB THRESHOLD, AS PROPOSED BY SOUTHALL ET AL (2007). THIS APPROACH HAS BEEN CHOSEN TO PROVIDE THE MOST CONSERVATIVE AND PRECAUTIONARY ESTIMATES RATHER THAN THE MOST LIKELY ESTIMATES.
- we assume that 25% of those animals with PTS die in the year of exposure.
- we assume that all females that are displaced do not breed, or produce pups that die (implemented by increasing pup mortality).

2. **Bottlenose dolphin model**

- WE USE AN UPDATED PREDICTION OF BOTTLENOSE DOLPHIN DENSITY, AS USED IN THE MORL ES, USING A SLIGHT MODIFICATION OF THE MODEL DESCRIBED IN THE ORIGINAL DOCUMENTATION TO BOWL on 11/11/11.
- ESTIMATES OF THE NUMBER OF ANIMALS WITH PTS ARE BASED ON THE SAFESIMM MODEL
- We then use the Horn's Rev II displacement curve and db$_{ht}$ noise contours to assess the probability of animals being displace from each grid square on any particular day.
- AS FOR HARBOUR SEALS, AND AS USED IN THE MORL ES, WE THEN EXPLORE THE LONG-TERM IMPLICATIONS OF ANIMALS BEING DISPLACED OR SUFFERENING PTS USING A STOCHASTIC POPULATION MODEL.
- IN THIS MODEL, WE ASSUME THAT ALL FEMALES THAT ARE DISPLACED DO NOT BREED, OR PRODUCE CALVES THAT DIE (IMPLEMENTED BY "HARVESTING" CALVES FROM THE POPULATION).

Framework for assessing impacts of pile-driving noise from offshore wind farm construction on a harbour seal population

Paul M. Thompson [a], Gordon D. Hastie [b], Jeremy Nedwell [c], Richard Barham [c], Kate L. Brookes [a], Line S. Cordes [a], Helen Bailey [d] and Nancy McLean [e]

a. University of Aberdeen, Institute of Biological and Environmental Sciences, Lighthouse Field Station, Cromarty, IV11 8YL, United Kingdom.
b. Scottish Oceans Institute, SMRU Limited, New Technology Centre, North Haugh, St Andrews, KY16 9RS, United Kingdom.
c. Subacoustech Environmental Ltd., Unit 9, Claylands Road, Bishops Waltham, Southampton, Hampshire, SO32 1QD, United Kingdom.
d. Chesapeake Biological Laboratory, University of Maryland Center for Environmental Science, Solomons, MD 20688, United States of America.
e. Natural Power Consultants, The Green House, Forrest Estate, Dalry, Castle Douglas, DG7 3XS, United Kingdom.

Abstract

Offshore wind farm developments may impact protected marine mammal populations, requiring appropriate assessment under the EU Habitats Directive. We describe a framework developed to assess population level impacts of disturbance from piling noise on a protected harbour seal population in the vicinity of proposed wind farm developments in NE Scotland. Spatial patterns of seal distribution and received noise levels are integrated with available data on the potential impacts of noise to predict how many individuals are displaced or experience auditory injury. Expert judgement is used to link these impacts to changes in vital rates and applied to population models that compare population changes under baseline and construction scenarios over a 25 year period. We use published data and hypothetical piling scenarios to illustrate how the assessment framework has been used to support environmental assessments, explore the sensitivity of the framework to key assumptions, and discuss its potential application to other populations of marine mammals.

Acknowledgements

1. Introduction

Growth in offshore wind generation is anticipated to play a major role in meeting the European carbon reduction targets that have been developed to mitigate potential impacts of climate change (Jay, 2011; Toke, 2011). In the North Sea, many proposed wind farm sites are on submerged offshore sandbanks, which also provide important habitats for marine mammals and seabirds. Previously, attention has focussed on the potential impacts of these developments upon bird populations (eg. Drewitt and Langston, 2006; Masden et al., 2010); however, several developments are in the vicinity of Special Areas of Conservation (SAC) that have been established to protect populations of marine mammals, such as harbour seals (*Phoca vitulina*) and bottlenose dolphins (*Tursiops truncatus*), under the EU Habitats Directive (92/43/EEC). Where developments have the potential to impact upon these species, Appropriate Assessments (AA) are required to establish that there will be no long-term impact on the integrity of these protected populations (Söderman, 2009).

There are three key potential impacts of offshore wind farm construction upon marine mammal populations. First, direct impacts of piling noise or other activities during the construction phase potentially causing direct injury or eliciting behavioural responses that could lead to displacement (Bailey et al., 2010; Brandt et al., 2011). Second, indirect impacts through long-term alteration of habitat that may, in turn, be either negative (through loss of habitat) or positive (though reef effects or changes in fishing activity) (Inger et al., 2009; Scheidat et al., 2011). Finally, disturbance or barrier effects resulting from operational turbines or maintenance vessels (Tougaard et al., 2009) may lead to displacement from areas or changes in movements. Given the high sound source levels resulting from pile-driving, the potential impacts that have been of greatest concern to stakeholders are the direct and indirect impacts of noise during construction (Dolman and Simmonds, 2010).

To obtain project consent, developers must provide information that allows regulators to conduct an AA to determine whether the proposal is likely to have a significant effect on the SAC's conservation objectives. Critically, this requires an assessment of whether the development may lead to long-term population change that would compromise the Favourable Conservation Status (FCS) of the wider population. However, whilst there is growing understanding that anthropogenic noise may affect individual behaviour in marine mammals (Southall et al., 2007; Carstensen et al., 2006; Brandt et al., 2011), the lack of guidance on how developers should assess the population consequences of disturbance from construction activities threatens to delay consenting decisions and efforts to meet 2020 carbon reduction targets.

To address this gap, we developed a framework for assessing population-level impacts of proposed wind farm construction on protected harbour seals using the Dornoch Firth and Morrich More SAC, within the Moray Firth, NE Scotland. The Moray Offshore Renewables Ltd (MORL) project comprises of three wind farms, with a combined output of 1.5 GW, located a minimum of 12 Nm from shore on the Smith Bank, leased under the Crown Estates Round 3 programme. The Beatrice Offshore Wind Limited (BOWL) project is a 1 GW development located adjacent to the MORL project within the 12 Nm limit, leased under the Crown Estates Scottish Territorial Waters (STW) programme. Construction of the projects is proposed to commence in 2015/16, which would allow both projects to be fully commissioned by 2020.

This paper provides an overview of the assessment framework that was developed to explore the long-term impact of different construction scenarios. Here, we illustrate the framework using pile-driving data collected in the Moray Firth during the

installation of the two 5 MW Beatrice Demonstrator turbines (Bailey et al., 2010), and scenarios involving construction of a hypothetical wind farm at this site using analogous construction techniques. The development of this framework benefitted from a long history of research on the Moray Firth harbour seal population and information gained during the Beatrice Demonstrator Project. We therefore conclude by discussing how the framework can be developed to incorporate new data sources, and explore its potential use for other harbour seal populations and marine mammal species.

2. General approach

Our general approach for assessing a development's impact on the SAC conservation objectives and the population's FCS involved three main elements (Figure 1). First, available data and existing modelling frameworks were used to describe the spatial distribution of both harbour seals and pile-driving noise. Second, these data were integrated with available data on the potential impacts of noise to assess the numbers of individuals impacted. Finally, impacts on individuals were translated into changes in vital rates (fecundity and survival), and applied to a population model to predict longer-term population level impacts (considered here over a 25-year time scale).

Whilst these first two elements are routinely used in environmental assessments, uncertainty over the links between individual impacts and changes in vital rates has previously constrained efforts to model long-term population change. Frameworks for understanding population consequences of acoustic disturbance are being developed in response to the recommendations of a National Research Council Committee (NRC, 2003). While these approaches show great promise, empirical data to inform these links will not become available within the timescales required for consenting current rounds of offshore wind farms within the UK. Because AA of FCS must consider whether or not protected populations are maintaining themselves in the long-term, it was essential to develop an alternative transparent way of linking predicted individual impacts to vital rates. We therefore based these links upon expert judgement, while ensuring that the sensitivity of our population models to these assumptions could be fully explored. Similarly, the population modelling framework was required to permit exploration of potential interactions with other cumulative impacts (such as persecution or by-catch), and comparison of different development or mitigation scenarios. Crucially, while this initial framework was based upon the best available scientific data, it was designed to ensure that key parameters or relationships could be updated once new data become available.

3. Components of the assessment framework

3.1 Characterising Seal Distribution

This element of the framework (see Figure 1) requires information on spatial variation in the density of harbour seals across the region. Estimates of regional population size were available from annual counts at harbour seal breeding sites (SCOS, 2011), which were inflated to estimate the total number of individuals within the population (Thompson et al., 1997). In 2010, the mean haul-out count for the inner Moray Firth was 721, which represented a total population size of 1,183 (95% CI = 1027-1329).

Information on the foraging distribution of seals was based upon tracking data from 37 individual seals, collected during a series of studies that were carried out in the area between 1989 and 2009 (Thompson et al., 1997; Cordes et al., 2011; Sharples

et al., 2012). These data were used to model seal occurrence and habitat preference using a generalised additive model (GAM) to specify presence-absence across a 4x4 km grid, which revealed a significant relationship between seal presence and water depth, slope and distance to the nearest haul-out site. The results of this GAM were then used to predict the probability of seal occurrence in each of the 4x4 km cells across the Moray Firth. The percentage of the regional population in each cell within the Moray Firth was estimated by dispersing the whole population to produce a density surface in relation to the predicted importance of each cell (Figure 2).

3.2 Characterising Noise Distribution

The predicted propagation of noise resulting from the piling operations required to install the wind turbine foundations was modelled using INSPIRE; a noise propagation model developed by Subacoustech Ltd. that has been widely used in environmental assessments for both renewable energy and oil and gas developments. This model uses a combined geometric and energy flow/hysteresis loss model suitable for pulsed noise such as impact piling to predict propagation in this shallow coastal environment. Comparison of INSPIRE model predictions with measured recordings from the Beatrice Demonstrator (Bailey et al., 2010) indicated that the model predictions for unweighted peak levels provide a relatively good fit of the measured data across the wider Moray Firth (Figure 3).

Received sound levels were frequency weighted to account for the characteristics of harbour seal hearing. As detailed in section 3.3, two different weightings were required. First, weighted sounds pressure levels (dB_{ht} (species)) (also termed "sensation level" (Yost, 2000)) were calculated based upon published data on the harbour seal audiogram (see Nedwell et al., 2007). Second, M-weighted sound exposure levels (SELs) were calculated based upon the approach proposed for all pinnipeds in water by Southall et al. (2007). Spatial variation in received sound levels was expressed as a series of contours representing the point within which a particular threshold (e.g. 90 dB_{ht} (species) or an M-weighted SEL of 186 dB re 1 $\mu Pa^2.s^{-1}$) was exceeded. When estimating M-weighted SEL contours, it was assumed that animals would flee from the sound source rather than remaining stationary.

Outputs were generated as GIS shape files and used within ArcGIS to assess the maximum received sound levels in each of the 4x4 km grid cells for which there were predictions of seal density (Figure 2). An example showing the resulting dB_{ht} (harbour seal) values from INSPIRE for each grid cell is shown in Figure 4.

3.3 Assessing impacts upon individual seals

The potential impacts of noise on marine mammals fall into three major categories; non-auditory injury, auditory injury, and behavioural (Richardson et al., 1995; Southall et al., 2007). These can each be further sub-divided depending upon the severity of the effect, as summarised in Table 1.

Traumatic non-auditory injury from loud sound sources at very close range is relatively well understood, and guidelines have been developed to mitigate against these risks (Southall et al., 2007; Dolman et al., 2009). While there is general agreement on this effect of sound, there is much more uncertainty over the mechanisms and received levels at which auditory injury and behavioural responses may occur. Drawing on the findings of a series of inter-disciplinary expert review groups, Southall et al. (2007) used available data to identify precautionary noise exposure criteria, weighting frequencies accordingly for different functional groups of

marine mammals (M-weightings). Developed initially to support implementation of the US Marine Mammal Protection Act, many stakeholders now see Southall et al.'s (2007) interim criteria as the benchmark for environmental assessments in other parts of the world. However, whilst this can help identify noise levels that might cause injury, data on behavioural responses are so limited that Southall et al. (2007) only developed exposure criteria for behavioural responses to single pulses of sound. Given that a key concern during wind farm construction is behavioural disturbance from extended periods of pile-driving, regulators within the EU clearly cannot base their AA solely upon Southall et al.'s (2007) criteria.

An alternative approach for assessing the impacts of anthropogenic noise, which focuses on behavioural responses, is the use of dB $_{ht}$ (species) values as described by Nedwell et al. (2007). This approach builds upon standard procedures for assessing impacts of industrial noise upon humans, and uses information on each species' hearing ability to provide species-specific frequency weightings. This allows an assessment of the "perceived loudness" of the sound to the animal. Similarly, cognitive studies of marine mammals have estimated "sensation levels" that represent received levels, frequency-weighted according to the study species' hearing ability (e.g. Yost, 2000; Götz and Janik, 2010). Drawing on public domain information and experimental evidence from fish, Nedwell et al. (2007) suggest that animals will show strong avoidance reactions to levels at and above 90 dB $_{ht}$ (species) and milder reactions to levels of 75 dB $_{ht}$ (species) and above. However, these behavioural response criteria remain untested for marine mammals, and it is also recognised that behavioural responses are likely to be context specific (Ellison et al., 2012).

Given that there are uncertainties surrounding both Southall et al.'s (2007) M-weighted criteria and Nedwell et al.'s (2007) dB $_{ht}$ (species) criteria, our approach has been to estimate received levels using both metrics, and select the most appropriate metrics to assess different types of impact at the individual animal level. As described in more detail below, we use dB $_{ht}$ (species) criteria to predict how many individuals will be displaced due to a behavioural reaction, and M-weighted criteria to predict how many individuals will be exposed to permanent threshold shift (PTS).

3.3.1 Risk of behavioural displacement.

Behavioural responses may occur at many levels (see Table 4 in Southall et al., 2007). Here, we focus on behavioural responses that are likely to result in displacement from impacted areas, as we assume that lower levels of response will have only very weak links with vital rates. While this may underestimate impacts from more subtle behavioural changes, this should be balanced by our conservative assumptions about both the time that it takes animals to return to impacted areas and the consequences of behavioural avoidance (see section 3.4).

Quantifying the levels of displacement is constrained by the absence of data on behavioural responses of harbour seals to known levels of multiple pulsed noise such as piling. However, Brandt et al. (2011) do provide data on changes in the occurrence of harbour porpoises at different distances from a piling event at Horns Rev II in Danish waters. These data were collected using moored echolocation detectors (C-PODs), and represent the difference between a baseline period and data collected during the hour after piling. We used these data from harbour porpoises as a proxy for harbour seals, and modelled the extent of the proportional change with distance from source by fitting a binomial relationship to the data (Figure 5a). We then took published data on the size of the pile, together with information on

local bathymetry, and used INSPIRE to estimate received noise levels (using dB $_{ht}$ (harbour porpoise) as a metric) at each of the C-POD sampling sites at Horns Rev II. The relationships from Figure 5a were then used to predict the proportion of animals exhibiting a response at different received noise levels using dB $_{ht}$ (harbour porpoise) (Figure 5b). Although the number of data points used is small, the fitted relationship in Figure 5b generally supports the definitions of the threshold values proposed by Nedwell et al. (2007). In the absence of similar empirical data for harbour seals, we assume that this relationship holds for similar values of dB $_{ht}$ (harbour seal), and use the relationship in Figure 5b to predict the level of displacement of seals in each 4x4 km grid cell in relation to predicted noise levels in that square.

3.3.2 Risk of Auditory Damage

Southall et al. (2007) provide interim noise exposure criteria for levels at which PTS becomes increasingly likely for the different functional groups of marine mammals. Given that one cannot experimentally induce PTS for ethical reasons, these noise exposure criteria for PTS-onset are conservative, and based upon assumed relationships between the relative levels of noise likely to cause Temporary Threshold Shifts (TTS) and PTS which, in turn, involves the use of proxy data from humans and other mammals.

We followed Southall et al.'s (2007) recommendation, and used an M-weighted PTS-onset threshold of 186 dB re 1 $\mu Pa^2.s^{-1}$ for harbour seals. The increase in likelihood of PTS at higher levels of noise was then estimated by scaling up Finneran et al.'s (2005) dose response curve for changes in levels of TTS at different Sound Exposure Levels (SEL), where the probability of seals experiencing PTS increases from an SEL of 186 up to 240 dB re 1 $\mu Pa^2.s^{-1}$; the point at which all animals are predicted to have PTS.

An alternative estimate of the number of individuals experiencing PTS was also generated using the SAFESIMM model, developed at the University of St Andrews as part of the Environmental Risk Management Capability (ERMC) (Mollet et al., 2009; Donovan et al., In Press). Originally developed to support the planning of Naval exercises, SAFESIMM is being adapted to support the management of marine renewable energy developments. SAFESIMM also used sound field data from INSPIRE, and a dose-response curve for PTS (also scaled from the TTS dose-response curve in Finneran et al. (2005)). However, it incorporates a more complex individual-based animal movement model when estimating accumulated sound exposure levels for different individuals in the population.

3.3.3 Estimating the number of individuals injured or displaced

To estimate the number of individual seals that would be exposed to injury, PTS or behavioural displacement, we used the relationships detailed above to assess the extent to which received noise levels in each 4x4 km grid square (e.g. Figure 4) might impact the seals present in that grid square.

This process is illustrated in Figure 6, where we estimate the number of harbour seals that may be displaced or suffer from PTS as a result of driving the Beatrice Demonstrator's 1.8 m piles. Figure 6a presents the maximum received levels in each cell both using dB $_{ht}$ (harbour seal) and fleeing animal M-weighted SEL as metrics. In this case, the 2010 estimated population of 1,183 seals was distributed across grid cells in relation to the values shown in Figure 2. We then predict the proportion of seals in each cell that would be displaced by the received levels in that cell as

estimated using the relationships for behavioural disturbance and PTS, and sum these proportions to provide the total number of individuals affected.

When modelling long-term population change, we also compared estimates of the number of animals experiencing PTS using this approach and using SAFESIMM.

3.4 Linking individual impacts to changes in vital rates

To model long-term population level effects of wind farm construction, we had to make a series of assumptions about potential changes in the reproductive and survival rates of those individuals that are predicted to be displaced or experience PTS. In the absence of empirical data, these assumptions were based upon expert opinion, where possible drawing from general ecological understanding or proxy studies. These assumptions were developed through a series of informal discussions and workshops with research scientists and other stakeholders. The rationale behind these assumptions is discussed below, and our proposed realistic worst-case scenarios are summarised in Table 2.

3.4.1 Death and non-auditory injury.

Extremely loud noises may have a direct effect on mortality at very close range, but this is expected to occur only at levels exceeding 220 dB $_{ht}$ (harbour seal). INSPIRE modelling indicated that received levels from the installation of the 1.8 m piles only reached this level within <50 m. These are potentially major impacts at close range, but will be avoided by developing mitigation procedures routinely used during oil and gas surveys (JNCC, 2010). This assessment therefore focuses on the less direct effects of PTS and behavioural avoidance.

3.4.2 Consequences of PTS.

Changes in hearing sensitivity might impact vital rates through changes in an individual's ability to forage, avoid predators or find mates. Harbour seals have extremely sensitive vibrissae which allow them to follow hydrodynamic trails from prey (Dehnhardt et al., 2001) and discriminate between different sized or shaped objects (Wieskotten et al., 2011). Given these capabilities, changes in hearing sensitivity from PTS appear less likely to have a direct impact on foraging ability compared with cetaceans. Furthermore, if PTS occurs in individuals that remain in an area from which other marine mammals have been disturbed, there could even be positive fitness consequences from reduced competition. Where killer whale predation is high, a decrease in hearing sensitivity could increase the seals' risk of predation (Deeke et al., 2002). However, killer whales are rarely encountered in the North Sea and it seems unlikely that PTS would increase the risk of predation in this area. Finally, males make broad band vocalizations during their reproductive displays (Van Parijs et al., 1997), and these sounds may form cues when females are selecting males (Hayes et al., 2004). However, a reduction in hearing ability within part of the hearing range would seem unlikely to significantly reduce reproductive success, given that displays involve other visual and geographical cues and often occur in areas with relatively high levels of masking noise (Van Parijs et al., 1997, 1999). Nevertheless, there may be unknown fitness costs resulting from a decline in hearing ability that could affect reproduction or survival, and there was general stakeholder agreement that assessments of population level impacts should take account of this.

We addressed this by assuming that individuals experiencing PTS should be subjected to an additional mortality risk factor. In the absence of any data that could provide direct information on the mortality risk from PTS, we assumed that it could at

worst be of a similar magnitude to the impact of old age. Information on age-specific survival in wild mammals is generally lacking but, typically, survival rates in the oldest age classes are 65-85% of adults in their prime (e.g. Loison et al.,1999; Beauplet et al., 2006). In our assessments, we assume that these costs are borne in the year after exposure, and impose an additional 25% risk of mortality on all animals that are estimated to have PTS. We assess the sensitivity of our results to this assumption by varying the severity of this additional mortality from 10-30%.

3.4.3 Consequences of Behavioural Displacement.

We assume that the main behavioural impacts of noise are likely to result from avoidance of preferred foraging areas. The widespread distribution of harbour seals around the UK and other North Atlantic waters demonstrates that suitable foraging habitat is widespread, and their broad diet highlights that this is an extremely adaptable species. However, individual harbour seals also demonstrate high levels of site-fidelity (Cordes, 2011) and foraging ranges may be constrained around these favoured breeding and haul-out sites. Displacement could therefore lead to increased competition for food, greater energetic cost of foraging, or reduced foraging opportunities. As capital breeders, harbour seals build up energy resources throughout the year, feeding little or not at all during the breeding season. Given this life-history pattern, individuals should be relatively well buffered against short-term variability in prey availability. We therefore assume that the most likely impact of any reduction in an individual seal's overall energy balance will be a decline in reproductive success, which may manifest itself either by a reduction in the number of pups born or post-weaning survival of pups. Here, we make the conservative assumption that female harbour seals that are continuously excluded from their foraging habitat will exhibit 100% breeding failure, whereas intermittent exclusion (for example due to periodic or seasonal piling activity) will result in a lesser reduction in reproductive success. In the absence of any empirical data to parameterise this relationship, we explore the consequences of different temporal patterns of disturbance by assuming a direct positive linear relationship between the proportion of the annual cycle in which disturbance occurs and the resulting reduction in reproductive success.

3.5 Modelling population level impacts

Population models have commonly been used to predict the future viability of age-structured vertebrate populations, including many species of pinnipeds. Such models are particularly useful for providing insights into the relative importance of different management options or anthropogenic impacts. In the context of offshore wind farms, population models have generally been considered in relation to assessments of the impact of bird strikes (Maclean et al., 2007).

Recently, simple models have calculated the Potential Biological Removal (PBR) to provide managers with estimates of acceptable mortality from harvesting, culling or by-catch (Wade, 1998; Butler et al., 2008). This approach is suitable for supporting the management of activities that directly cause mortality, but is not adequate for assessing non-lethal anthropogenic impacts. Therefore, we adapted the stage-based matrix model previously used to estimate the impact of shooting on the Moray Firth harbour seal population (Thompson et al., 2007). By taking this approach, we were also able to explore potential changes in reproductive output or mortality that affect just certain age-classes or sexes. Furthermore, this approach allows us to incorporate cumulative impacts if, for example, licences are being granted to shoot seals within this management region (Butler et al., 2008)

We consider three life-history stages (Figure 7) and model just the female component of the population, using an assumed equal sex ratio to inflate to total population size. Our baseline model uses the same input parameters as Thompson et al. (2007), supplemented by more recent analyses of photographic sightings of >150 individually recognisable harbour seals in Loch Fleet (Cordes, 2011), which is the harbour seal breeding site closest to the proposed development areas (Cordes et al., 2011). The input parameters used in the baseline model are presented in Table 3.

4. Application of the framework to assess construction scenario and sensitivity to key assumptions

We illustrate the use of this framework, and explore its sensitivity to key parameters, using a hypothetical construction scenario in which piling occurs simultaneously at two sites within the MORL development area over a four year period, starting in 2015.

The potential long-term impact of this hypothetical construction scenario on the Moray Firth harbour seal population was modelled by adjusting reproductive and mortality rates for the proportion of the population that were predicted to be affected by piling noise, as outlined in Table 2. We assume that any risk of immediate direct mortality can be avoided by mitigation, and that behavioural displacement occurs throughout the piling period (i.e. 100% of the year). The consequences of behavioural displacement were modelled as reduced reproductive success of displaced females by removing an appropriate number of pups (stage 1 seals in our model) in each of the construction years. To model the effects of PTS, we calculated the number of individuals that may suffer PTS, and removed 25% of these individuals from the population in each year. The resulting changes in population size over a 25-year period were then compared with a baseline model in which no construction took place.

Table 4 presents data on the number of animals that are estimated to be displaced or experience PTS as a result of noise exposure from our hypothetical construction scenario. For displacement, we present values estimated using the upper, best-fit and lower behavioural response curves in Figure 5b. For PTS, we present estimates made using our assessment framework (see Figure 6) and estimates that have been produced by applying the same spatial distribution of received noise levels and initial animal densities within SAFESIMM.

As seen in Table 4, our estimates of the number of animals experiencing PTS were lower than those derived from SAFESIMM. Figure 8 then compares the long-term population consequences of these individual impacts for the three different levels of behavioural displacement, and two different values for the severity of mortality resulting from PTS. In this case, we used SAFESIMM's estimates of PTS to illustrate worst-case scenarios. Figure 9 compares long-term trends resulting from the use of these two different estimates of the number of animals experiencing PTS, and also explores the consequences of varying our assumption about the carrying capacity of the population. These comparisons highlight that the population trends appear to be driven largely by the baseline dynamics of the population. Although worse-case scenarios of impacts during construction could potentially lead to a short-term reduction in numbers, the long-term dynamics are not especially sensitive to uncertainty, for example, over the level of mortality resulting from PTS.

5. Assessing the significance of impacts

The spatial overlap of received sound levels and seal distribution, in combination with estimates of the impacts of noise exposure, potentially predicts a large number of seals being either displaced or experiencing PTS (Table 4). However, the population modelling used within the framework indicates that this should not result in long-term changes to the viability of this population. The use of different behavioural response curves and methods for estimating the impact of PTS resulted in variations in detail, but the common pattern at the population level was of short term reductions in abundance during and immediately after the construction period, followed by recovery that resulted in no observable difference between impact and baseline scenarios after 25 years.

A key issue when conducting Appropriate Assessments is the reversibility of any potential impacts on SACs. We therefore developed criteria that allowed us to use the outputs of our assessment framework to summarise the significance of impacts of different construction scenarios, taking account of both the magnitude and duration of those impacts (Table 5). The magnitude scale was guided by the principle that a high magnitude change should be measureable within the relevant timescale, taking account of background variation and sampling variability. At short durations (here a number of months), the magnitude of impacts can be assessed using estimates of the number of individuals in the population affected. At medium durations (a number of years), magnitude can be assessed by comparing the maximum difference between predicted population sizes for construction and baseline scenarios (typically at the end of the construction period). Finally, the long-term significance of developments can be assessed by comparing construction and baseline population sizes after 25 years.

6. Assessing and reducing uncertainty

Even for this relatively well-studied population, there remains enormous variation in the quality of data available to parameterise the different components of this framework and several key parameters have to be based upon expert opinion. Although some stakeholders would prefer to see additional data collected before decisions are made, this is impractical within the consenting timelines for the majority of proposed offshore wind farms. Instead, consenting decisions must be made utilising the information available so that regulators can achieve a balance between international agreements on climate change and nature conservation.

There are serious limitations in the amount of data available to assess the impacts of noise on marine mammal populations (Southall et al., 2007). Furthermore, even when data are available, they are often based upon small samples, with some key studies being based on single captive individuals. Consequently, the level of scientific uncertainty underpinning each element of our assessment framework for the Moray Firth harbour seal population varies considerably. A key aim in our approach was to ensure that uncertainty was explicitly recognised, and that the framework could be used to assess where to focus efforts on additional data collection. In response to stakeholder advice, we used the Intergovernmental Panel on Climate Change guidance upon the classification of uncertainty (Mastrandea et al., 2010) to provide an indication of the relative confidence in different components of our framework. In the supplementary information, Table S1 outlines the IPCC's recommended scale for characterising confidence in a dataset or assumption, based upon expert judgement. This scale is then used in Table S2 to summarise the confidence that we place in the different data available to us for use in this assessment framework, and Table S3 outlines the key assumptions that we made,

together with an indication of the sensitivity of our results to each of these assumptions.

6.1 Seal distribution.

The telemetry data available from Moray Firth harbour seals provided a relatively high quality dataset on foraging distribution, with consistent patterns seen over a twenty year period (Cordes et al., 2011). Nevertheless, sample sizes were still small when extrapolating to the whole population, and biased towards the summer period. This currently constrains our ability to compare potential seasonal differences in foraging area use. Additional telemetry tag deployments could address this and provide better estimates of contemporary distribution and winter use prior to assessments of any changes in distribution that may occur in response to construction.

6.2 Noise distribution.

Despite differences in the approaches used for noise propagation modelling, underwater acoustics is relatively well understood and we have a high level of confidence in estimates of received noise levels. Here, we used the INSPIRE model and found a reasonably good fit between modelled peak to peak levels and measured data. However, further calibrated recordings made throughout the frequencies used by marine mammals would improve confidence in model predictions, and permit comparison between different propagation models, particularly in the far field.

6.3 Assessing impacts on individuals.

There is far less certainty about the extent to which predicted noise levels may impact individual seals. The preliminary nature of the noise exposure criteria developed by Southall et al. (2007) highlights the evolving nature of understanding in this area. Planned research in the USA should provide additional data on TTS-onset to pulsed sounds such as pile-driving (Southall, Pers. Comm.) but this remains an area where it is difficult to obtain robust data with sufficient sample sizes. Studies of individual variability in the hearing thresholds in wild harbour seals should provide an additional tool for understanding issues. Recent studies of captive marine mammals have used measurements of auditory evoked potential (AEP) to assess hearing ability (e.g. Lucke et al., 2009) and this technique has excellent potential for use on wild animals; for example when individual seals are being caught and instrumented with tracking devices. In future, routine AEP tests during captures of wild seals should provide an important baseline to underpin future studies of changes in hearing ability over time.

Given the lack of data on marine mammal behavioural responses to different levels of pulsed noise, we used published data from Horns Rev II to provide an interim proxy for a dose-response curve. This is a first step, based on small sample sizes and a study of harbour porpoises rather than harbour seals. Furthermore, these data represent displacement for only a one-hour period after piling had ceased. There is a critical need for better data on recovery times after these displacements, particularly as these will affect the cumulative extent of displacement throughout a season of intermittent piling.

Our assessment framework applied the same dose-response curve for PTS that is used within SAFESIMM. Based on data from Finneran et al. (2005), this assumes that around 18% of animals exposed to Southall et al.'s (2007) PTS-onset criteria experience PTS, with the proportion gradually increasing at higher SEL's. However,

SAFESIMM estimates of the number of seals experiencing PTS were higher. Further exploration is required to determine why these results differ, although it is likely to be due to differences in the way that the two approaches model how seals move away from loud noises. In the meantime, we take the more conservative approach and use estimates based upon SAFESIMM outputs in the AA.

6.4 Linking individual impacts to demographic parameters.

Even with more certain data on the number of individuals displaced or experiencing PTS, there remains huge uncertainty over their subsequent consequences for individual fitness. It is these parameters that currently depend entirely upon expert judgement. Here, we use values that represent reasonable worst-case scenarios, but the modelling framework has been constructed so that these can be modified to explore sensitivity to variation in these values. This also allows us to explore where further research effort might best be placed. For example, there are clear limitations in carrying out further work to understand how variation in received noise affects the likelihood of PTS. Instead, it is likely to be most appropriate to use expert judgement to inform these parameters in the short term and, in the longer term, to directly assess relationships between noise exposure and key demographic parameters using the Population Consequences of Acoustic Disturbance (PCAD) framework developed by NRC (2003). As an interim measure, the approach could also be developed to more formally collate expert opinions (e.g. Martin et al., 2012) and sample from resulting parameter distributions within a stochastic modelling framework.

The Moray Firth harbour seal population offers excellent opportunities to develop detailed PCAD studies to test the assumptions made in this assessment process. Individually identifiable seals at the haul-out sites closest to proposed wind farms have been studied since 2006, providing estimates of survival and fecundity, while direct measures of pupping date and lactation duration provide information on year-to-year variation in female condition (Cordes, 2011). Combined with established methods for tracking seals, and realistic potential for field based measurements of hearing ability and noise exposure, the PCAD approach could be integrated into construction monitoring.

6.5 Harbour seal population model.

The final element of our framework involves a simple deterministic population model for this regional population of harbour seals. Initial analyses of the distribution of seals were conducted within ArcGIS, but the resulting grid based data could then be easily manipulated within a MS Excel framework. We used a stage-base population model within Excel using the Pop Tools add-in (http://www.poptools.org). This approach also allowed us to either include or exclude other factors such as the PBR-based quota of seals that may be removed by fishermen under licence by Marine Scotland (Butler et al., 2008). One advantage of this deterministic framework is its quick operation, which allows rapid exploration of different scenarios and model sensitivity, potentially in workshop situations with different stakeholder input. Like the PCAD models discussed above, future work would benefit from using stochastic models to incorporate uncertainty into model predictions.

7. Applicability of the framework to other marine mammal populations.

This framework was developed to inform consenting decisions in NE Scotland, where potential impacts on local harbour seal SAC's have been identified. However, consent of these and other Scottish Territorial Water and Round 3 sites will also

depend upon AA for species such as bottlenose dolphins, and different populations of harbour seals. Similarly, some North Sea developments are likely to require AA for harbour porpoises.

The history of research on the Moray Firth harbour seal population has clearly been a great benefit in the development of this framework, but a lack of such detailed site specific data should not constrain the use of this approach for other regional harbour seal populations. While the temporal spread of telemetry data in the Moray Firth is unique, extensive tracking has been conducted in other parts of the UK over the last 10 years (e.g. Sharples et al., 2012), and these data are currently being used in broader-scale habitat models to characterise harbour seal foraging distribution around the UK. Similarly, although annual haul-out counts are made at only a few UK sites, a regular programme of surveys provides broad-scale data on abundance and trends in different UK regions (Lonergan et al., 2007; SCOS, 2011).

One concern is the extent to which less frequent surveys in other areas accurately reflect recent regional trends. This will be important to establish, as initial model runs highlight that predicted long-term trends are driven largely by the underlying baseline trend. When baseline conditions are favourable, harbour seal populations can grow rapidly as demonstrated by rapid recovery from major natural mortality events such as Phocine Distemper Virus outbreaks (Härkönen et al., 2006). In contrast, some Scottish populations have shown marked declines over the last decade (Lonergan et al., 2007) and added pressures from renewable developments may exacerbate these declines even where they are not driving them. A good regional time-series of annual haul-out counts is therefore an important pre-requisite if this framework is to be used in other areas. It is likely to prove more difficult to obtain comparable demographic data in other regions and, even where individual-based studies can be initiated, several years of intensive research will be required before robust survival estimates can be made. On the other hand, fecundity estimates could be based on other data sources, as for UK grey seals, which may be collected more easily at other sites over shorter periods. Alternatively, it is a common approach to "borrow" data from better studied populations, or even other species (e.g. Caswell et al., 1998), when developing population models. Such uncertainty should therefore not constrain the development of similar modelling frameworks for other populations.

A more challenging issue results from species differences in ranging patterns. For example, evidence from a series of harbour seal tracking studies highlights that individual seals repeatedly spend several days at a time in the same foraging areas, travelling to and from favoured haul-out sites that provide a central place for their foraging activity (Thompson et al., 1998). In contrast, many cetaceans such as bottlenose dolphins range widely, often in large groups, visiting favoured foraging hot spots (Hastie et al., 2004). Sometimes groups may even travel several hundred miles in a few days (Wilson et al., 1999; Cheney et al., In Press), highlighting the need to assess cumulative impacts upon harbour seals and bottlenose dolphins at different scales. Applying the relationship in Figure 5b to estimate the probability of displacing animals from a particular grid cell is also more problematic for mobile bottlenose dolphins, as these animals would be expected to have moved from that area after a few hours in the absence of any noise impact. For both bottlenose dolphins and harbour porpoises, consequences of displacement may be less critical, given these species are not tied to local breeding or resting sites, but assessments of these consequences must consider conditions at potential feeding sites elsewhere in geographical range. As for seals, assumptions could be made about the individual fitness consequences of displacement or PTS, and these effects applied to

population models such as those developed to assess the impacts of porpoise by-catch (Moore and Read, 2008; Winship, 2009).

8. Conclusions

In an ideal world, assessments of the consequences of pile-driving or other industrial noise on protected marine mammal populations would be based upon a detailed understanding of dose-response relationships between received noise levels and changes in vital rates. This may become possible in the future through initiatives such as the PCAD programme, but alternative interim approaches are required to provide regulators with confidence that proposed developments will not significantly impact the long-term integrity of populations using SACs. We argue that this must involve some element of expert judgement, and that the framework we have developed for assessing impacts upon the Moray Firth harbour seal population illustrates how this can be achieved in a transparent and adaptable way. Whilst we recognise that this approach involves considerable uncertainty, this framework clearly documents which elements are based upon empirical data and outlines the rationale underlying parameters that are informed by expert knowledge. In the future, uncertainty could be quantified more formally within a stochastic modelling framework that samples from distributions representing the full breadth and weight of expert opinion. However, to provide assessments within the timescales currently demanded by regulators, we have instead dealt with this issue by selecting conservative estimates for individual parameters. Whilst this is appropriate for ensuring that the worst-case scenarios do not compromise long-term population viability, this conservatism accumulates through the framework. This leads to much more significant short term impacts than we anticipate are likely. It is therefore important that stakeholders recognise that this assessment framework is assessing worst-case impacts to meet the needs of the EU Habitats Directive, and that these require moderation where assessments of most likely impact are required under the EU Environmental Assessment regulations. Research and monitoring programmes should be carefully designed around key consented wind farms to explicitly test these assumptions, and inform the development of this framework to reduce uncertainty and conservatism in future assessments.
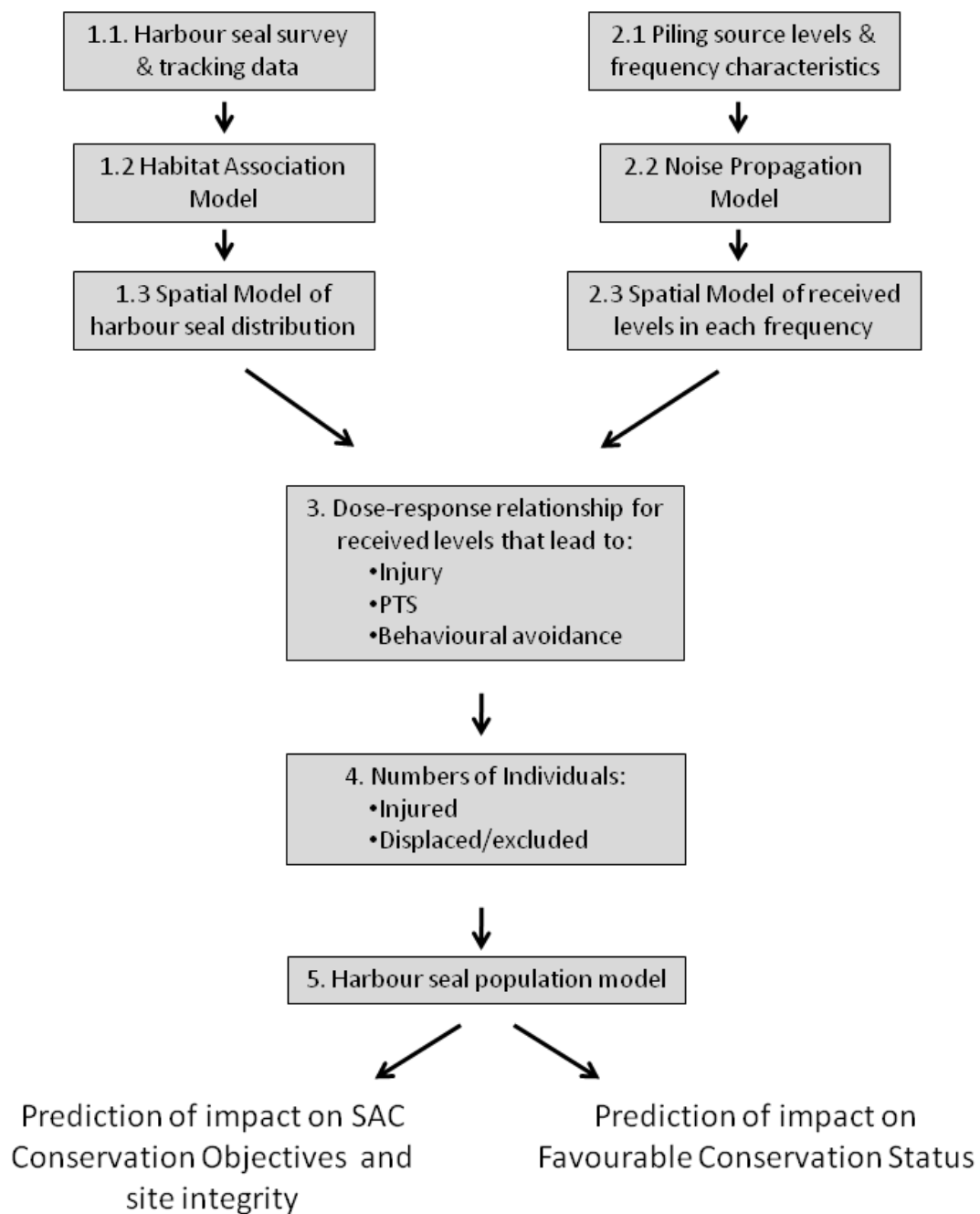
Figure 1

Figure 2

Figure 3
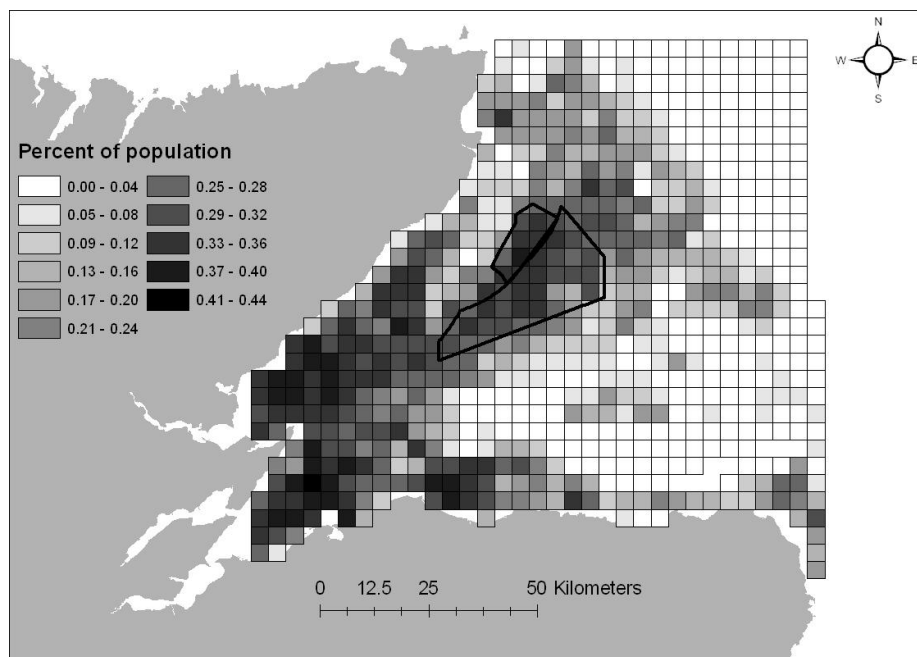
Figure 4

Harbour seal dBht
Beatrice DOWNVInD piling

	45
	50
	55
	60
	65
	70
	75
	80
	85
	90
	100
	110
	130

0   10   20   40 Kilometers
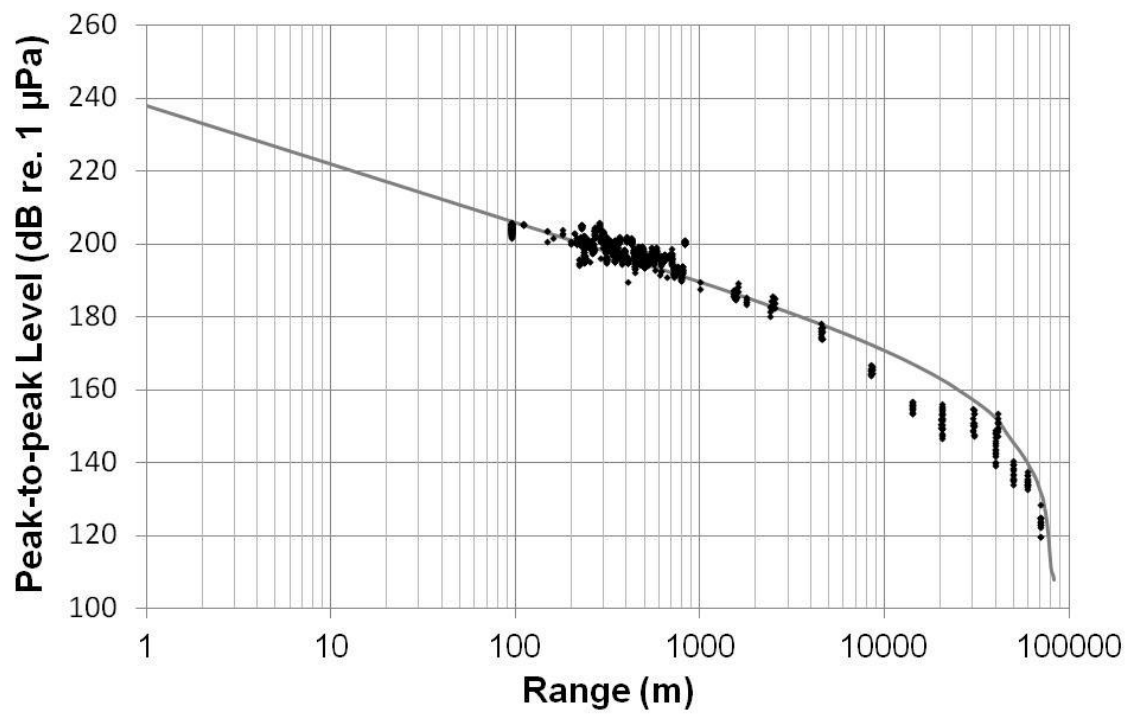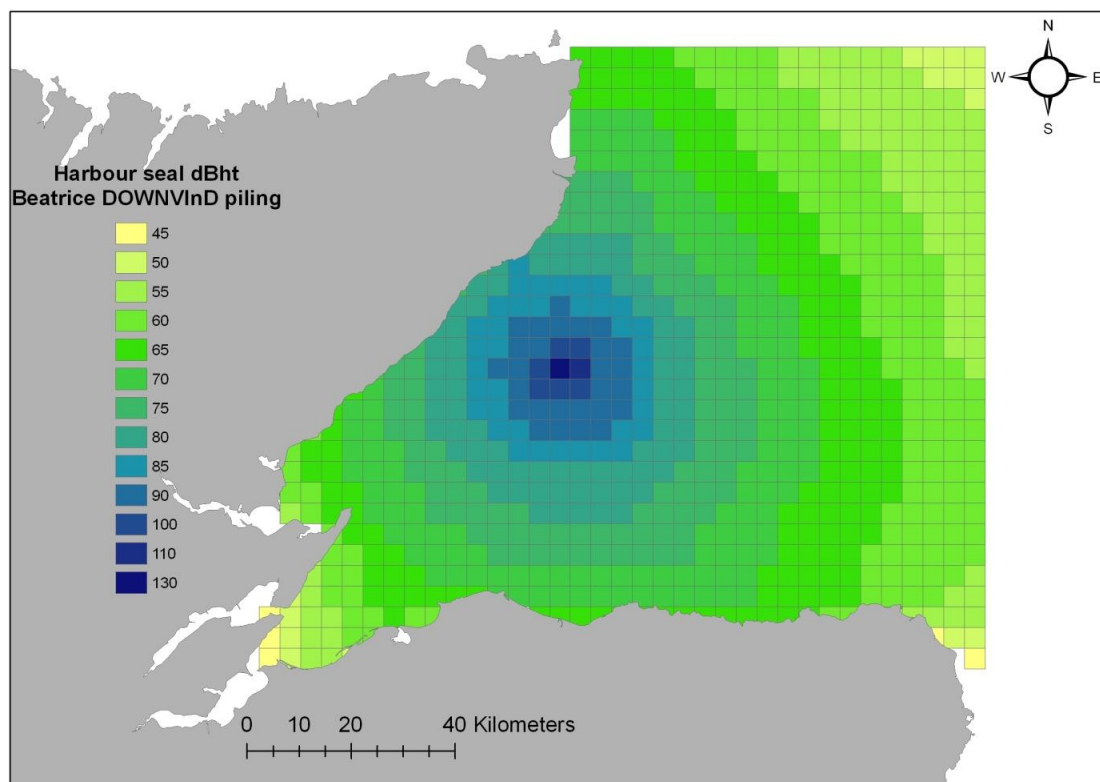
Figure 5

a)
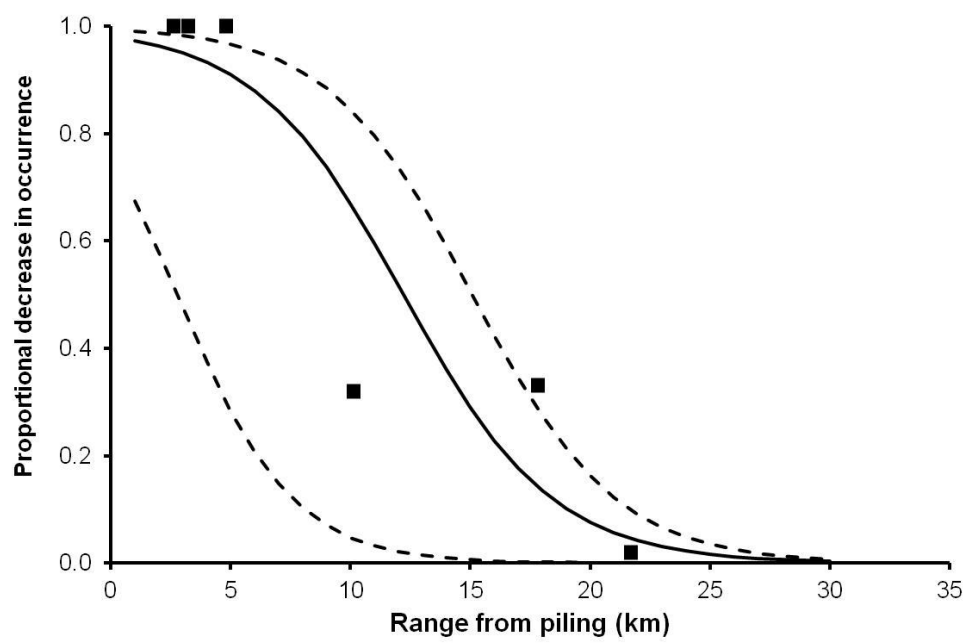
b)
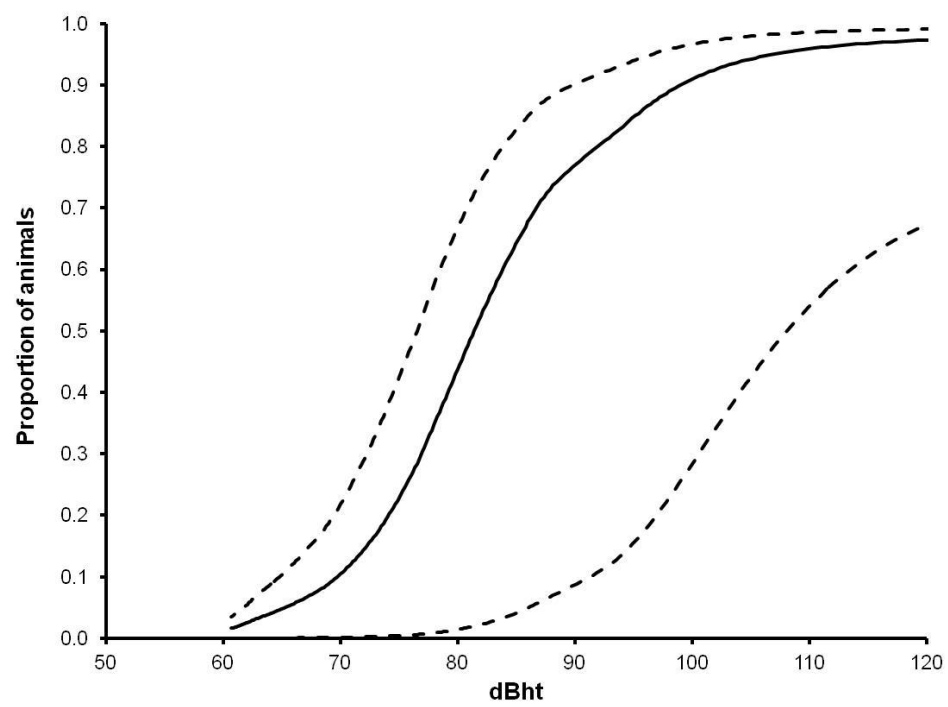
Figure 6

a) Map Received Sound levels

dB$_{ht}$

M-weighted SEL
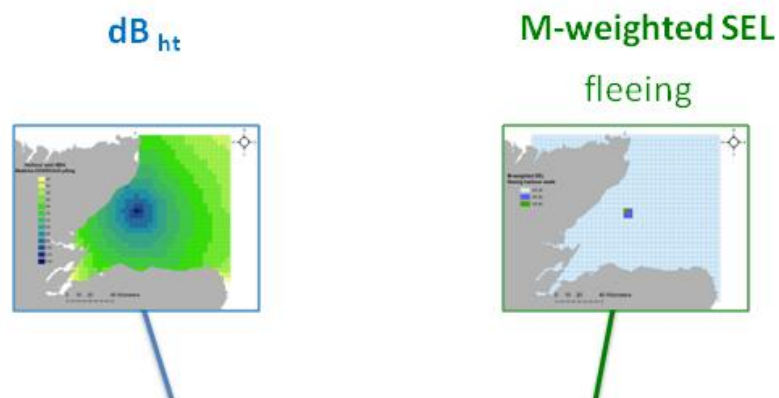
fleeing

b) Estimate number of seals in each cell that are:

Displaced
(Figure 5b)
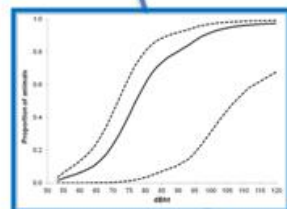
PTS
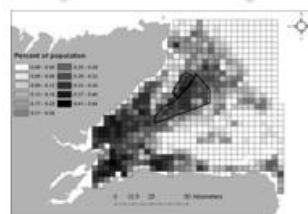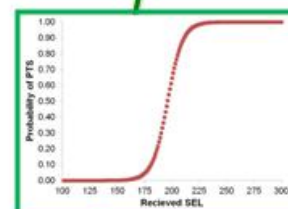
Based upon seal
distribution
(Figure 2)

c) Sum to estimate total number of animals affected

Behavioural Displacement			PTS
Upper	Best fit	Lower	Fleeing Model
690	511	55	56

Figure 7
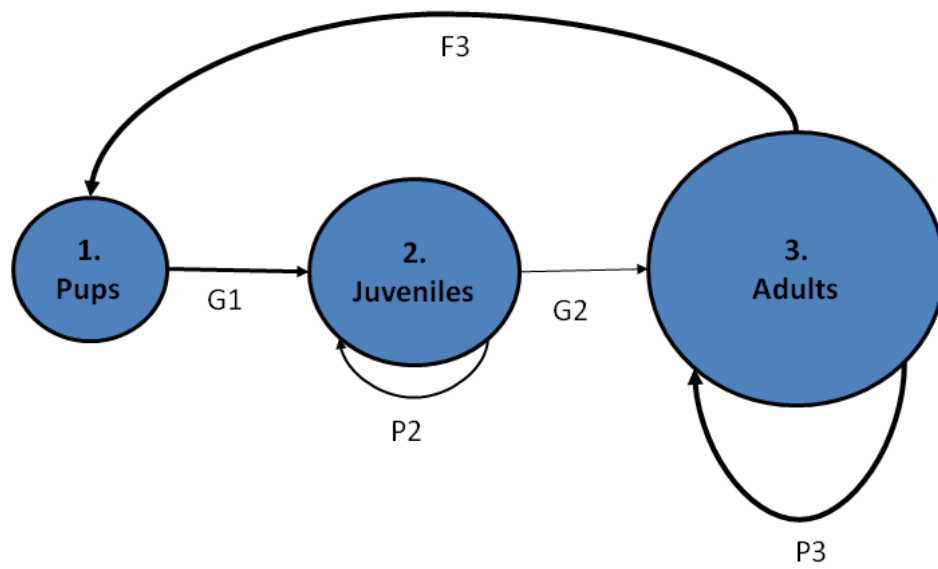
Stage	1	2	3		Stage	1	2	3
1	P1	F2	F3		1	0	0	0.88
2	G1	P2	0		2	0.7	0.66	0
3	0	G2	P3		3	0	0.19	0.97

Figure 8

		Level of assumed mortality resulting from PTS	
		10%	30%
Behavioural displacement curve used	Upper		
	Best		
	Lower		

Figure 9

	Estimate of PTS using seal assessment framework	Estimate of PTS using SAFESIMM
K= 4000		
K= 2000		
K= 1000		

Figure legends

Figure 1. Schematic of the approach proposed for assessing the impact of wind farm construction on the harbour seal Special Area of Conservation (SAC) and Favourable Conservation Status (FCS).

Figure 2. At-sea distribution of harbour seals in the Moray Firth. Data are predictions from habitat association modelling based upon telemetry data from 37 harbour seals, and show the percentage of the population that is expected to be found in each of the different 4 x 4 km grid cells. The boundaries of the proposed MORL and BOWL wind farms are shown as a solid black line.

Figure 3. The level of sound in unweighted peak to peak levels as a function of range in meters for a 1.8 m diameter pile at the Beatrice demonstrator site. Predictions from the INSPIRE model (solid line) are presented alongside measured levels (circles) from Bailey et al. (2010).

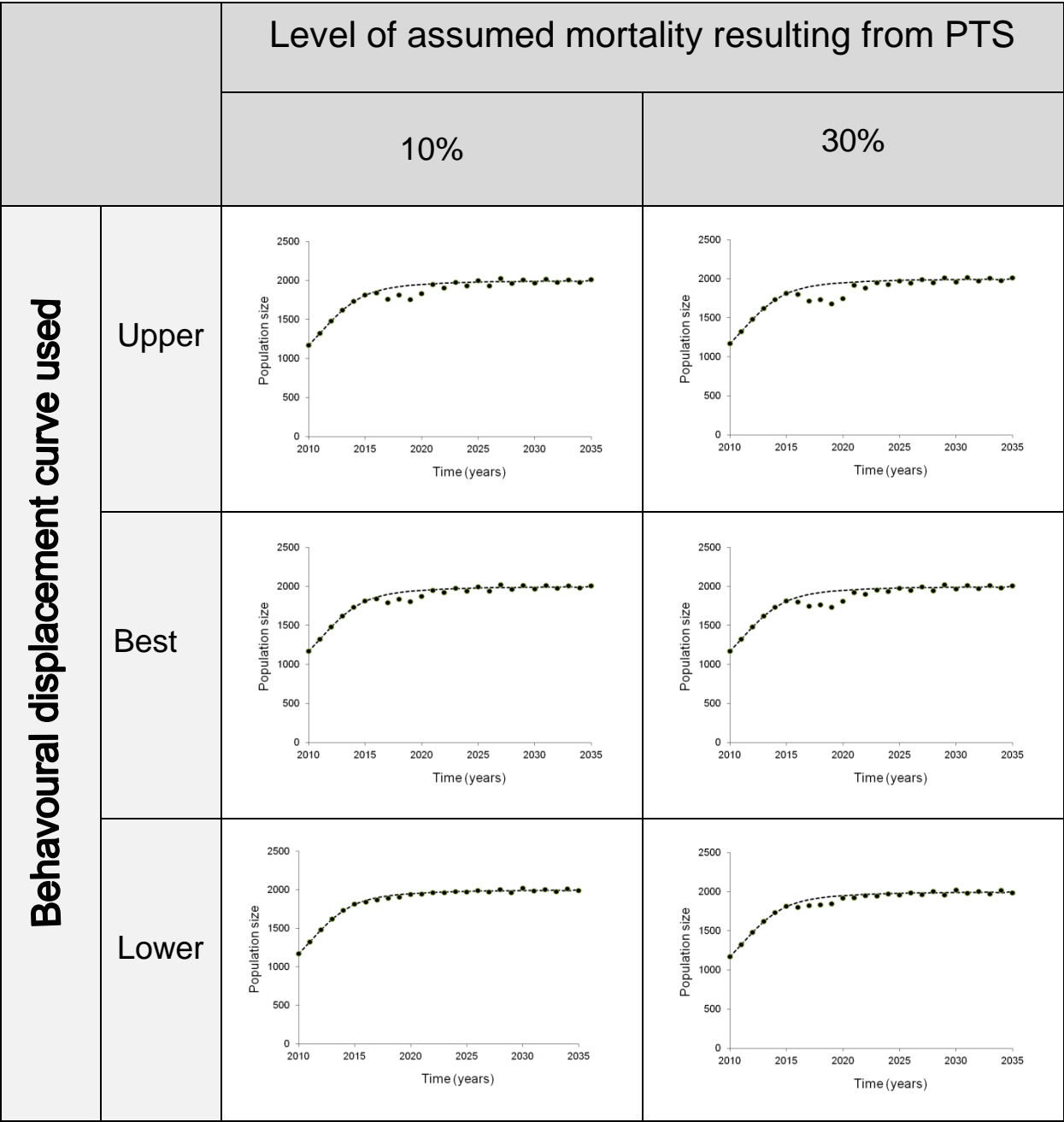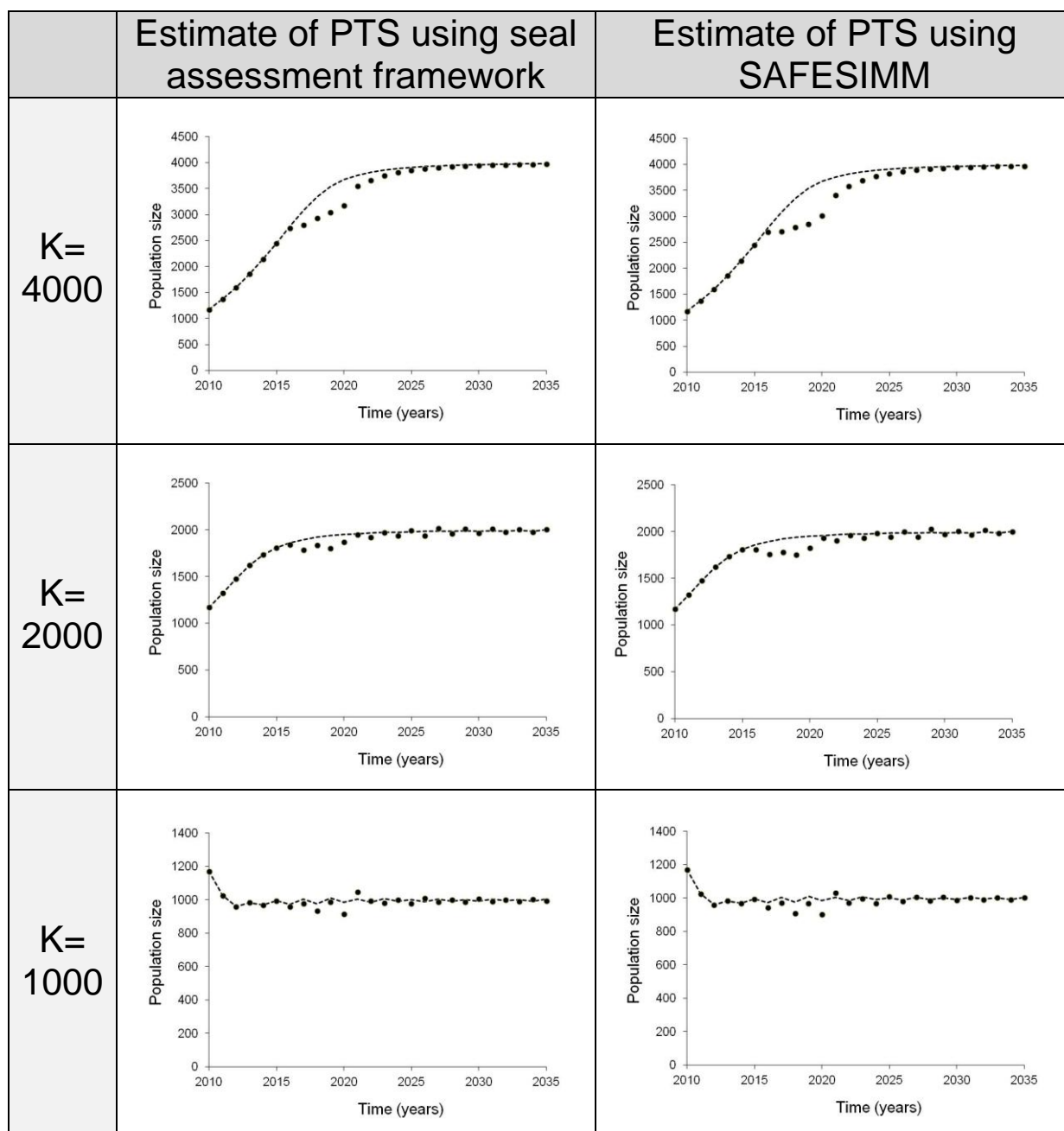Figure 4. Predicted received levels of pile-driving noise generated from Subacoustech's INSPIRE model, showing variation in received levels (dB $_{ht}$ (harbour seal)) interpolated for each of the 4 x 4 km grid cells.

Figure 5: a) Predicted relationship between range from the Horns Rev II piling operation and the proportional decrease in harbour porpoise occurrence (mean porpoise positive minutes from CPODs (from Brandt et al., 2011)) in the hour after the event; relationship for the line of best fit (deviance = 4.19; d.f.=1; P<0.05; Intercept=3.9 (se=2.77; Range= -0.32 (SE=0.23)). The best fitted relationship is shown as a solid line. Standard errors were used to provide confidence limits around this relationship. However, because small sample sizes resulted in the upper bound showing almost no variation across the range of distances studied, we instead produced an upper bound for the relationship by weighting the line to include all data points. The lower bound is based upon the standard error of the coefficients. b) The relationship between dB $_{ht}$ (harbour porpoise) and the predicted proportion of animals excluded from the area (using the upper, best and lower fitted relationship from Figure 5a).

Figure 6. Schematic illustrating the approach used to assess the number of harbour seals from an estimated population of 1,183 individuals that are displaced and vulnerable to Permanent Threshold Shift from an event involving the installation of two piles in 24hrs.

Figure 7. A life-cycle graph for the stage-classified single sex harbour seal model. Values for reproduction and survival rates, which represent the transition between stages, are taken from Table 3.

Figure 8. Comparison of impact and baseline population trends using different behavioural response curves and severity levels for the mortality that may result from Permanent Threshold Shift (PTS). The percentages of the population affected in each of the 4 years of construction are taken from the hypothetical example in Table 4. Estimates of PTS are based upon numbers for the SAFESIMM modelling to provide a more precautionary approach.

Figure 9. Comparison of the effects of using different estimates for the number of individuals experiencing Permanent Threshold Shift (PTS) and of varying assumptions about the carrying capacity (K) of the environment. Mortality resulting from PTS is assumed to be 25% in all of these model runs.

Table 1. Potential effects of noise upon marine mammals in order of severity.

1. Lethality and physical injury	
• Immediate death	Typically associated with rapid compression of air containing structures
• Physical Injury	
2. Auditory Damage	
• Permanent Auditory Trauma/ (Permanent Threshold Shift)	Permanent elevation of hearing threshold, caused by prolonged exposure to noise
• Temporary Threshold Shift	Temporary elevation of hearing threshold.
3. Behavioural Effects	
• Avoidance	See Table 4 in Southall et al. (2007) more detailed breakdown of behavioural effects
• Changes in foraging or social behaviour	

Table 2. Assumed worst-case fitness consequences for individual seals exposed to different levels of pile-driving noise.

Effect	Consequence	
	Intermittent exposure	**Constant exposure**
Immediate death	Immediate Mortality	Immediate Mortality
Physical Injury	Immediate Mortality	Immediate Mortality
Permanent Threshold Shift	25% risk of mortality	25% risk of mortality
Behavioural Avoidance	Proportional reduction in reproductive success/and or juvenile survival	100% reproductive failure

Table 3. Values used for the life-history parameters and ecological characteristics used as input parameters in our baseline model

Parameter	Values used	Source
Starting population size	1183	Estimate based upon SMRU 2010 surveys.
Age at first reproduction ♂	5 ♂, 4 ♀	Härkönen and Heide-Jørgensen (1990).
Reproductive rate	88%	Cordes (2011).
Sex ratio	0.5	Boulva and McLaren (1979).
Density dependent variation in reproduction	Yes	Using equation 3 in Taylor and DeMaster (1993) to vary reproductive rate between maximum literature value at low population size (0.95 (Boulva and McLaren, 1979)) and a value of 0.1 at K (based on observed change in other pinnipeds (Fowler, 1990)).
Carrying Capacity	2000	Conservative estimate based upon a value that is ~ 20% higher than the maximum abundance estimate in the last 20 yrs.
Pup/Juvenile Mortality	30%	Harding et al., 2005; Härkönen and Heide-Jørgensen, 1990.
Adult mortality	11% ♂; 3% ♀	Cordes (2011).

Table 4. Variation in the estimated number of individuals displaced and experiencing Permament Threshold Shift (PTS) when using different curves to estimate behavioural resposnes, and when using our assessment framework and SAFESIMM to assess levels of PTS. Simulations use a hypothetical scenario involving simultaneous piling at two sites in the Moray Firth, and a population size of 1183 harbor seals.

	Number of individuals	% of population
Behavioural Displacement		
Upper curve	690	58.7%
Best fit curve	511	43.2%
Lower curve	55	4.7%
Permanent Threshold Shift		
Seal assessment Framework	56	4.75%
SAFESIMM	175	14.8%

Table 5. Criteria used for predicting significance from magnitude of impact and duration.

Magnitude	Duration		
	Short (days)	**Medium (construction years)**	**Long-term (25yrs)**
High (>20%) of population	Major significance (short term)	Major significance (medium term)	Major significance (long-term)
Medium (>10%)	Minor significance (short term)	Medium significance (medium term)	Medium significance (long-term)
Low (<10%)	Negligible significance	Minor significance (medium term)	Minor significance (long-term)

References

Bailey H, Senior B, Simmons D, Rusin J, Picken G, Thompson PM. Assessing underwater noise levels during pile-driving at an offshore windfarm and its potential effects on marine mammals. Mar Pollut Bull. 2010;60:888-97.

Beauplet G, Barbraud C, Dabin W, Kussener C, Guinet C. Age-specific survival and reproductive performances in fur seals: evidence of senescence and individual quality. Oikos. 2006;112:430-41

Boulva J, McLaren IA. Biology of the harbour seal, Phoca vitulina, in eastern Canada. Bulletin of the Fisheries Research Board of Canada. 1979;200:1-24.

Brandt MJ, Diederichs A, Betke K, Nehls G. Responses of harbour porpoises to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. Marine Ecology-Progress Series. 2011;421:205-16.

Butler J, Middlemas S, McKelvey S, Mcmyn I, Leyshon B, Walker I, et al. The Moray Firth Seal Management Plan: an adaptive framework for balancing the conservation of seals, salmon, fisheries and wildlife tourism in the UK. Aquatic Conservation-Marine and Freshwater Ecosystems. 2008;18:1025-38.

Carstensen J, Henriksen OD, Teilmann J. Impacts of offshore wind farm construction on harbour porpoises: acoustic monitoring of echolocation activity using porpoise detectors (T-PODs). Marine Ecology-Progress Series. 2006;321:295-308.

Caswell H, Brault S, Read AJ, Smith TD. Harbor porpoise and fisheries: An uncertainty analysis of incidental mortality. Ecological Applications. 1998;8:1226-38

Cheney B, Thompson PM, Ingram SN, Hammond PS, Stevick PT, Durban JW, et al. Integrating multiple data sources to assess the distribution and abundance of bottlenose dolphins Tursiops truncatus in Scottish waters. Mammal Review. In Press.

Cordes L. Demography and breeding phenology of a marine top predator. PhD Thesis: University of Aberdeen; 2011.

Cordes LS, Duck CD, Mackey BL, Hall AJ, Thompson PM. Long-term patterns in harbour seal site-use and the consequences for managing protected areas. Animal Conservation. 2011;14:430-8.

Deecke VB, Slater PJB, Ford JKB. Selective habituation shapes acoustic predator recognition in harbour seals. Nature. 2002;420:171-3

Dehnhardt G, Mauck B, Hanke W, Bleckmann H. Hydrodynamic trail-following in harbor seals (Phoca vitulina). Science. 2001;293:102-4.

Dolman S, Simmonds M. Towards best environmental practice for cetacean conservation in developing Scotland's marine renewable energy. Mar Pol. 2010;34:1021-7.

Donovan CR, Harris CM, Harwood J, Milazzo L. A simulation-based method for quantifying and mitigating the effects of anthropogenic sound on marine mammals. Acoust Soc Am. *Proceedings of Meetings on Acoustics* In Press

Drewitt AL, Langston RHW. Assessing the impacts of wind farms on birds. Ibis. 2006;148:29-42.

Ellison WT, Southall BL, Clark CW, Frankel AS. A New Context-Based Approach to Assess Marine Mammal Behavioral Responses to Anthropogenic Sounds. Conserv Biol. 2012;26:21-8.

Finneran JJ, Carder DA, Schlundt CE, Ridgway SH. Temporary threshold shift in bottlenose dolphins (Tursiops truncatus) exposed to mid-frequency tones. Journal of the Acoustical Society of America. 2005;118:2696-705.

Fowler CW. Density Dependence in Northern Fur Seals (Callorhinus ursinus). Marine Mammal Science. 1990;6:171-95.

Götz T, Janik VM. Aversiveness of sounds in phocid seals: psycho-physiological factors, learning processes and motivation. Journal of Experimental Biology. 2010;213:1536-48.

Harding KC, Fujiwara M, Axberg Y, Härkönen T. Mass-dependent energetics and survival in Harbour Seal pups. Functional Ecology. 2005;19:129-35.

Härkönen T, Dietz R, Reijnders P, Teilmann J, Harding K, Hall A, et al. The 1988 and 2002 phocine distemper virus epidemics in European harbour seals. Dis Aquat Organ. 2006;68:115-30.

Härkönen T, Heide Jorgensen MP. Comparative life histories of East Atlantic and other Harbor Seal populations. Ophelia. 1990;32:211-35.

Hastie G, Wilson B, Wilson L, Parsons K, Thompson P. Functional mechanisms underlying cetacean distribution patterns: hotspots for bottlenose dolphins are linked to foraging. Mar Biol. 2004;144:397-403.

Hayes SA, Kumar A, Costa DP, Mellinger DK, Harvey JT, Southall BL, et al. Evaluating the function of the male harbour seal, Phoca vitulina, roar through playback experiments. Animal Behaviour. 2004;67:1133-9.

Inger R, Attrill MJ, Bearhop S, Broderick AC, Grecian WJ, Hodgson DJ, et al. Marine renewable energy: potential benefits to biodiversity? An urgent call for research. Journal of Applied Ecology. 2009;46:1145-53.

Jay S. Mobilising for marine wind energy in the United Kingdom. Energy Policy. 2011;39:4125-33.

Loison A, Festa-Bianchet M, Gaillard JM, Jorgenson JT, Jullien JM. Age-specific survival in five populations of ungulates: Evidence of senescence. Ecology. 1999;80:2539-54.

Lonergan M, Duck CD, Thompson D, Mackey BL, Cunningham L, Boyd IL. Using sparse survey data to investigate the declining abundance of British harbour seals. Journal of Zoology. 2007;271:261-9.

Lucke K, Siebert U, Lepper PA, Blanchet M-A. Temporary shift in masked hearing thresholds in a harbor porpoise (Phocoena phocoena) after exposure to seismic airgun stimuli. J Acoust Soc Am. 2009;125:4060-70.

Maclean IMD, Frederiksen, M, Rehfisch, MM. Potential use of population viability analysis to assess the impact of offshore wind farmwind farmswind farm on bird populations.2007; BTO Research Report No. 480 to COWRIE, BTO, Thetford.

Martin TG, Burgman MA, Fidler F, Kuhnert PM, Low-Choy S, Mcbride M, et al. Eliciting Expert Knowledge in Conservation Science. Conserv Biol. 2012;26:29-38.

Masden EA, Fox AD, Furness RW, Bullman R, Haydon DT. Cumulative impact assessments and bird/wind farm interactions: Developing a conceptual framework. Environmental Impact Assessment Review. 2010;30:1-7.

Mastrandrea M, Field C, Stocker T, Edenhofer O, Ebi K, Frame D, et al. Guidance Note for Lead Authors of the IPCC Fifth Assessment Report on Consistent Treatment of Uncertainties. Intergovernmental Panel on Climate Change (IPCC). 2010. http://www.ipcc.ch/pdf/supporting-material/uncertainty-guidance-note.pdf , Accessed 9 Oct 2012.

Matthiopoulos J, Cordes L, Mackey B, Thompson D, Duck C, Thompson P. State-space modeling reveals proximate causes of harbour seal population declines. Journal of Applied Ecology. In Review.

Moore JE, Read AJ. A Bayesian Uncertainty Analysis of Cetacean Demography and Bycatch Mortality Using Age-at-Death Data. Ecological Applications. 2008;18:1914-31.

Nedwell J, Turnpenny A, Lovell J, Parvin S, Workman R, Spinks J, et al. A validation of the dB_{ht} as a measure of the behavioural and auditory effects of underwater noise. Department fo Business, Enterprise and Regulatory Reform: Subacoustech; 2007. http://www.subacoustech.com/information/downloads/reports/534R1231.pdf , Accessed 9 Oct 2012.

NRC. Ocean Noise and Marine Mammals. Washington, DC: National Academies; 2003.

Richardson W, J, Greene C, Malme C, Thomson D. Marine Mammals and Noise. San Diego, California: Academic Press; 1995.

Scheidat M, Tougaard J, Brasseur S, Carstensen J, Petel TvP, Teilmann J, et al. Harbour porpoises (Phocoena phocoena) and wind farms: a case study in the Dutch North Sea. Environmental Research Letters. 2011;6:025102.

SCOS. Scientific Advice on Matters Related to the Management of Seal Populations: 2011. NERC Sea Mammal Research Unit; 2011. http://www.smru.st-and.ac.uk/documents/678.pdf , Accessed 8 Oct 2012.

Sharples RJ, Moss SE, Patterson TA, Hammond PS. Spatial Variation in Foraging Behaviour of a Marine Top Predator (Phoca vitulina) Determined by a Large-Scale Satellite Tagging Program. Plos One. 2012;7: DOI: 10.1371/journal.pone.0037216

Söderman T. Natura 2000 appropriate assessment: Shortcomings and improvements in Finnish practice. Environmental Impact Assessment Review. 2009;29:79-86

Southall BL, Bowles AE, Ellison WT, Finneran JJ, Gentry RL, Jr CRG, et al. Marine Mammal Noise Exposure Criteria: Initial Scientific Recommendations. Aquat Mamm. 2007;33:411-521.

Taylor BL, Demaster DP. Implications of Nonlinear Density-Dependence. Marine Mammal Science. 1993;9:360-71.

Thompson P, Mackay A, Tollit D, Enderby S, Hammond P. The influence of body size and sex on the characteristics of harbour seal foraging trips. Can J Zool. 1998;76:1044-53.

Thompson P, Tollit D, Wood D, Corpe H, Hammond P, Mackay A. Estimating harbour seal abundance and status in an estuarine habitat in north-east Scotland. J Appl Ecol. 1997;34:43-52.

Thompson PM, Mackey B, Barton TR, Duck C, Butler JRA. Assessing the potential impact of salmon fisheries management on the conservation status of harbour seals (Phoca vitulina) in north-east Scotland. Anim Conserv. 2007;10:48-56.

Toke D. The UK offshore wind power programme: A sea-change in UK energy policy? Energy Policy. 2011;39:526-34.

Tougaard J, Carstensen J, Teilmann J, Skov H, Rasmussen P. Pile driving zone of responsiveness extends beyond 20 km for harbor porpoises (Phocoena phocoena (L.)). J Acoust Soc Am. 2009;126:11-4.

VanParijs SM, Thompson PM, Tollit DJ, Mackay A. Distribution and activity of male harbour seals during the mating season. Animal Behaviour. 1997;54:35-43.

Van Parijs SM, Hastie GD, Thompson PM. Geographical variation in temporal and spatial vocalization patterns of male harbour seals in the mating season. Animal Behaviour. 1999;58:1231-9.

Wade PR. Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. Marine Mammal Science. 1998;14:1-37.

Wieskotten S, Mauck B, Miersch L, Dehnhardt G, Hanke W. Hydrodynamic discrimination of wakes caused by objects of different size or shape in a harbour seal (Phoca vitulina). Journal of Experimental Biology. 2011;214.

Wilson B, Hammond P, Thompson P. Estimating size and assessing trends in a coastal bottlenose dolphin population. Ecol Appl. 1999;9:288-300.

Winship A. Estimating the impact of bycatch and calculating bycatch limits to achieve conservation objectives as applied to harbour porpoise in the North Sea. PhD Thesis: University of St Andrews; 2009.

Yost WA. Fundamentals of hearing: An introduction. New York: Academic Press; 2000.

Supplementary Information

Table S1. Quantitatively calibrated levels of confidence taken from the IPCC guidance upon classification of uncertainty. Taken from Mastrandea et al. (2010).

Terminology	Degree of confidence in being correct
Very high confidence	At least 9 out of 10 chance of being correct
High confidence	About 8 out of 10 chance
Medium confidence	About 5 out of 10 chance
Low confidence	About 2 out of 10 chance
Very low confidence	Less than 1 out of 10 chance

Table S2. Overview of availability and quality of data available to support this assessment framework for the Moray Firth harbour seal population.

	Data Quality	Comments
1. Seal Distribution		
1.1. Harbour seal survey and tracking data	High	Integration of data from three different tracking studies conducted within the Moray Firth between 1989 and 2009
1.2 Habitat association model	Medium	Integrated dataset from 37 individuals to model habitat preference using a generalised additive model (GAM)
1.3 Spatial model of distribution	Medium	Use of GAM results and population estimates to predict probability of current seal density in each 4 x 4km grid square across the Moray Firth.
2. Noise Distribution		
2.1 Piling source levels and frequency characteristics	Very High	Robust knowledge base from Beatrice Demonstrator, other piling operations and engineering surveys carried out in the MORL and BOWL project areas
2.2 Noise propagation model	High	Established modelling approaches available, validated through measurements including the Beatrice Demonstrator project.
2.3 Spatial model of received levels	High	Established modelling approaches available.
3. Assess impact on individuals		
3.1 Identify thresholds for received levels that lead to:		
Non auditory injury	Medium	Based on data from human divers
PTS	Very Low	Southall et al. (2007) guidance based on TTS onset from a cetacean and TTS/PTS relationship in terrestrial mammals.
TTS	Low	Southall et al. (2007) guidance – pulsed noise TTS onset in a cetacean and the known pinniped-to-cetacean difference in TTS onset for non-pulsed noise.
Behavioural avoidance	Low	No empirical data available for seals. Southall et al. (2007) provide no guidance on behavioural disturbance from continuous pulsed such as piling. Alternative approaches such as dB $_{ht}$ (species) (Nedwell et al., 2007) not validated for seals. In absence of data, highly precautionary approach used.
3.2 Estimate # individuals:		
Non auditory injury	Low	No thresholds provided
Auditory injury	Low	Based on Southall et al. (2007)
Displaced/excluded	Low	As for behavioural avoidance.

		Data Quality	Comments
4. Assess impact on population			
	4. 1 Link individual impacts to demographic parameters	Very Low	No empirical data available for any sites to directly estimate nature and extent of links.
	4. 2 Harbour seal population model	Medium	Modelling frameworks available, but no empirical data for some key parameters.

Table S3. Summary of key assumptions used within our seal assessment framework. We express the level of confidence in each of these assumptions based upon the IPCC criteria in Table S1. We also provide a qualitative measure of the sensitivity of results to this assumption.

	Confidence	Sensitivity
A. Seal Distribution		
The movements of the sample of 37 tagged harbour seals are representative of the whole Moray Firth population.	Medium	Medium
Predictions from the habitat association model using these different data sources from 1989-2010 represent the current at-sea distribution of harbour seals, and represent distribution at all times of year.	Medium	Medium
75% of the population are assumed to be at sea at any particular time, with the remaining individuals associated with coastal haul-out sites.	High	Medium
B. Noise distribution		
Fleeing animals move away from the noise source at an average of 1.5 m/sec.	High	Medium
C. Assessment of impacts on individuals		
The probability of harbour seals being displaced can be based on the observed responses of harbour porpoises in the hour after pile driving ended at Horns Rev II.	Medium	High
Based upon porpoise data from Horns Rev II, animals are likely to be displaced for periods of up to 2-3 days after each piling event.	Medium	Medium
Thresholds for PTS-onset can be based upon experimentally derived TTS-onset thresholds for pulsed noise.	Very Low	High
The M-weighted SEL at which PTS onset occurs in harbour seals is 186 dB	Very Low	High
A generalised PTS dose-response curve for pulsed noise can be based upon an extrapolation of Finneran et al's (2005) dose-response curve for intermittent tones.	Low	High

	Confidence	Sensitivity
D. Linking individual impacts to demographic parameters		
Direct injury and death at close range can be avoided through established mitigation measures	High	Low
PTS fitness consequences are expressed as an 25% additional mortality risk in the year of exposure	Very Low	High
Behavioural displacement fitness consequences can be expressed as a reduction in fecundity.	Low	High
There is a direct linear relationship between the amount of the year that individuals are displaced from foraging areas and consequent reduction in reproductive success.	Very Low	Medium
E. Harbour seal population model		
Estimates of fecundity and adult survival from Loch Fleet are representative for the whole Moray Firth population	Very high	Low
Pup and juvenile rates can be based upon published dataset from the Kattegat-Skaggerak	Medium	Low
There is an equal sex-ratio	Medium	Low
Reproduction is density-dependent	High	High
The form of density dependent reproduction can be described by Equation 3 in Taylor and DeMaster 1993.	Medium	Medium
The carrying capacity is fixed at 2000, 20% above the maximum abundance estimate since 1990	Medium	Medium