moray offshore renewables ltd

Environmental Statement

Technical Appendix 7.3 B - Framework for assessing the impacts of pile-driving noise from offshore wind farm construction on Moray Firth harbour seal populations

Telford, Stevenson, MacColl Wind Farms and associated Transmission Infrastructure Environmental Statement

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Document Owner	**Nancy McLean**				
Document Status	**Final**				
File Name					
Revision	**Date**	**Description**	**Originated By**	**Checked By**	**Approved By**
A1	November 2011	For review	GH/PT	SJC	NM
A2	May 2012	Final Draft	GH/PT	SJC	NM
A3	June 2012	Final			

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Framework for assessing the impacts of pile-driving noise from offshore wind farm construction on Moray Firth harbour seal populations

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12th December 2011

1. Background

Many European offshore wind farm sites are used as foraging areas by harbour seals. In several cases, proposed developments are also close (<100km) to harbour seal Special Areas of Conservation (SAC), meaning that they are likely to require Appropriate Assessments under the EU Habitats Directive.

The key potential impacts of offshore wind farm construction upon harbour seal populations are recognised to be:

1. Direct impacts of piling noise or other activities during the construction phase, potentially causing direct injury or displacement of seals.
2. Indirect impacts through long-term alteration of habitat. These may be either negative (loss of habitat) or positive (reef effects or changes in fishing activity).
3. Disturbance or barrier effects resulting from operational turbines or maintenance vessels.

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.

To obtain project consents, regulator guidance highlights that to meet the requirements of the EU Habitats Directive; developers must provide them with information that allows them to:

- Determine whether or not the proposal is directly connected with or necessary to site management for conservation; and, if not,
- Determine whether the proposal is likely to have a significant effect on the site either individually or in combination with any other plans or projects; and if so, then
- Make an appropriate assessment of the implications (of the proposal) for the site in view of that site's conservation objectives.

In addition, if pile-driving activity has the potential to disturb harbour seals at their haul-out sites, a licence to disturb may now also be required under the Marine (Scotland) Act 2010.

Two offshore wind developments have been proposed in the Moray Firth and hold agreements with the Crown Estates. The Moray Offshore Renewables Ltd (MORL) project is a 1.5GW development located on the Smith Bank a minimum of 12nm from shore leased under the Crown Estates Round 3 programme. The Beatrice Offshore Wind Limited (BOWL) project is a 1GW development located adjacent to the MORL project within the 12nm limit leased under the Crown Estates Scottish Territorial Waters (STW) programme. Construction of the projects is proposed to commence in 2014/2015 which would allow both projects to be fully commissioned by 2020.

The aim of this document is to provide a framework that can be used to assess the significance of impacts from pile-driving noise during construction of the MORL and BOWL offshore wind farms in the Moray Firth. Given current understanding of harbour seal movements, the harbour seal SAC of concern for these developments is the Dornoch Firth and Morrich More SAC.

It is recognised that there is a benefit to the offshore renewable industry in taking a consistent approach to assessment of impacts to allow a more robust understanding of cumulative impacts. It is therefore hoped that the development of this framework, in an area where there is a relatively high level of scientific understanding of harbour seal population ecology, may help support assessments in other areas, or potentially for other species. However, it is recognised that the adoption of this approach outside of the Moray Firth would require support from the regulators and their advisors, in addition to other developers. Further discussion is presented later in this document on the applicability and limitations of the framework, both to other species and to sites out with the Moray Firth.

The conservation objectives for the Dornoch Firth and Morrich More SAC's harbour seal interest consider various key attributes, including the population using the site, the distribution of animals within the site, the distribution of habitats (within and without the site) that supports this population, and levels of disturbance to the population. The aim of this framework is to predict the long-term population level impacts of piling activity so that this information can be used to assess potential impacts on these conservation objectives and the integrity of the SAC, and the Favourable Conservation Status (FCS) of the wider population.

2. General Approach

Our general approach for assessing a development's impact on the SAC conservation objectives and the population's FCS is illustrated in Figure 1. Assessments involve four main elements; a description of spatial distribution patterns of both seals and noise, the integration of this information with available data on the potential impacts of noise to assess the numbers of individuals impacted and, finally, the use of these data in a population model to predict longer-term population level impacts.

Whilst several elements of this approach are comparable to previous offshore wind farm assessments, a major development is the use of population modelling to predict the long-term consequences of these new activities. We argue that this is an essential extension to previous assessments given the guidance that assessments of FCS must consider whether or not protected populations are maintaining themselves ***in the long term*** (Annex II, EU 2010). Crucially, the incorporation of a population model permits an exploration of any potential interactions with other cumulative impacts, the sensitivity of different assumptions made to produce the framework, and comparison of different development or mitigation scenarios. Whilst based on the best available scientific data, it also offers potential to update key parameters or relationships should new data become available.

In this context we suggest that "long-term" be considered to be a 25 year time-scale. First, this is the time-scale typically considered by the IUCN when assessing conservation status. Second, it is equivalent to approximately 1-2 times the generation time for harbour seals, and thus seems an appropriate period for assessing longer-term population change. We recognise that it is a legal requirement for SAC site condition monitoring assessments to be made every 6 years. These more regular assessments would therefore need to be interpreted in the context of likely patterns of longer-term population change. However, this issue already exists because the monitoring programmes for harbour seals and many other protected species do not have sufficient power to provide robust assessments of population status over a 6-year reporting window (see power analyses in Thompson *et al*. 1997, Thompson *et al*. 2000 & Wilson *et al*. 1999). Thus, a 25 year timescale should be appropriate for assessing the long-term status of long-lived species such as harbour seals on both ecological and statistical grounds.

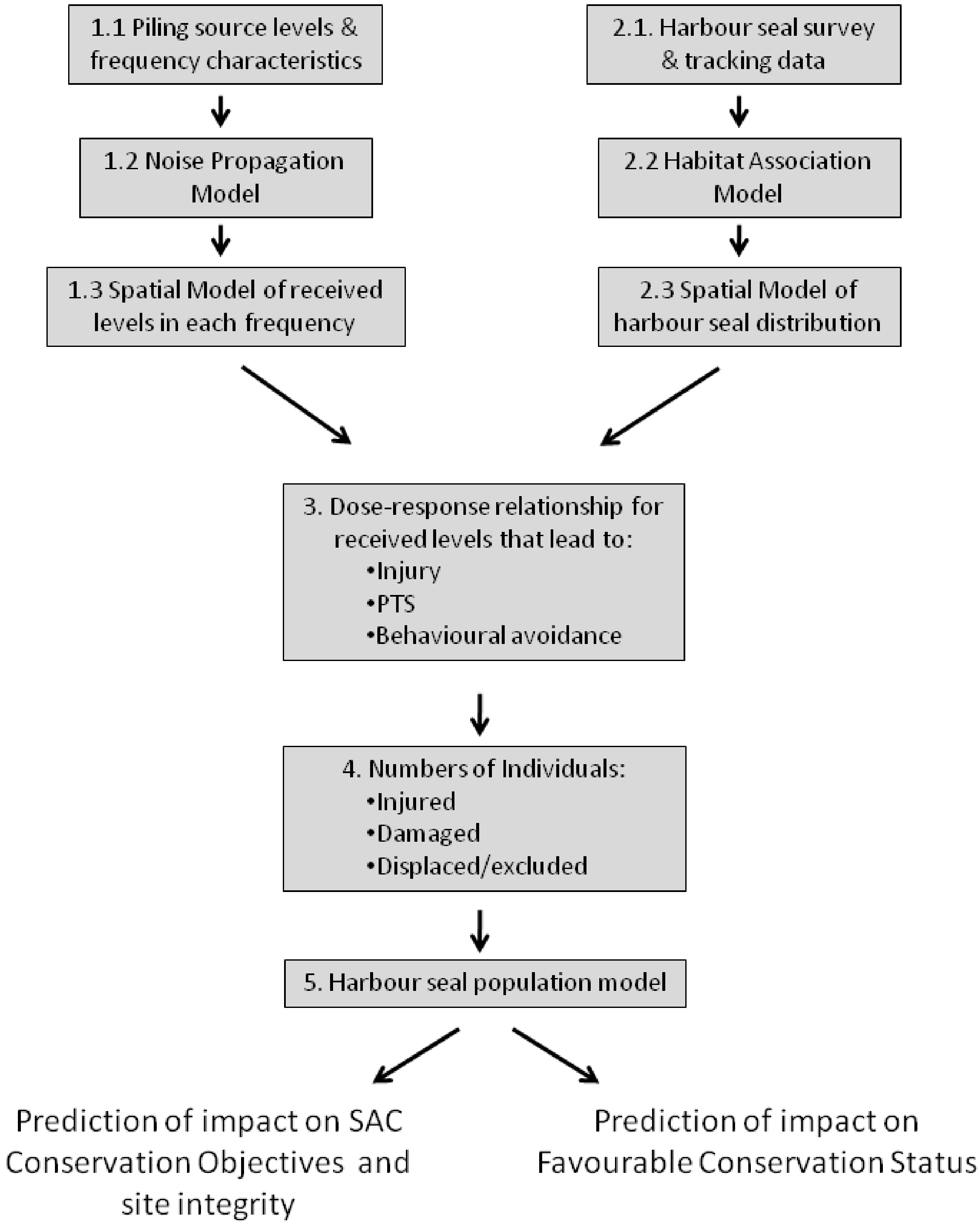


Figure 1: Schematic of the approach proposed for assessing the impact of wind farm construction on the harbour seal SAC and FCS.

3. Structure and application of the assessment framework

The following sections outline how this framework will be used to assess the impact of the construction of the MORL and BOWL wind farms on the Moray Firth harbour seal population. In this context, as advised by JNCC and Marine Scotland Science, we consider the regional Moray Firth population to be equivalent to the Marine Scotland seal management area for the Moray Firth.

Our aim is to outline our generic approach for assessments that will be made for different construction scenarios within future environmental assessments. In this document we illustrate the framework using data from pile-driving activities experienced during the construction of the Beatrice Demonstrator, and scenarios involving construction of a hypothetical wind farm using analogous construction techniques.

Each of the following sections outlines our approach to dealing with each element of the framework illustrated in Figure 1.

3.1 Seal Distribution

This element of the framework requires information on spatial variation in the density of harbour seals across the region.

3.1.1 Harbour seal survey & tracking data

Annual surveys of harbour seals in this region mean that current information is available on the number of individuals at haul-out sites within the Dornoch Firth & Morrich More SAC (the nearest harbour seal SAC to these developments), the Loch Fleet NNR (the nearest harbour seal breeding site to these developments) and the wider Moray Firth population with which these animals interact (see Thompson *et al*. 2007, Cordes *et al*. 2011). Using estimates of the respective proportion of time seals spent hauled out and in the water developed during research on this same population (Thompson *et al*. 2007) we can inflate these survey counts to estimate the total number of individuals within the population. In 2010, the mean haul-out count for the inner Moray Firth was 721 (SMRU Unpub. Data), which represents a total population size of 1,183 (95% CL = 1027-1329).

Information on the foraging distribution of seals from this population is based upon an integration of data from three different tracking studies which were carried out between 1989 and 2009. Further information on these data and the techniques used to standardise them are given in Bailey & Thompson (2011).

3.1.2 Habitat Association Model

We used this integrated dataset from 37 individual harbour seals to model seal occurrence and habitat preference using a generalised additive model (GAM) as described in Bailey & Thompson (2011). This GAM used a presence-absence approach across a 4x4km grid, and found a significant relationship between seal presence and depth, slope and distance to the nearest haul-out site.

3.1.3 Spatial Model of distribution

We then used the results of the GAM to predict the probability of seal occurrence in each of the 4x4km cells across the Moray Firth. The percentage of the population in each cell within the Moray Firth was then estimated by dispersing the whole population across this density surface in relation to the predicted importance of this cell (Figure 2). The number of seals predicted to be in each cell can then be estimated for different population sizes using this distribution. Depending upon the impacts being considered, one can assume either that all the seals that potentially use a grid square may be impacted (e.g. behavioural exclusion) or that only those individuals at sea at that time were impacted (eg. PTS). In the latter case, based upon data from the Moray Firth (Thompson *et al*. 1998), we assume that harbour seals typically spend around 75% of their time at sea. This is conservative because this value is known to be lower during the breeding season, and the approach could in future be developed to account for variation in haul-out frequency according to season or other factors such as age and sex (e.g. see Härkönen *et al*. 1999 and Thompson *et al*., 1998).

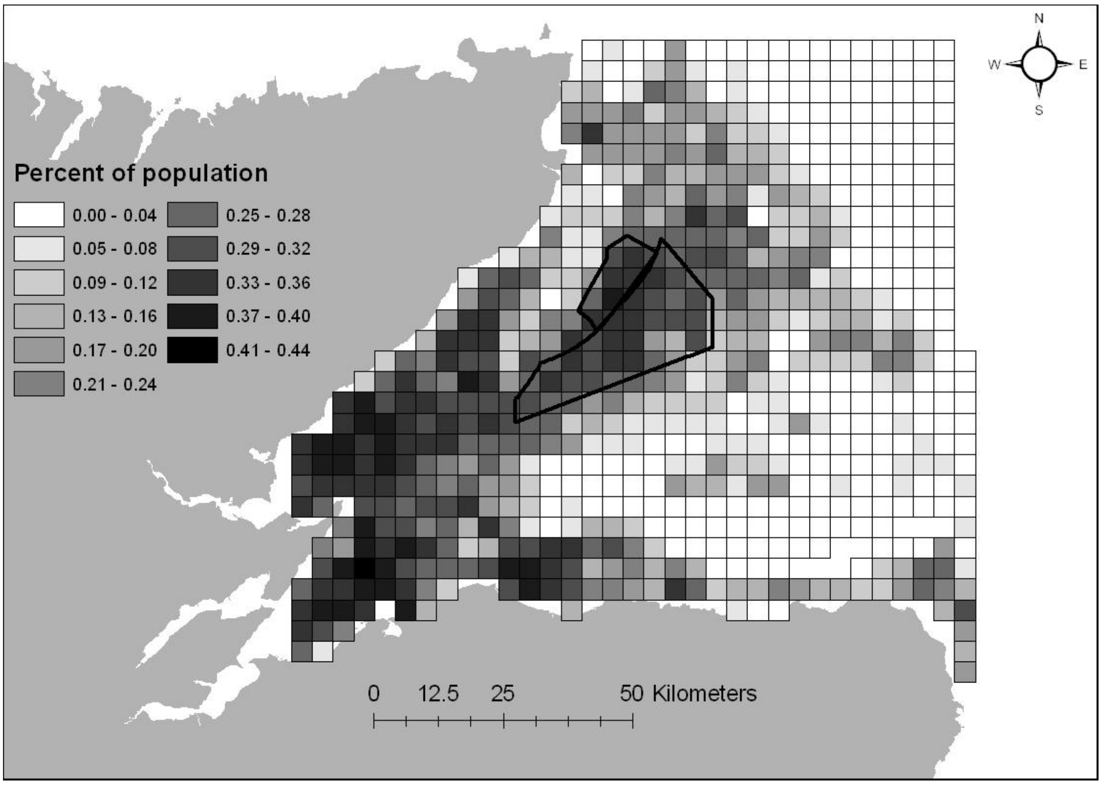


Figure 2: At-sea distribution of harbour seals in the Moray Firth. Data are based on the habitat association analysis in Bailey & Thompson (2011), and show the percentage of the population that is expected to be found in each of the different 4x4 km grid cells.

3.2 Noise Distribution

3.2.1 Piling source levels & frequency characteristics

The source levels and frequency characteristics of the modelled piling operations have been taken from published reports from the Beatrice Demonstrator scheme that was constructed within the Moray Firth in 2006. The 1.8m diameter piles were pile-driven into the seabed using a 500kJ hammer, and the predicted dB_{ht} and Sound Exposure Level (SEL) (Section 2.3) contours have been modelled using the blow energies recorded for the driving of the two pin piles that were installed on the 21st of July 2006.

3.2.2 Noise Propagation Model

The predicted propagation of the noise resulting from the piling operations required for the construction of the wind turbine foundations was modelled using the INSPIRE model. This model uses a combined geometric and energy flow/hysteresis loss model to predict propagation the relatively shallow coastal environments which are typical of wind farm locations such as those in the Moray Firth.

Comparison of INSPIRE model predictions with published measured recordings from the Beatrice Demonstrator (Bailey *et al*. 2010) indicate that the model predictions for unweighted peak levels provide a relatively good fit of the measured data and provide a conservative prediction of sound levels across the wider Moray Firth (Figure 3).

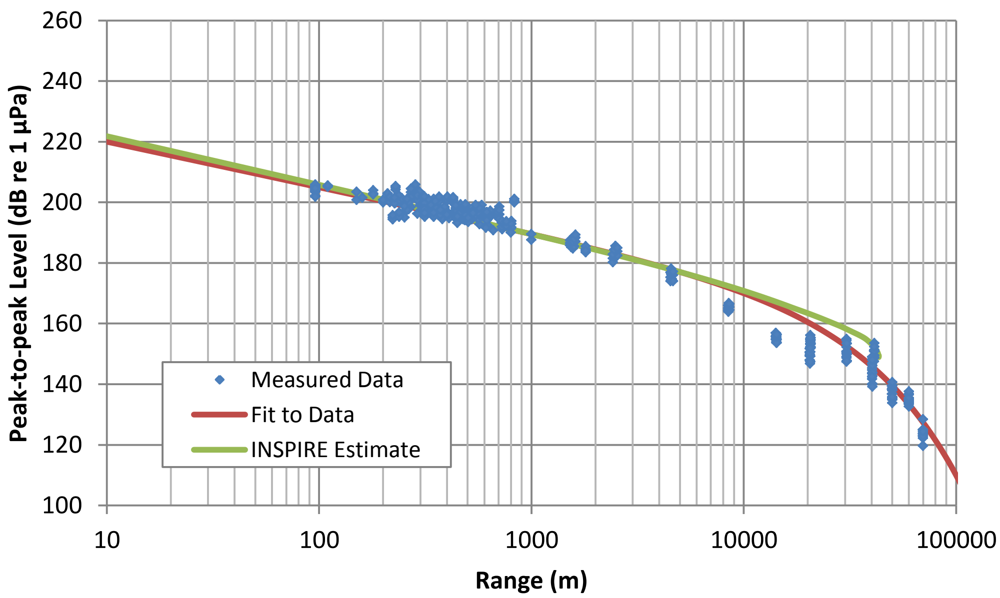


Figure 3: The level of sound in un-weighted peak to peak levels as a function of range in meters for a 1.8 m diameter pile at the Beatrice demonstrator site. Predictions from the INSPIRE model are presented alongside measured data from Bailey *et al*. (2010)

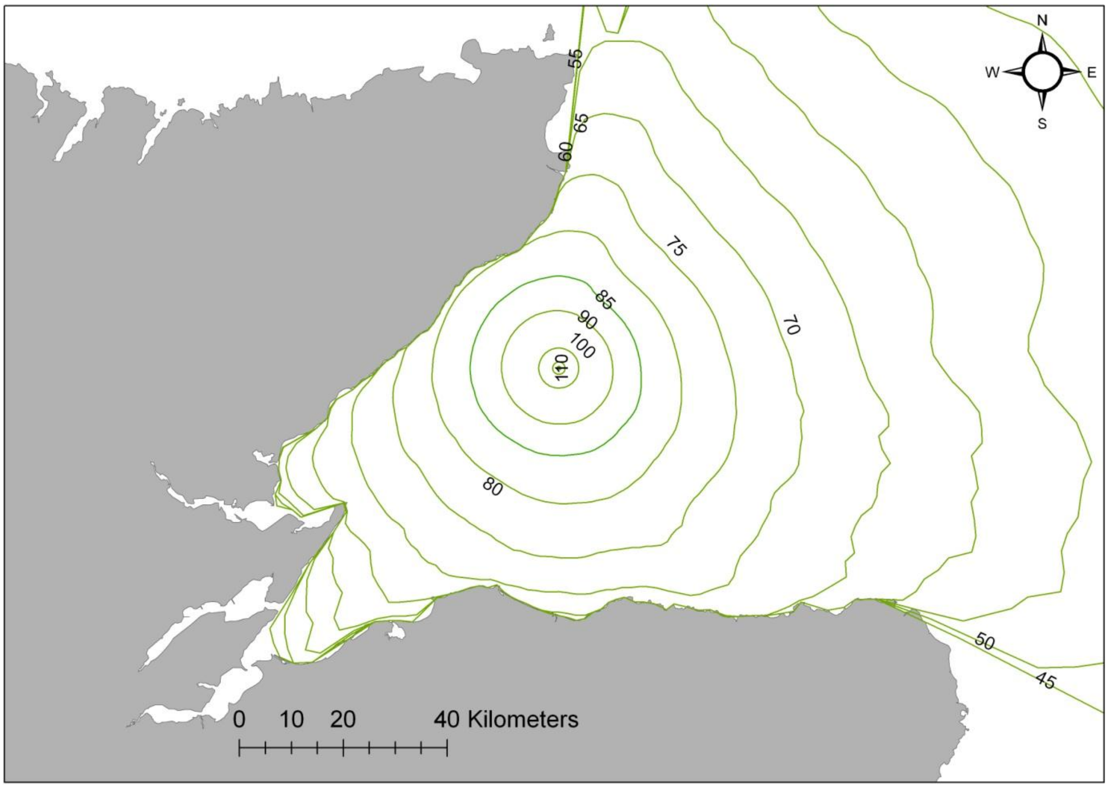
3.2.3 Spatial Model of received levels

Based upon this pattern of noise propagation, the INSPIRE model was used to predict received noise levels in different parts of the Moray Firth from the piling activities associated with wind farm construction. In our example here, we again used the blow energies and strike rates required to drive 1.8m diameter pin piles as used for the Beatrice Demonstrator (see Bailey *et al*. 2010). For calculations of Sound Exposure Level (SEL), INSPIRE was used to model two such pin piles driven within a 24hr period.

Received noise levels were frequency weighted to account for the characteristics of harbour seal hearing. Two different weightings were used; first, dB_{ht} values were calculated based upon published data on the harbour seal audiogram (see Nedwell *et al*. 2007). Second, M-weighted values were calculated based upon the approach proposed for all pinnipeds in Southall *et al*. (2007). Spatial variation in received noise levels was expressed as a series of contours representing the point within which a particular threshold (e.g. 90 dB_{ht} or an M-weighted SEL of 198 dB) was exceeded.

Discussion with the Statutory Nature Conservation Agencies (SNCA) highlighted uncertainties over the most appropriate threshold to use for these assessments. Furthermore, it is clearly unrealistic to expect all animals in a population to respond in exactly the same way to a particular noise threshold level (see section 3.0). We therefore used INSPIRE to model three sets of contours, one for received levels using dB_{ht} metric, the other two used M-weighted SEL as a metric (one calculated for fleeing animals as requested by JNCC, the second calculated for stationary animals as requested by SNH). In the first case, dB_{ht} contours were generated at 5 dB_{ht} increments, between 25 dB_{ht} and 130 dB_{ht} (Table 1). In the other cases, M-weighted SEL contours were generated at key levels of relevance to the Southall *et al*. (2007) criteria, and at regular 5 dB increments within the range of values in which PTS-onset might occur (Table 1).

These outputs were generated as GIS shape files and used within ARC GIS to assess the maximum received levels in each of the 4x4 km grid cells for which there were predictions of seal density (Figure 2). An example which uses dB_{ht} as a metric, and shows the output from INSPIRE and the resulting values for each grid cell, is shown in Figure 4.



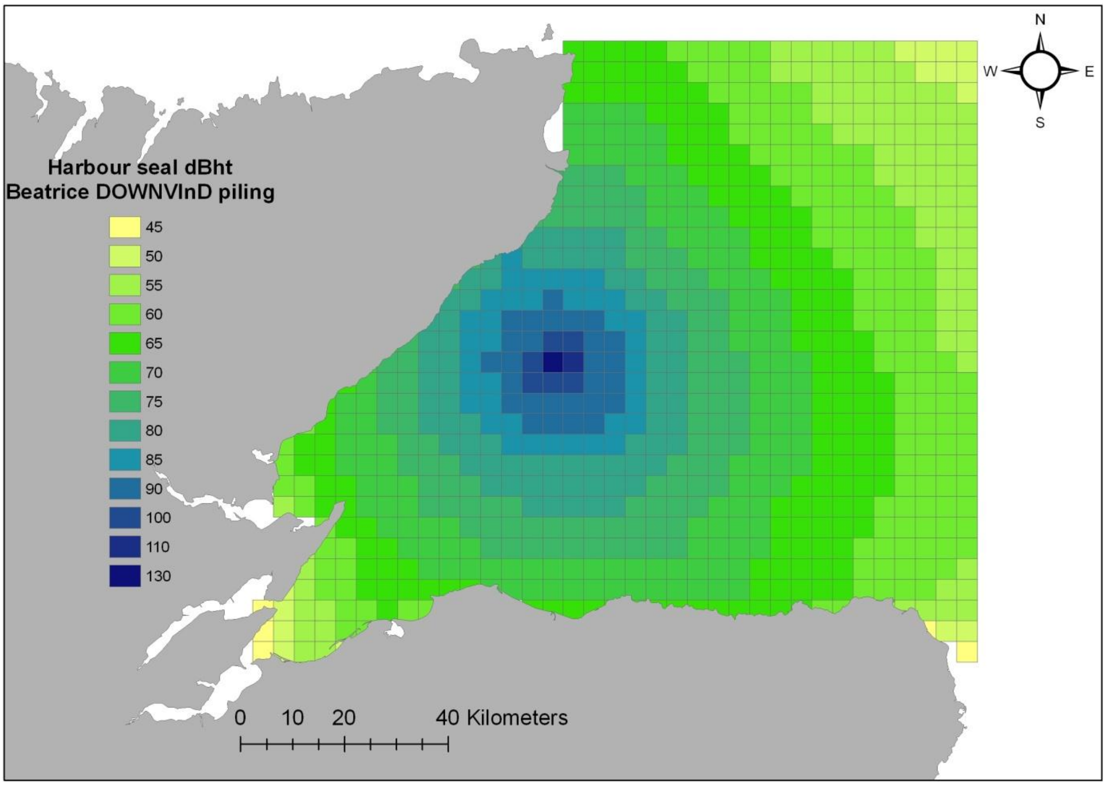


Figure 4: Top) contour map showing the output generated from the Subacoustech INSPIRE model; Bottom) map showing variation in received levels across the 4x4 km grid cells.

Table 1: Received noise levels that were modelled using INSPIRE. Shapefiles representing areas in which each of these levels was exceeded were then used within ARC GIS to determine the maximum received levels within each 4x4 km grid cell (see Figure 4).

Received noise levels	
dB $_{ht}$	**M-weighted SEL**
220	220
130	218
120	213
110	208
100	203
90	198
85	193
80	188
75	186
70	
65	
60	
55	
50	
45	
40	
35	
30	
25	

3.3 Assessment of impacts upon individuals

There is widespread interest and concern about the impacts of underwater noise on marine mammals, but very little empirical data available to underpin any predictions about the likely impact of particular developments. This is because it is incredibly difficult to collect the empirical data one ideally requires to parameterise models that could predict impacts at the population level. Frameworks for understanding the consequences of acoustic disturbance are currently being developed in response to the recommendations of a National Research Council Committee (NRC 2003). Using case studies from a small selection of exceptionally well-studied marine mammal populations; the Population Consequences of Acoustic Disturbance (PCAD) project is using a state-space modelling approach to explore how disruptions to normal behaviour patterns may impact individual fitness and, ultimately, population dynamics. These approaches show great promise, but it will be several years before they will be at a stage where they can be used in a generic fashion to support appropriate assessments for offshore developments.

The most complete review of other studies in this area can be found in Southall *et al*. (2007). This work draws on the deliberations of several years work by a series of inter-disciplinary expert review groups that aimed to develop noise exposure criteria to support the implementation of the US Marine Mammal Protection Act. The resulting exposure criteria are now being used widely elsewhere in the world. When doing so, however, it should be noted that the authors recognise that these criteria are often highly precautionary and typically based upon very sparse data (Southall *et al*., 2007). Crucially, data on behavioural responses are so limited that "*insufficient information exists to assess the use of SEL as a relevant metric in the context of marine mammal behavioural disturbance for anything other than a single pulse exposure*" (Southall *et al*., 2007).

Given that a key impact of wind farm construction is behavioural disturbance from extended periods of pile-driving, our environmental assessments cannot be based upon the Southall *et al*. (2007) criteria alone. Even if ongoing work is successful in addressing some of these gaps, other approaches are required if assessments are to be carried out within the time-frame required to meet the EU 2020 carbon reduction targets.

One alternative approach to assessing the impacts of anthropogenic noise, which focuses on behavioural responses, is the use of dB_{ht} 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 received levels of sounds in the frequency bands which animals are most likely to hear and respond to, or in essence the "perceived loudness" of the noise to the animal. Similarly, several research studies use an equivalent approach by estimating "sensation levels" that represent received levels, frequency-weighted according to the study species' hearing ability (eg. Gotz & Janik 2010).

Given that there are uncertainties surrounding both Southall *et al*'s (2007) M-weighted criteria and Nedwell *et al*'s. (2007) dB $_{ht}$ 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 level. The sections below explain how these values have been selected.

3.3.1 Identify thresholds for received levels that lead to behavioural avoidance, PTS and injury

All assessments of the impact of noise on marine mammal populations recognise that potential effects fall into three major categories (non auditory injury, auditory injury, and behavioural). These can each be further sub-divided depending upon the severity of the effect within each of these, as summarised in Table 2. Further details can be found in Southall *et al*. (2007).

Whilst there is general agreement on this hierarchy of effects of noise upon marine mammals, there is much more uncertainty, and consequently less agreement, on the received noise levels at which these effects might occur. These issues are discussed in the sections below.

Table 2: Potential effects of noise upon marine mammals.

1. Lethality & 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 high/prolonged exposure to lower levels of noise
• Temporary Threshold Shift	Temporary elevation of hearing threshold.
3. Behavioural Effects	
• Avoidance	See Annex I for Southall's *et al*'s more detailed breakdown of behavioural effects
• Changes in foraging or social behaviour	

Behavioural avoidance.

In future, the PCAD project and the related IWC Large Scale Whale Watching Experiment (Lusseau 2010) will better inform this issue. In the meantime, we address the need to assess long-term population level impacts using a simpler deterministic approach that can be used to compare the impact of different scenarios, focusing primarily on behavioural changes that may result in avoidance of impacted areas. This may underestimate impacts from more subtle behavioural changes, but we suggest that this will be balanced by our use of conservative assumptions about the consequences of behavioural avoidance (see section 4.1).

In the absence of exposure criteria to prevent behavioural disturbance in Southall *et al*. (2007), this part of our framework draws upon the dB $_{ht}$ approach developed by Nedwell *et al*. (2007). 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 $_{ht}$ and milder reactions to levels of 75 dB $_{ht}$ and above (Table 3).

Table 3: Assessment criteria used by Subacoustech to assess the potential impact of underwater noise on marine species (from Nedwell *et al*. 2007).

Level in dB$_{ht}$(species)	Effect
0 – 50	Low likelihood of disturbance
75 and above	Mild avoidance reaction by the majority of individuals but habituation or context may limit effect
90 and above	Strong avoidance reaction by virtually all individuals
Above 130	Possibility of traumatic hearing damage from single event

However, one criticism of the dB $_{ht}$ approach is that these behavioural response criteria remain untested for marine mammals. A further issue is that individuals in wild populations are unlikely to respond at consistent received levels, and it is more appropriate to consider responses in terms of a curve that describes the relationship between sound level and the proportion of animals predicted to respond rather than a simple step-change threshold (eg.of 90 dB $_{ht}$).

To support the development of our assessment framework, we carried out an initial test of the threshold values used by the dB $_{ht}$ approach (Table 3) using published passive acoustic monitoring data (using C-PODS) on the extent to which porpoises responded to pile driving activity at Horns Rev 2 (from Brandt *et al*. 2011).

To estimate variation in the level of behavioural response, data from Brandt *et al*. (2011) were used to model changes in the occurrence of porpoises in relation to predicted received sounds levels resulting from a nearby piling event. This peer-reviewed publication provides data on the proportional change in the detection of porpoises on C-PODs moored at different distances from a piling event at Horns Rev 2. This proportional change was based upon the difference between a baseline period and data collected during the hour after piling. We used these data to model the extent of the proportional change with distance by fitting a binomial relationship to the data (Figure 5). We then took published data on the size of the pile, together with information on local bathymetry, and used INSPIRE to estimate received dB $_{ht}$ levels for harbour porpoise at each of the C-POD sampling sites at Horns Rev 2. In Figure 5, the most parsimonious fitted relationship is shown as a solid line, and a more precautionary relationship that is weighted to include the higher response levels is shown as a dashed line. The precautionary relationship from Figure 5 was then used to predict the response of animals at different received noise levels using dB $_{ht}$ (Figure 6).

Although the number of data points used is small, the fitted relationship in Figure 6 generally supports the definitions of the threshold values in Table 2, as proposed by Nedwell *et al*. (2007). In the absence of similar empirical data for harbour seals, we use the relationship in Figure 6 as a proxy for harbour seals, assuming that this relationship holds for similar values of dB $_{ht}$ for harbour seals.

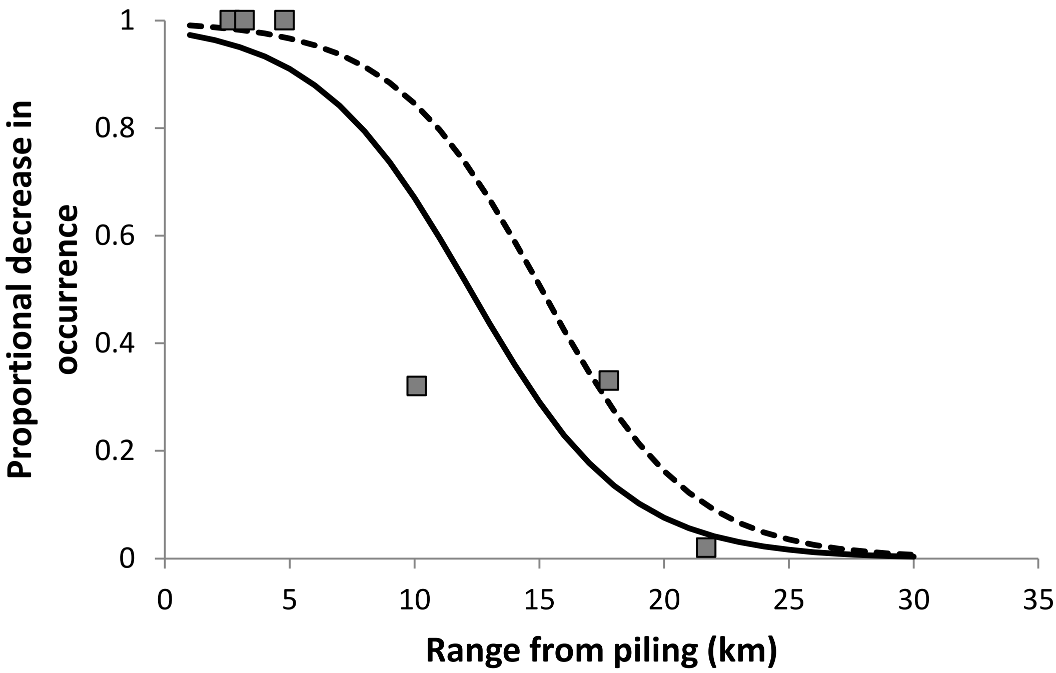


Figure 5: Predicted relationship between range from the Horns Rev piling and the proportional decrease in harbour porpoise occurrence (mean porpoise positive minutes from CPODs (from Brandt *et al*., 2011)) before and in the hour after the event; the figure shows the line of best fit (deviance = 4.19, df=1, P<0.05). Intercept=3.9146 (se=2.7666), Range=-0.3205 (se=0.2248).

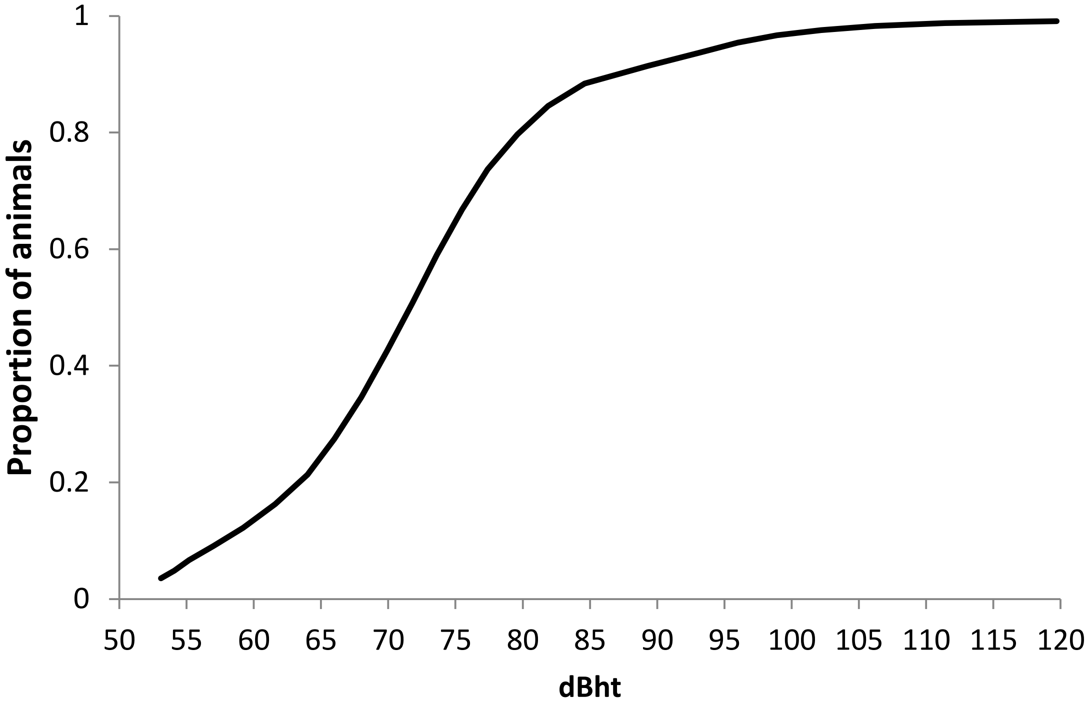


Figure 6: The relationship between dB_{ht} for harbour porpoise and the predicted proportion of animals excluded from the area (using the precautionary relationship from Figure 5).

PTS-onset.

In humans, it is well known that prolonged exposure to loud noise can cause permanent auditory damage and hearing loss (NRC 2006). Whilst the processes remain unclear, similar effects are expected to occur in marine mammals. However, assessments of the noise levels likely to result in permanent threshold shifts (PTS) in marine mammals depend upon a series of assumptions and the use of proxy data from other mammals. Given that one cannot experimentally induce PTS, noise exposure criteria for PTS-onset are based upon assumed relationships between the relative levels of noise likely to cause Temporary Threshold Shifts (TTS) and PTS.

Using this approach, 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. As detailed in a separate document (Thompson & Hastie 2011 in Technical Appendix 7.3 E), we argue that there is insufficient evidence to support Southall *et al*.'s (2007) proposal for different PTS-onset criteria for cetaceans and pinnipeds, and highlight that no studies have been carried out at high enough levels of pulsed noise to induce TTS in pinnipeds. In this framework, we therefore use an M-weighted PTS-onset threshold of 198 dB, taken from the only studies available to Southall *et al*. (2007) in which exposure to pulsed noise induced TTS in marine mammals. Experiments are currently being planned to provide better data for pinnipeds, but these will not be carried out until at least mid 2012 (Southall Pers Comm) and additional peer-reviewed data will therefore not be available to inform our assessment process.

As discussed in relation to behavioural impacts (see Figure 6), PTS is not likely to occur at the same noise threshold in all individuals or circumstances, and we would expect an increasing likelihood of PTS in relation to the noise dose. It is important to note that the PTS-onset criteria proposed by Southall *et al*. (2007) represents the noise levels at which these effects ***start*** to occur. This is illustrated in Finneran *et al*. (2005), who produced a dose response curve by assessing the proportion of trials at different Sound Exposure Levels (SEL) that resulted in TTS. As highlighted by these authors, one would have to extrapolate this curve well beyond the range of measured data (and 11dB above the TTS-onset level) to reach the point where 50% of the population were predicted to experience TTS (Finneran *et al*. 2005).

In contrast, the common assumption by many stakeholders, and in many environmental assessments, is that all animals within the PTS threshold will experience PTS. A more realistic approach has been taken in the SAFESIMM model, developed at the University of St Andrews as part of the Environmental Risk Management Capability (ERMC) assessment framework developed to support planning of Naval exercises (Mollet *et al*. 2009). This model uses a theoretical dose-response curve for PTS (which is scaled from the TTS dose-response curve in Finneran *et al*. (2005)), where the probability of animals experiencing PTS increases from a SEL of 198 dB up to 250 dB; the point at which all animals are predicted to have PTS. SAFESIMM is currently being adapted to support the management of marine renewable energy developments and could in future be used within our framework to provide the most robust estimates of the number of animals exposed to PTS. In the meantime, we use their proposed PTS dose-response relationship (as shown in Figure 7) to provide an indication of the proportion of animals within the PTS-onset threshold that are exposed to levels believed to cause PTS.

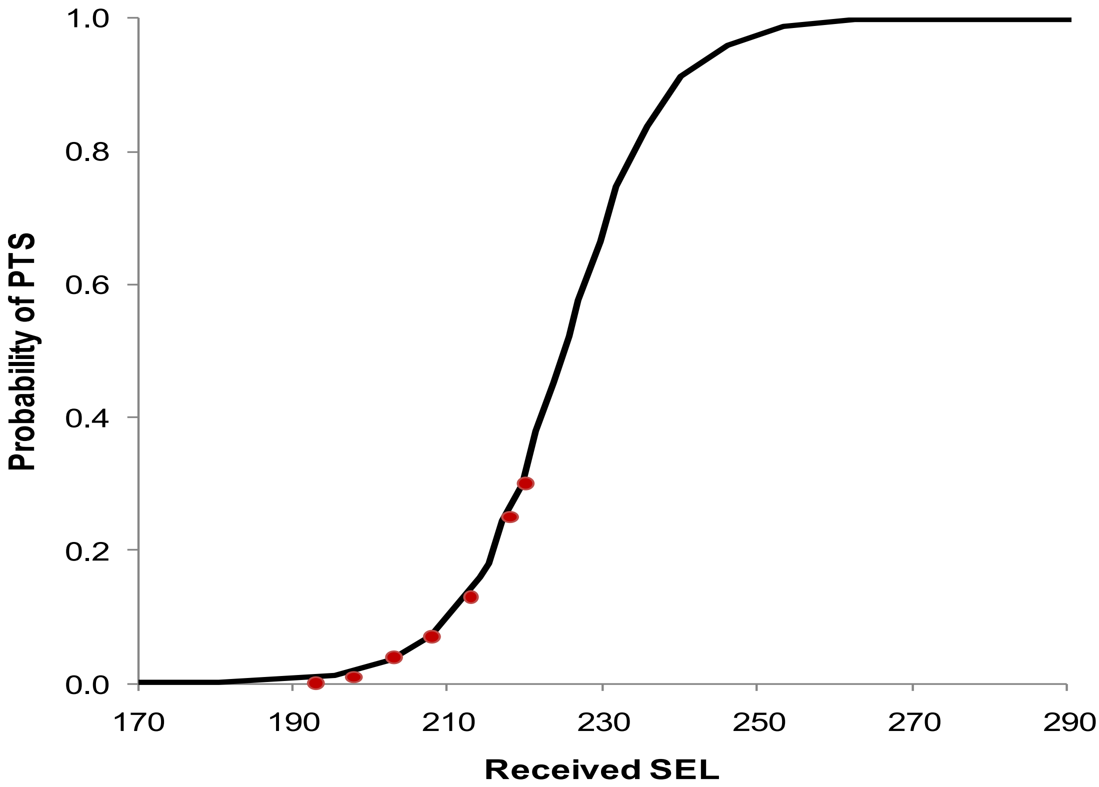


Figure 7: Generalised PTS dose-response curve for a PTS-onset value of 198 dB, as used within SAFESIMM. Points in red correspond to the SEL contours generated using INSPIRE (see Table 2).

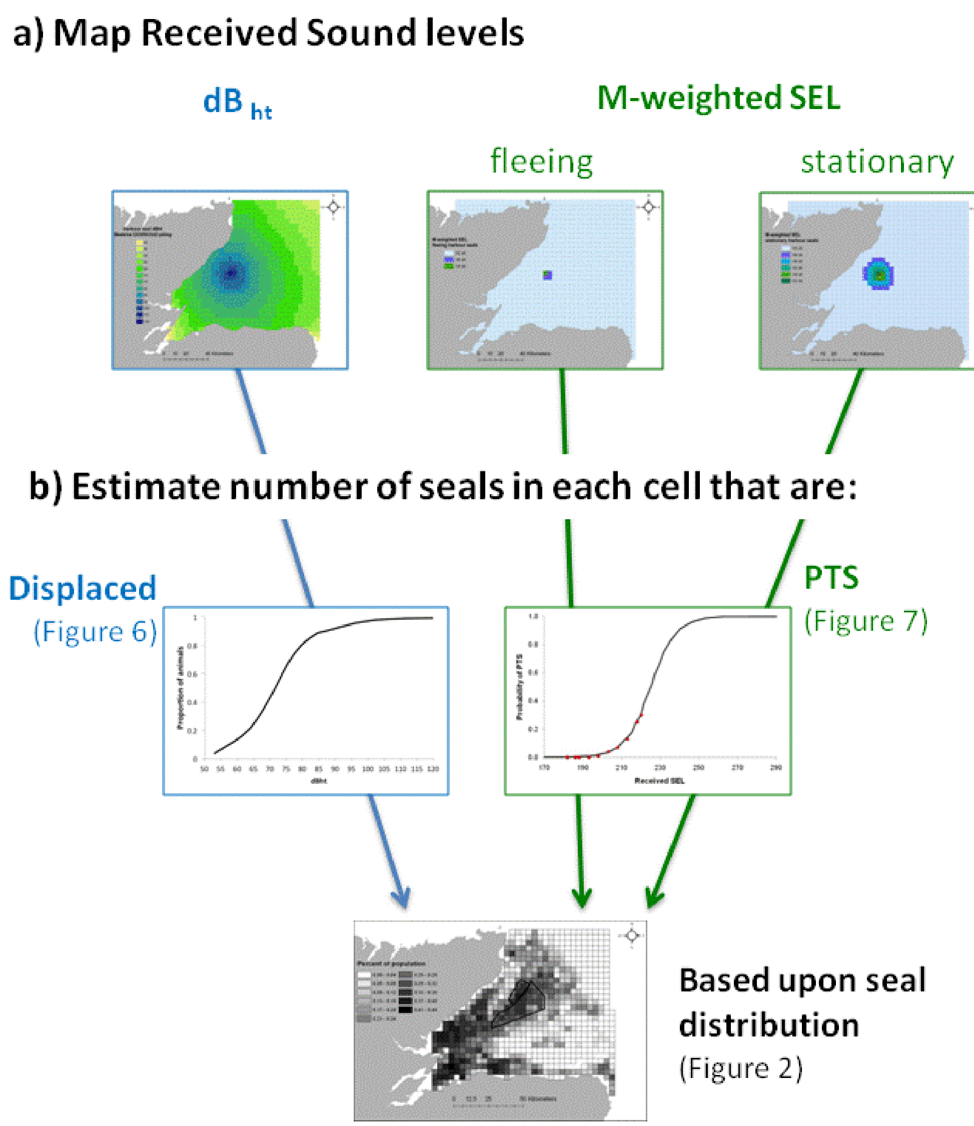
3.3.2 Estimate the number of individuals injured, damaged or displaced/excluded

To estimate the number of individual seals that would be exposed to injury, PTS or behavioural displacement, we used the thresholds and relationships presented in Table 4 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 (as estimated for different population sizes from the data presented in Figure 2).

Table 4: Proposed thresholds for different effects of noise upon harbour seals. With the exception of those for behavioural disturbance (see Fig. 5), M-weighted values are taken from Southall *et al*. (2007) and dB ht values are taken from Nedwell *et al*. (2007).

Effect	M-weighted SEL Threshold	dB $_{ht}$ Threshold
Immediate death		240 dB (unweighted)
Physical Injury		220 dB (unweighted)
Permanent Threshold Shift	See Fig 7	
Behavioural Avoidance		See Fig 6

This process is illustrated in Figure 8, where we estimate the number of harbour seals that may be displaced or suffer from PTS displaced as a result of driving the Beatrice Demonstrator's 1.8m piles. Figure 8a presents the maximum received levels in each cells both using dB_{ht} as a metric, and using the two different estimates of M-weighted SEL; one based on a fleeing model the other on a stationary model. 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 (Figure 6) and PTS (Figure 7), and sum these proportions to provide the total number of individuals affected.



c) Sum to estimate total number of animals affected

Behavioural Displacement	PTS	
	Stationary Model	Fleeing Model
615	<1	<1

Figure 8: 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 PTS from a single event involving the installation of two 1.8m piles in 24hrs.

3.4 Assessment of impacts upon the population

3.4.1 Link individual impacts to demographic parameters

The approach outlined in Figure 8 provides information on the number of individuals displaced or vulnerable to PTS. However, to assess the long-term population level effects of longer period of piling during wind farm construction, we must next make assumptions about how the effects outlined in Table 4 might influence demographic parameters. This is especially challenging because there are no empirical data available from any marine mammal population to directly relate these individual effects to changes in reproductive and survival rates. We therefore use a conservative approach to provide realistic worst case scenarios. The basis of these assumptions is discussed below and our proposed realistic worst case scenarios are summarised in Table 5.

Death & non-auditory injury.

The clearest links are the direct effects on mortality at very close range. INSPIRE modelling indicated that received levels from the installation of the 1.8m piles only exceeded 220 dB_{ht} within <50m. Whilst these are potentially major impacts at close range, they will be mitigated against using standard procedures. This assessment therefore focuses on the less direct effects of PTS and behavioural avoidance.

PTS.

Harbour seals have extremely sensitive vibrissae which allow them to follow hydrodynamic trails from prey (Dehnhardts *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 seals have been disturbed, there could even be positive fitness consequences from reduced foraging competition. In those parts of their range where predation is high, the harbour seal's hearing ability does underpin differential responses to those groups of killer whales most likely to take seals (Deeke *et al*. 2002); thus, a decrease in hearing sensitivity could increase the seals risk of predation. However, killer whales are rarely encountered in the Moray Firth and we suggest that it is most 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, we suggest that it is unlikely that a reduction in hearing ability within part of the hearing range would significantly reduce reproductive success, given that males use a series of other visual and geographical cues also 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 is general stakeholder agreement that assessments of population level impacts should take account of this.

In the past, some environmental assessments have assumed 100% mortality for all animals that were exposed to SEL above the PTS-onset threshold. We suggest that this is inappropriately conservative, and instead propose that a) the PTS dose-response curve should first be used to estimate the number of animals that may have PTS and b) those individuals should then be subjected to an additional mortality risk factor. In the absence of any data that provide direct information on the mortality risk of PTS, our framework assumes that this is likely to be of a similar magnitude to the impact of old age. Information on age-specific survival in wild mammals is rare, but survival rates in the oldest age classes tend to approximately 65-85% of adults in their prime (eg. Loison *et al*.1999; Beauplet *et al*. 2006). In our impact assessments, we assume that these costs are borne immediately, and impose an additional 25% risk of mortality on all animals that are estimated to have PTS.

Behavioural avoidance.

We assume that the main impacts of noise are likely to result from behavioural avoidance of preferred foraging areas. The widespread distribution of harbour seals around the UK and other North Atlantic waters demonstrates that suitable foraging habit is widespread, and their broad diet highlights that these are 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 completely 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 lower 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 linear relationship between the proportion of the annual cycle in which disturbance occurs and the resulting reduction in reproductive success (Figure 9).

Table 5: Assumed worst case fitness consequences for individual seals that may be exposed to different levels pile driving noise (see Table 4 for threshold noise levels considered for different effects).

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 (Fig 8).	100% reproductive failure

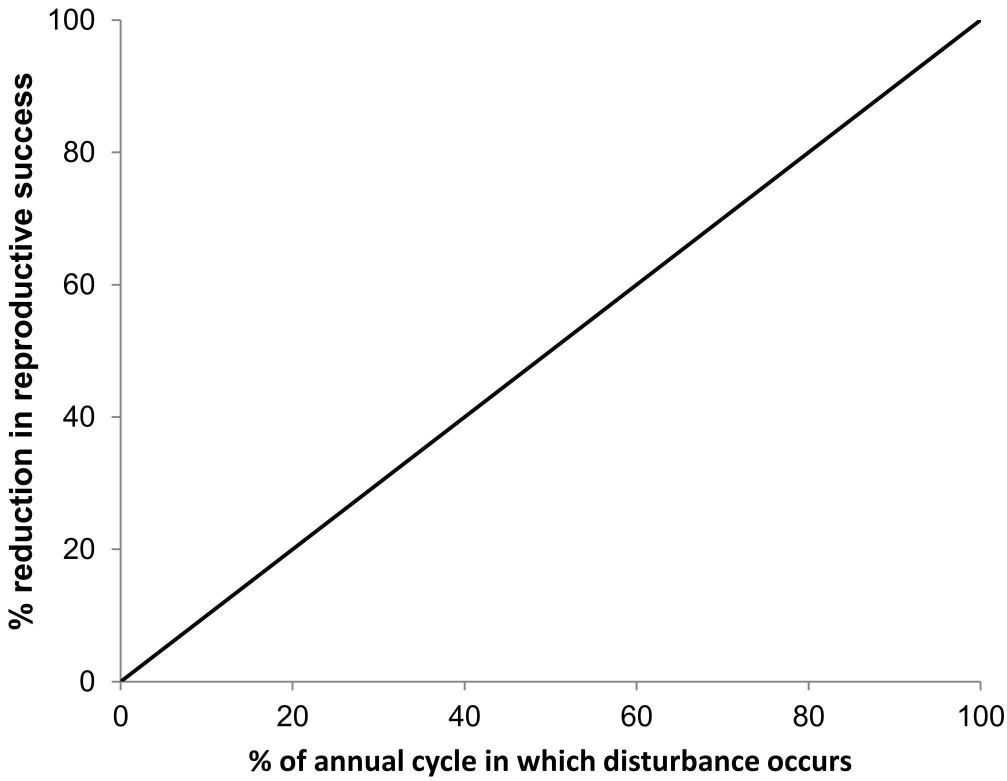


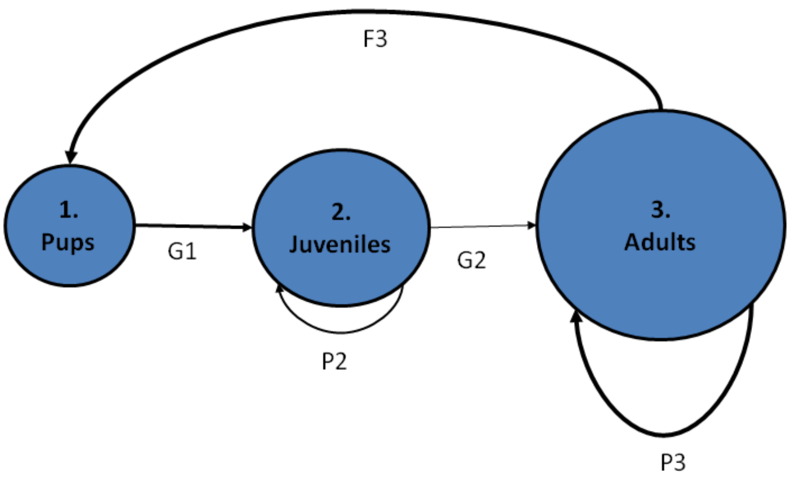
Figure 9: Hypothetical relationship between the amount of the year that individuals were displaced from foraging areas and the consequent reduction in their reproductive success.

3.4.2 Harbour seal population model

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). This approach is, for example, now used to support the Scottish Government's seal licensing system (http://scotland.gov.uk/Topics/marine/Licensing/SealLicensing/PBR).

However, whilst this approach can support the management of activities that directly cause mortality, it 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*., 2008). By taking this approach, we are 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.



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 10: A life-cycle graph for the stage-classified single sex harbour seal model. F1 to F3 represent the three female stage classes for males and females respectively.

We consider three life-history stages (Figure 10), 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*. (2008), supplemented by more recent analyses of photographic sightings of >150 individually recognisable harbour seals in Loch Fleet (Cordes 2011). Analyses of this 5 year data set have provided the first concurrent real-time estimates of survival and female reproductive success in a naturally regulated population of harbour seals. The availability of such data from the harbour seal breeding colony closest to the MORL and BOWL development sites makes this modelling approach particularly appropriate for this assessment. The input parameters used in the baseline model are presented in Table 6.

Table 6: 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 & Heide-Jørgensen 1990
Reproductive rate	88%	Cordes (2011)
Sex ratio	0.5	Boulva & McLaren 1979
Density dependent variation in reproduction	Yes	Using equation 3 in Taylor & DeMaster (1993) to vary reproductive rate between maximum literature value at low population size (0.95 (Boulva & 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 & Heide-Jørgensen 1990
Adult mortality	11% ♂; 3% ♀	Cordes (2011)

3.4.3 Predicted population consequences of displacement & PTS.

Impacts of wind farm construction were modelled by adjusting reproductive rates for the proportion of the population that were predicted to be affected by piling noise (Figure 8), as outlined in Table 7. In the illustrative scenario used here, we compare three construction scenarios, each starting in year 4 (Figure 11). In these scenarios, we assumed that construction was based on the same 1.8m piles used in the Beatrice Demonstrator. Our scenarios compared a three-year construction programme with year-round piling, with two five-year programmes, one piling year-round and the other for just 6 months in the summer. In these examples we use a single location for all piling events, but this could be developed to take account of spatial variation in piling locations, or multiple piling events

We assume that any risk of direct mortality can be avoided by mitigation, and that behavioural displacement occurs during the piling period (100% of the year in two scenarios and 50% in the third). In future scenarios this parameter can be varied depending upon temporal patterns of piling and likely recovery times between piling events. Similarly, there is potential to vary other assumed parameters to explore their sensitivity. The received dB $_{ht}$ levels shown in Figure 4 and the response curve shown in Figure 6 were then used to assess the proportion of foraging seals using each of the 4x4 km cells (see Figure 2) which would displaced. Similarly, the received SEL levels and PTS-onset curve in Figure 7 were used to assess the proportion of seals in each 4x4 km cell that might be exposed to levels that caused PTS. In this example (Fig 8), the sum of these proportions 0.03, indicating that less than one individual should be exposed to SEL likely to cause PTS. In comparison, a total of 3.4 individuals were predicted to be present within the 198dB PTS-onset threshold.

To model the effects of behavioural displacement, we reduced the reproductive success of displaced females (see Table 5) by removing an appropriate number of stage 1 (0-1 year old) seals in each of the construction years. Whilst we could have simply reduced the fecundity of those females, this approach also captures the possibility that females may reproduce, but produce poorer quality pups that are then less likely to survive their first year.

To model the effects of PTS (Table 5), we calculated the number of individuals that may suffer from PTS, and removed 25% of these individuals from the population in each year. The model is constructed so that this parameter can be easily varied to explore its sensitivity.

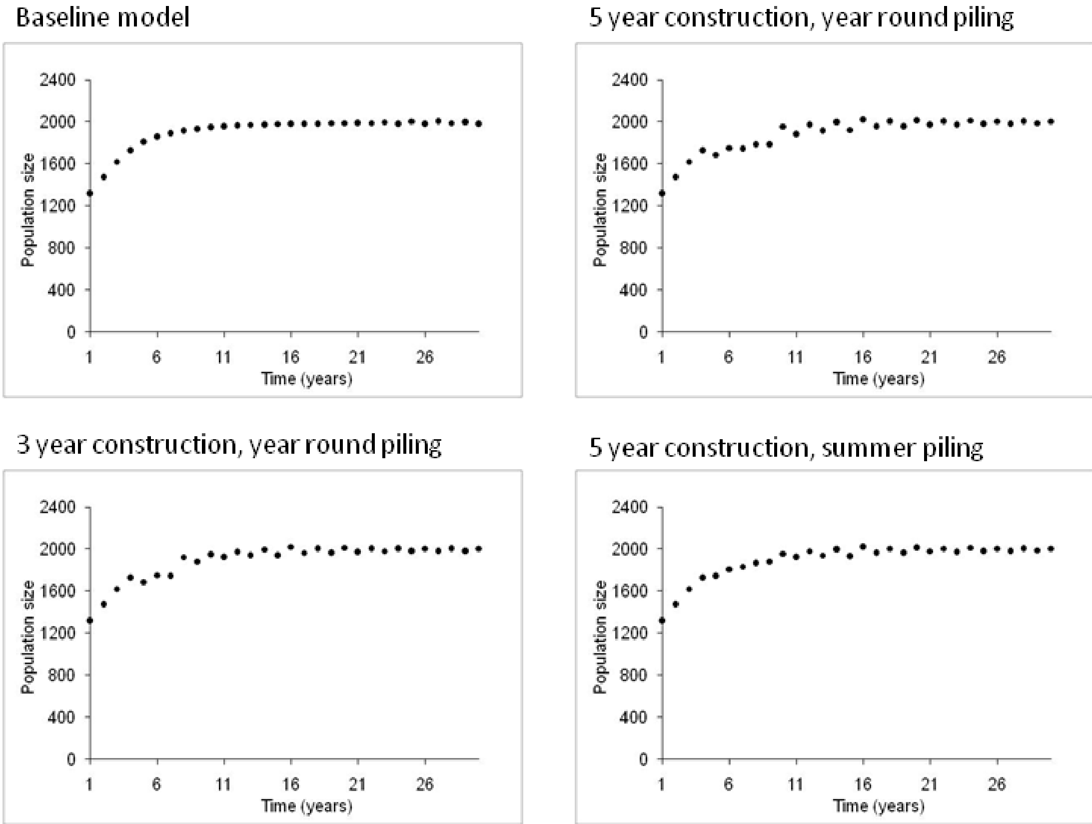


Figure 11: Illustration of how projections of change in total population size can be compared for different construction scenarios. Here the three scenarios vary in duration and intensity within years, and can be compared with the baseline model in which no construction occurs. Seals continue to be shot in each year at a level reflecting current use of PBR models, updated by the previous year's abundance. It should be noted that these different construction patterns are provided purely to illustrate how the model can be used, and do not represent construction programmes that are comparable in terms of technical or economic feasibility.

3.5 Discussion

This document provides a first attempt to develop a modelling framework to explore the potential impacts of pile-driving noise on the Moray Firth harbour seal population. As we discuss below, there is enormous variation in the quality of data available to parameterise the different components of our framework. Indeed, several parameters are based only upon specialist's educated best guesses. Faced with such uncertainty, some stakeholders may believe that one should not attempt to assess population level impacts, calling instead for additional data to be collected before decisions can be made. However, within the consenting timelines for the majority of the current Scottish Territorial schemes and for the early Round 3 projects, we feel that this is not possible. Consenting decisions will need to be made utilising the information available to achieve a balance between international agreements on climate change and nature conservation.

In the sections below we discuss key issues relating to data uncertainty, and ways in which the framework could be developed over the medium term to improve assessments of any impacts on the Moray Firth harbour seal population. We then explore the potential for extending this approach both to other harbour seal populations, and to other species of marine mammals.

3.5.1 Data availability & uncertainty

Common to many aspects of the Environmental Impact Assessment process, there are serious limitations in the amount of data available to assess the impacts of noise on marine mammal populations. Furthermore, even when data are available, these are sometimes based upon small samples, with some key studies being based on single captive individuals (see Annex I). Consequently, the level of scientific uncertainty underpinning each element of our assessment framework for the Moray Firth harbour seal population varies. As requested by JNCC, we use the Intergovernmental Panel on Climate Change (IPCC) guidance upon the classification of uncertainty (IPCC 2005) to provide an indication of the relative confidence in different components of our framework.

In Table 7, we reproduce the IPCC's recommended scale for characterising confidence in a dataset or assumption, based upon expert judgement. This scale is then used in Table 8 to summarise the confidence that we place in the different data available to us for use in the Moray Firth harbour seal assessment framework. These issues are discussed in more detail below, together with potential opportunities for reducing uncertainty in each area of the framework in the medium term, perhaps during monitoring programmes during wind farm construction activities.

Table 7: Quantitatively calibrated levels of confidence taken from the IPCC guidance upon classification of uncertainty. Taken from IPCC (2005)

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

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. Nevertheless, sample sizes are still relatively 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.

Noise distribution.

Despite slight differences in the approaches used by different noise propagation models, INSPIRE's predictions of received noise levels is one of elements of the framework in which we have has the highest level of confidence. Comparison of modelled and measured data from the Beatrice Demonstrator support this, but it would be beneficial to extend this comparison of peak-to-peak levels to compare predicted and measured dB_{ht} values. A DECC-funded comparison using recordings of seismic surveys (sampling up to 500kHz) is currently underway, and this data gap should be addressed during 2012.

Assessing impacts on individuals.

In contrast, there is much less certainty about the extent to which these received noise levels may impact individual seals in these areas. The preliminary nature of the noise exposure criteria developed by Southall *et al.* (2007) highlights the evolving nature of understanding in this area. Recent work, for example, now indicates that SEL measurements overestimate levels of TTS (Finneran *et al.* 2010). Planned research in the US 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 and representative data. Studies of individual variability in the hearing thresholds in wild harbour seals could 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). This technique has excellent potential for use on wild animals, for example when individual seals are being caught and instrumented with tracking devices. The University of St Andrews are currently planning collaborative studies of this kind on UK harbour seals. If successful, routine AEP tests during captures of wild seals could provide an important baseline to underpin future studies of changes in hearing ability over time.

Given the lack of data on how marine mammals behave in relation 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 also 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. It is anticipated that additional data will become available to test the generality of this dose-response relationship, and to assess recovery times, through DECC funded studies of harbour porpoise reactions to a seismic survey in the Moray Firth in 2011. In addition, it is hoped that harbour seal specific studies may be conducted in 2012 around wind farm construction sites and/or met mast installations.

In this case study we applied a dose-response curve for PTS (Fig 7), as used within SAFESIMM. Using this relationship illustrated that only a small proportion of animals within Southall *et al*'s (2007) PTS-onset criteria received noise doses that might lead to PTS. Where such effects only occur at short range, our application of peak noise levels in each 4km x 4 km grid square to an average density of animals in that whole cell tends to inflate the number of affected individuals. In future, using SAFESIMM to estimate PTS within our framework may provide more robust estimates. In the meantime, we propose to assess the number of animals exposed to PTS by using INSPIRE to identify which grid squares are exposed to SEL equal to or exceeding the PTS-onset level of 198 dB. We then assume that 10% of the animals within this area will develop PTS; a conservative estimate based upon comparisons in the Demonstrator case study.

Linking individual impacts to demographic parameters.

Even with better data on levels of displacement and PTS, there remains huge uncertainty over their subsequent consequences for fitness. It is these parameters within our framework that depend entirely upon expert judgement rather than even sparse data. Here, we use values which we suggest represent a sensible worst case scenario, but the modelling framework has been constructed so that these can be altered to explore the sensitivity of our overall results 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 assess variation in the levels of TTS, understanding of how variation in received noise affects TTS and the assumptions that link these to PTS.

Table 8: 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 & tracking data	High	Integration of data from three different tracking studies conducted within the Moray Firth between 1989 and 2009 (Bailey & Thompson 2011)
1.2 Habitat Association Model	Medium	Integrated dataset from 37 individuals to model habitat preference using a generalised additive model (GAM) (Bailey & Thompson 2011)
1.3 Spatial Model of distribution	Medium	Use of GAM results and population estimates to predict probability of current seal density in each 4x4km grid square across the Moray Firth.
2. Noise Distribution		
2.1 Piling source levels & 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 such as 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} (Nedwell *et al*., 2007) not validated for seals. In absence of data, highly precautionary approach used.
3.2 Estimate # individuals:			
	Non Auditory Injured	Low	No thresholds provided
	Auditory injured	Low	Based on Southall *et al*.(2007)
	Displaced/excluded	Low	As for behavioural avoidance.
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.

Instead, it is likely to be more productive to directly assess relationships between noise exposure and key demographic parameters using the PCAD framework developed in NRC (2003).

However, given the high level of uncertainty over the link between PTS and demographic parameters, our required assessments of population level impacts may benefit little from a better demographic parameters using the PCAD framework developed in NRC (2003). Whilst we might prefer a good understanding of the underlying mechanisms, the regulators main concern (in Europe at least) is whether anthropogenic noise impacts these protected populations, not whether it does so as a result of exclusion from feeding areas or a reduction in feeding success due to hearing damage. The Moray Firth harbour seal population offers excellent opportunities to develop PCAD studies such as this. Individually identifiable seals at the haul-out sites closest to proposed wind farms have been studied since 2005, 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 at the BOWL and MORL sites.

Harbour seal population model.

The final element of our framework involves a simple deterministic population model for the regional population of harbour seals. Initial analyses of the distribution of seals noise were conducted within ARCGIS, but the resulting grid based data can 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 allows 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. One advantage of the determinist framework is its quick operation, which allows us to quickly explore different scenarios and model sensitivity, potentially in a workshop situation with different stakeholder input. In parallel to this work, a more complex state-spaced model of Moray Firth harbour seal dynamics has been developed by Jason Matthiopoulos at the University of St Andrews. Like the PCAD models discussed above, future work would benefit from using these Bayesian approaches to incorporate uncertainty into model predictions, and use available data to estimate key unknown parameters.

Both these models of the Moray Firth harbour seal population draw heavily on individual based studies from Loch Fleet. Between 1995 and 2005, abundance at this site increased whilst abundance in the Dornoch Firth SAC decreased (Cordes *et al*. 2011). This has raised some concerns about whether these estimates of mortality and fecundity are representative of the wider Moray Firth population. However, all demographic data were collected between 2006 and 2010 (Cordes 2011), and inspection of the abundance data from these two areas (see Fig 12 of Bailey & Thompson 2011) suggests that abundance at both sites has increased slightly over this period. This, and the fact that demographic estimates are in line with those from overseas populations of harbour seals (see Cordes 2011), gives us confidence that these data are suitable for parameterising models for the regional population.

3.5.2 Applicability of the framework to other UK harbour seal populations.

The Moray Firth is one of, if not the, most intensively studied harbour seal populations in the world. Whilst this has been a great benefit in the development of this framework, this need not constrain the use of this approach in other UK regions. Whilst the temporal spread of telemetry data in the Moray Firth is unique, more extensive tracking has been conducted by SMRU over the last 10 years (e.g. Cunningham *et al*. 2009) and these data are currently being used in broader-scale habitat models to characterise foraging distribution around the UK. Similarly, whilst annual haul-out counts are made at only a few UK sites, a regular programme of moult surveys by SMRU provides broad-scale data on abundance and trends in different UK region. One concern is the extent to which less frequent surveys in other areas accurately reflect recent 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 using this framework 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 (eg. Caswell *et al*. 1998), when developing population models. Such uncertainty should therefore not constrain the development of similar modelling frameworks for other populations.

3.5.3 Applicability of the framework to other marine mammal populations.

A wide range of marine mammal species may occur in or around marine renewable development sites in UK waters (Reid *et al*. 2003), but the species most commonly encountered are likely to be grey seal, bottlenose dolphin, harbour porpoise and minke whale. Currently, SAC's have only been designated for grey seals and bottlenose dolphins. But disturbance of other cetaceans requires an EPS licence, which involves consideration of impacts on FCS. These assessments also require consideration of population impacts, which could take a similar approach to that used for harbour seals in this project. In many respects, the key areas of uncertainty relate to generic issues over the levels of noise at which animals may respond or suffer auditory injury. Because bottlenose dolphins have been a model study species for such work, data sources on hearing effects can sometimes be better for these species. However, in general, the issues over the level of uncertainty in this element of the framework is similar for all species of interest, especially when considering likely fitness consequences of displacement or PTS.

One major difference when applying the framework to cetaceans is that the underlying information on animal distribution is typically collected using large-scale visual surveys rather than through telemetry studies. As with telemetry studies, these data are generally used in habitat association models to predict distribution over broader areas. However, the source data are restricted to areas visited by survey platforms and they may not sample all areas used by the animals. This contrasts with telemetry studies which gain information on all areas visited by individual animals, but which may not sample the full distribution of the population given the relatively small number of individuals studied.

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. In contrast, bottlenose dolphins are highly mobile animals that range widely, often in large groups, visiting favoured foraging hot spots (Hastie *et al*. 2004), but sometimes travelling between areas as far apart as the Moray Firth and Firth of Forth in a few days (Wilson *et al*. 1999; Cheney *et al*. In Press). Thus, whilst aerial or boat-based survey data can be used to predict the average density of bottlenose dolphins across the Moray Firth, this provides a poor representation of the population's distribution at any moment in time. This is best illustrated by contrasting the SCANSII estimate of bottlenose dolphin density in this area (0.11 individuals per km^2) with the groups of 20-30 individuals that are typically seen along the Moray Firth coast in summer. Similarly, applying the relationship in Figure 6 to estimate the probability of displacing animals from a particular grid square is also more problematic for mobile bottlenose dolphins, as these animals would probably have moved through that area after a few hours in the absence of any noise impact.

Harbour porpoises probably fall between these two extremes. They are often seen as individuals or small groups, and occur at high density across the Moray Firth. Although information on the extent of individual movements is sparse, passive acoustic monitoring demonstrates that porpoises are present in these areas throughout the year. Given that our data on behavioural responses to noise were derived from studies of porpoises in similar habitats (Brandt *et al*. 2011) our approach to estimating the numbers of animals displaced is likely to be more suitable for this species. For both bottlenose dolphins and harbour porpoises, any consequences of displacement may be less critical given these species are not tied to local breeding or resting sites. Assessment of these consequences would then need to 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 & Read 2008; Winship 2009).

3.5.4 Conclusions

It is clearly unrealistic to expect any model, whether ecological or economic, to make accurate predictions about the future with a high level of certainty. Nevertheless, when used appropriately, models can play a crucial role in underpinning a wide range of management decisions.

Like any other piece of science, the development of this framework has required us to make a number of key assumptions. These are summarised in Annex I. The framework has been designed to provide an opportunity to explore the sensitivity of predictions to variations in these assumptions. In the meantime, a qualitative assessment of the relative importance of these different assumptions is also provided in Annex I. These evaluations can help direct decisions about future monitoring, provide feedback on whether these assumptions were appropriate, and identify future research requirements. The modular nature of this framework provides opportunities for new information to be readily incorporated as this becomes available.

We highlight several parallel initiatives which have the potential to provide more robust population assessments in future, particularly if focused studies can be integrated into monitoring programmes at consented sites. However, the timeframes for such work mean that they cannot be used to support assessments for Scottish Territorial Waters and Round 3 wind farm sites. This framework provides an interim tool that be used to explore the relative impacts of different construction scenarios on the long term dynamics of a protected harbour seal population. The hypothetical case study used here illustrates how the temporary impacts of pile-driving noise during construction can be assessed in relation to baseline population trends, and can incorporate cumulative impacts such as changes in levels of seal mortality from shooting.

4. Addendum to Seal Assessment Framework (March 2012)

The general methods used to assess the extent of behavioural displacement and numbers of individuals subjected to PTS were as described in the Seal Assessment Framework document. However, the following aspects of the framework were adapted as follows in response to comments from Marine Scotland the SNCAs.

4.1 Behavioural displacement

We now use upper, lower and best fit estimates for the relationship between the probability of displacement and received dBht levels in each grid cell. The upper level is based on the precautionary fit used previously. The best fit uses the predicted coefficients from logistic regression and the lower fit uses the lower standard error of those coefficients. These revised relationships are illustrated in Figure 12 and Figure 13 below.

Using these different curves, we provide upper, best and lower estimates of the number of individuals displaced for all the species assessed.

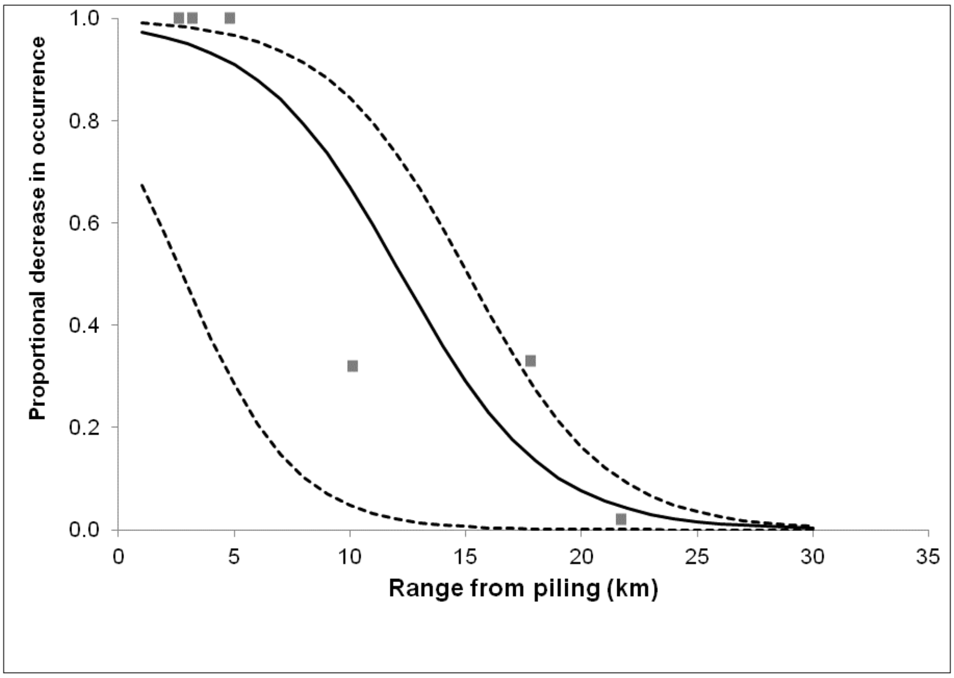


Figure 12: Predicted relationship between range from the Horns Rev piling and the proportional decrease in harbour porpoise occurrence (mean porpoise positive minutes from CPODs (from Brandt et al 2011)) before and in the hour after the event; the solid line in the figure shows the line of best fit. The upper dashed line is the precautionary fit used in the draft assessment, and the lower dashed line uses the regression co-efficients – 1 SE).

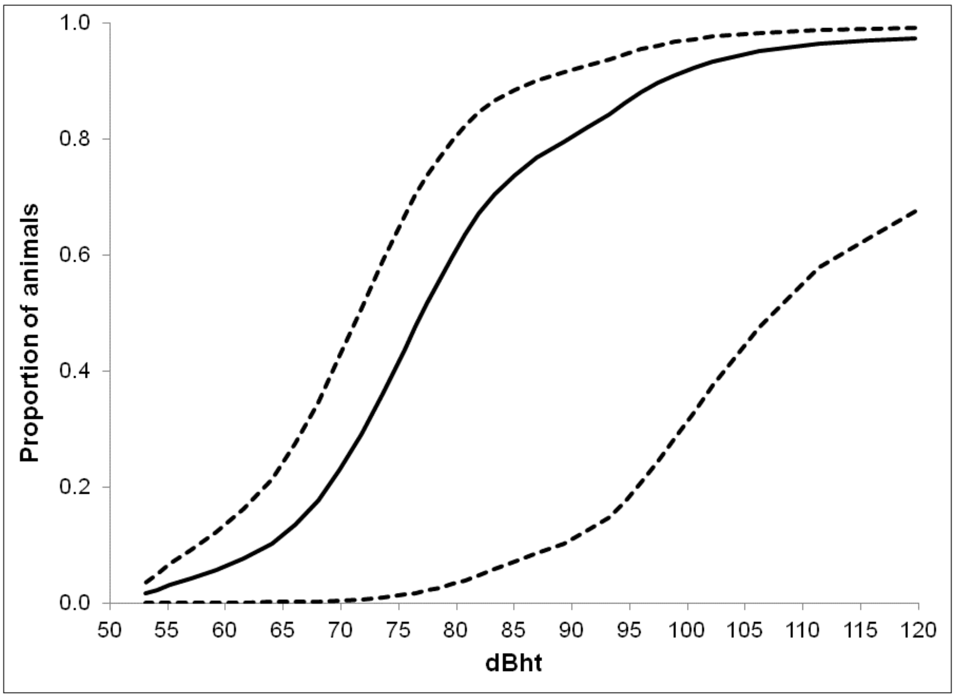


Figure 13: The relationship between dB$_{ht}$ for harbour porpoise and the predicted proportion of animals excluded from the area using the upper, best and lower fitted relationships from Figure 12.

4.2 PTS

For harbour seals, we now use a fleeing PTS model for the population projections, and present trajectories for both 186 dB and 198 dB PTS onset thresholds. We use the predicted at-sea distribution based upon telemetry data, but assume that 25% of animals are hauled out when assessing SEL and PTS. Numbers of animals affected are based on a total population of 1183 individuals. We initially explored the impact of using the different 186 and 198 dB PTS onset thresholds using an example of a seven year construction phase of a hypothetical wind farm in the Moray Firth (Figure 14 below). Comparison of these outputs highlighted little difference between the two PTS onset values in terms of population trajectories and further scenarios were modelled using 186 dB as the most conservative value. We also compared these results with those from SAFESIMM simulations of the same hypothetical wind farm construction programme using 186 dB PTS onset threshold (Figure 15). Given the use of SAFESIM resulted in the prediction of higher number of individuals with PTS, we use the outputs of SAFESIMM at 186 dB for the predictions of PTS within the ES.

Following requests from the regulator's advisers, we assessed the effect of varying the change in survival probability resulting from PTS from 0-30% for the same hypothetical wind farm's seven year construction phase (Figure 16). Clearly the use of different mortality rates lead to slight differences in predicted population size during the construction phase, but the long term population trajectory was similar in each case. In the EIA process, we continue to use 25% mortality as a conservative estimator.

In addition, we assessed the effect of changing assumptions about carrying capacity of the Moray Firth for harbour seals (Figure 17). This shows that the difference between baseline and impact population sizes during construction is strongly affected by assumptions made with regards to carrying capacity. We continue to use an estimated carrying capacity of 2000 within our modelling (approximately twice the current population size and slightly higher than previously recorded maximum).

For grey seals, we use at-sea density data as provided by SMRU. The percentage of animals affected relate dot the relative number of animals at sea within the Moray Firth.

For harbour porpoises, we use the approach described in the seal assessment framework to estimate the number of animals displaced. Estimates of the numbers subjected to PTS are based upon the SAFESIMM models that use 198 dB PTS onset thresholds. Data on densities of porpoises are based upon information provided in the Baseline Technical Appendix (4.4 A).

For bottlenose dolphins and minke whales, we use the approach described in the seal assessment framework to estimate the number of animals displaced. Estimates of the numbers subjected to PTS are based upon the predicted distribution in Figure 6.2 of Baseline Technical Appendix (4.4A), and assume that 50% of the East coast population of 195 animals are present within this area. Data on spatial variation in the density of bottlenose dolphins across the Moray Firth are based upon the predicted distribution in Figure Baseline Technical Appendix (4.4 A), and assume that 50% of the East coast population of 195 animals are present within this area. Values for the percentage of the population affected relate to the total east coast population. Densities of minke whales are based upon the SCANS density estimate of 0.022/km2 across the whole Moray Firth

4.3 Harbour seal population model

We assess the consequences of these individual impacts by modelling long term population trends for each scenario as described in the seal assessment framework. In each case, these trends are compared with a baseline trend based upon best estimates of Loch Fleet demographic rates

The current PBR take is included in both the baseline model and all pile-driving scenarios

The impacts of pile driving are assessed by assuming a) that 25% of the animals with PTS die over the year following exposure and b) that all females that are displaced do not breed, or produce pups that die (implemented by increasing pup mortality in the following year).

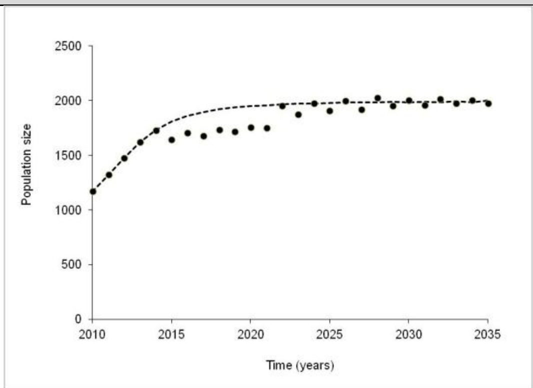
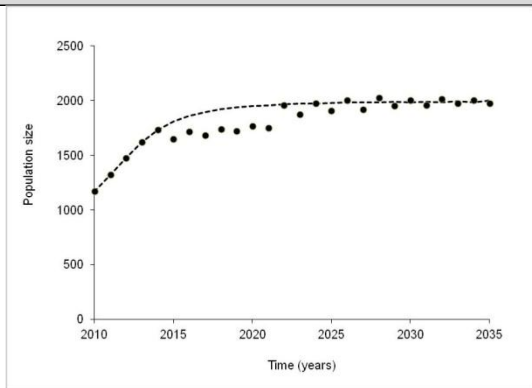
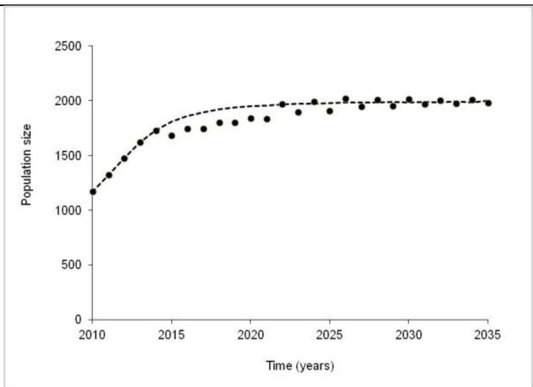
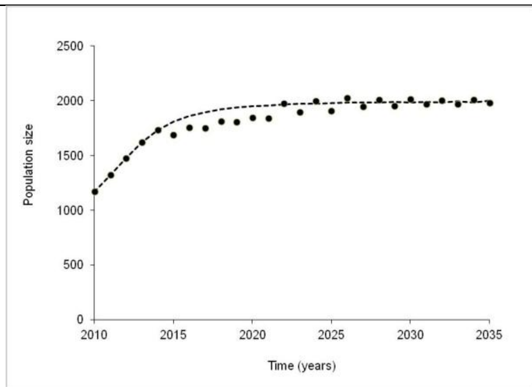
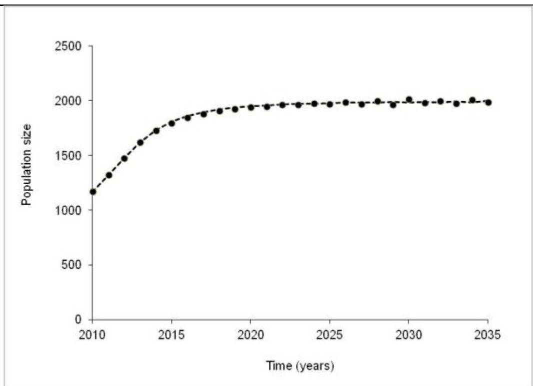
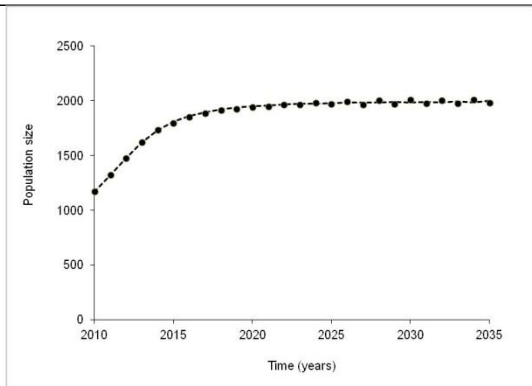
	186dB PTS Onset	198 dB PTS Onset
Upper		
Best		
Lower		

Figure 14. Comparison of using 186 dB (left) and 198 dB (right) PTS onset levels when modelling long term trends in the Moray Firth harbour seal population. Each sub-figure compares the baseline trend (dotted line) with the impact scenario (solid circles). Upper, mid and lower panels use different displacement dose-response curves.

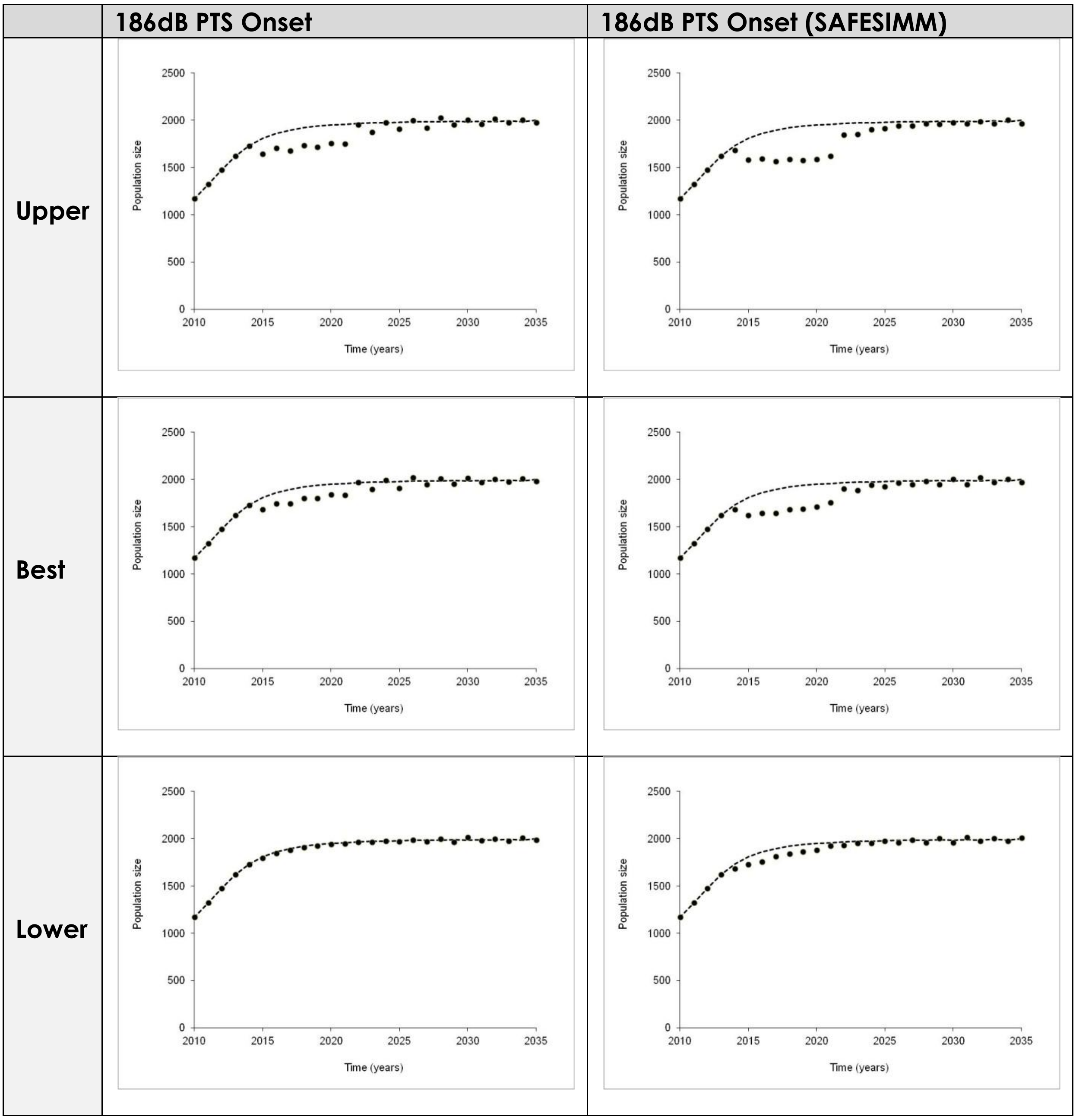
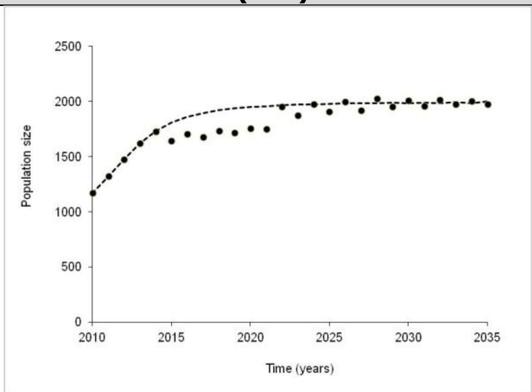
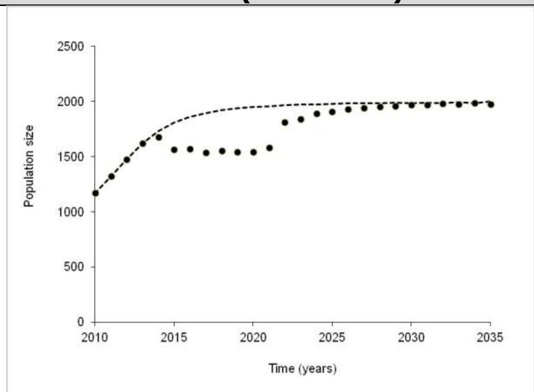


Figure 15. Comparison of using the seal assessment framework (left) and SAFESIMM (right) to estimate the number of animals with PTS when modelling long term trends in the Moray Firth harbour seal population. Each sub-figure compares the baseline trend (dotted line) with the impact scenario (solid circles). Upper, mid and lower panels use different displacement dose-response curves. Both approaches use 186 dB PTS onset levels to estimate the number of animals with PTS

<table>
<tr><th></th><th>186dB PTS Onset (SAF)</th><th>186dB PTS Onset (SAFESIMM)</th></tr>
<tr><td>30%</td><td>
</td><td>
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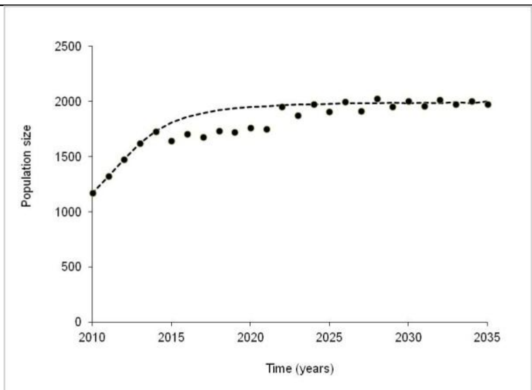
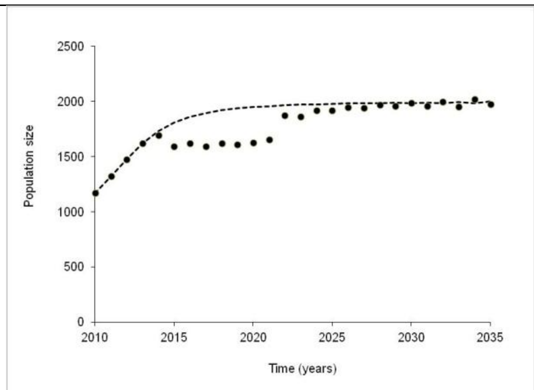
<tr><td>20%</td><td>
</td><td>
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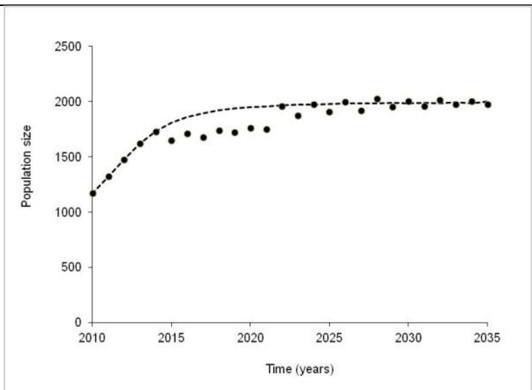
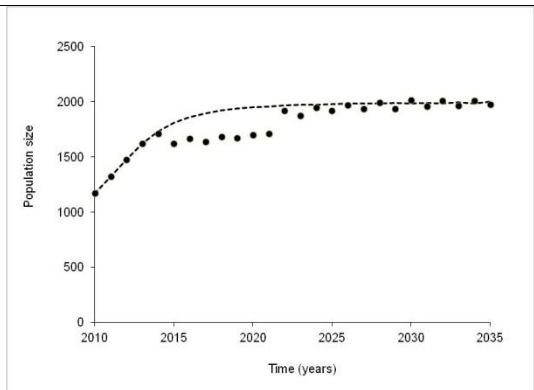
<tr><td>10%</td><td>
</td><td>
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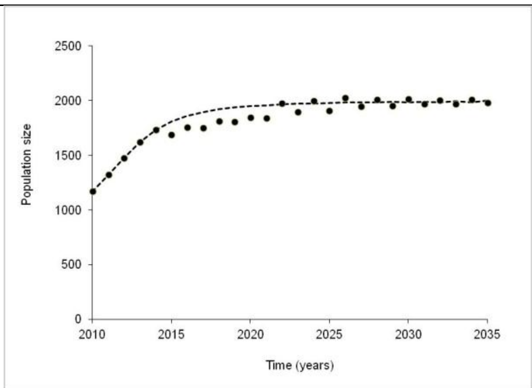
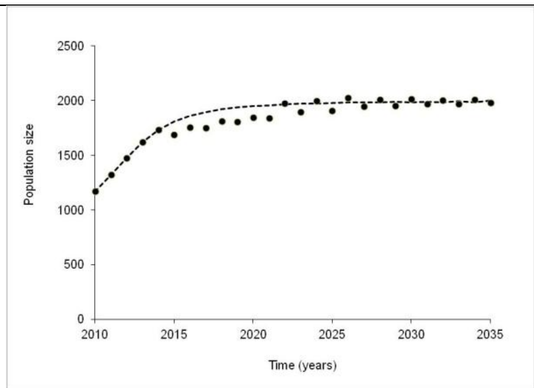
<tr><td>0%</td><td>
</td><td>
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Figure 16. Comparison of the impact of different additional mortality rates using both the seal assessment framework and SAFESIMM estimates of the number of animals with PTS.

<table>
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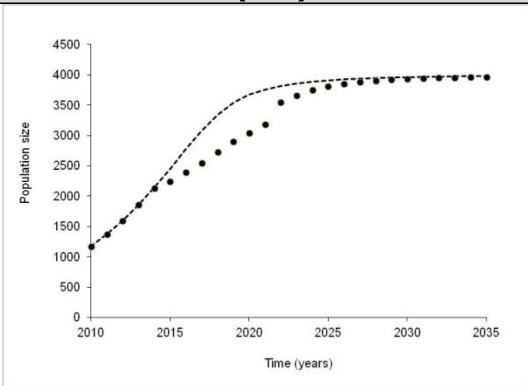

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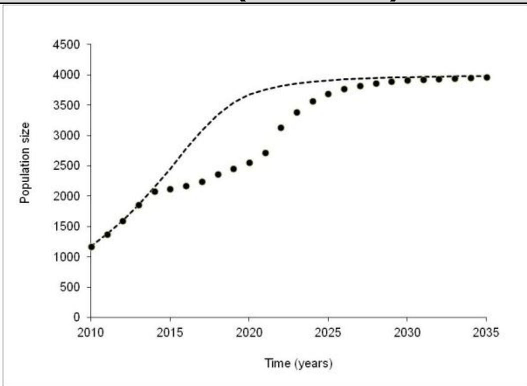

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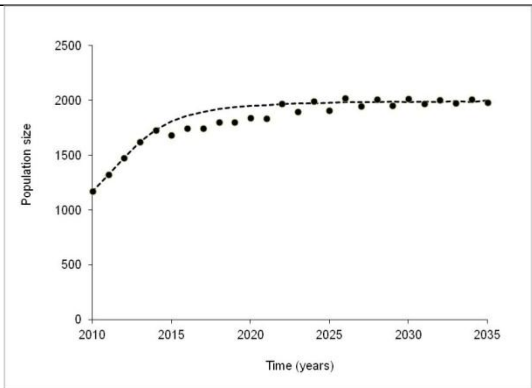

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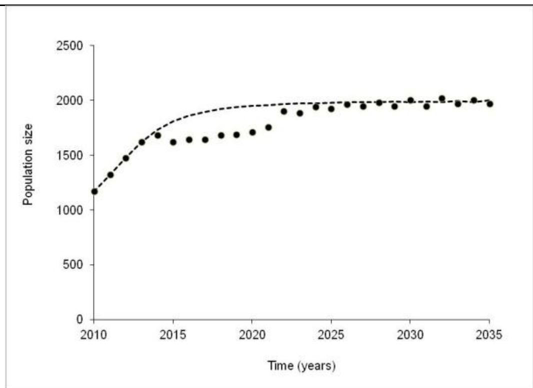

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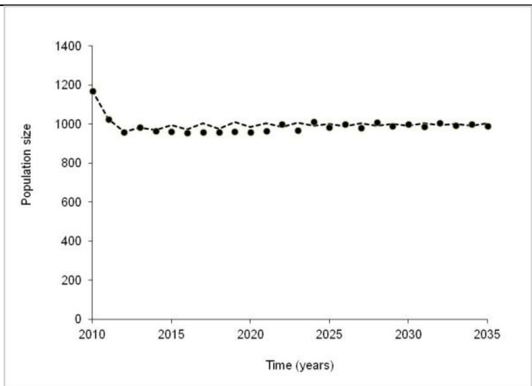

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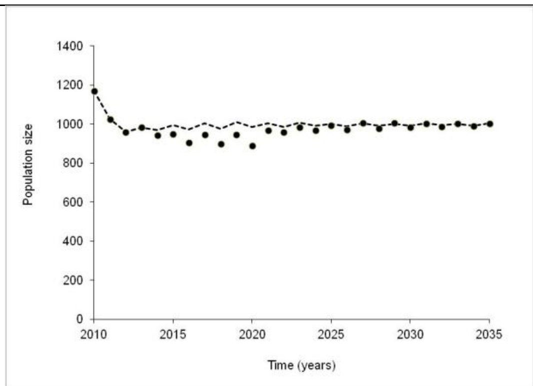

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Figure 17. Comparison of using different values for carrying capacity when modelling long term trends in the Moray Firth harbour seal population. Models were run using estimates of PTS from both the seal assessment framework (left) and SAFESIMM (right) to estimate the number of animals with PTS.

4.4 Bottlenose dolphin population model

We assess the potential impacts of displacement on bottlenose dolphins using a published individual-based model that has previously been used to assess different monitoring and management scenarios for the Moray Firth population (Thompson et al. 2000). This uses available literature values for bottlenose dolphin demographic and life-history parameters in the programme VORTEX to produce a baseline model with a stable population growth rate. Although this uses data from other populations, we suggest that the model's stable growth rate is appropriately conservative given that the most recent assessment of the East coast bottlenose dolphin is of a stable or increasing population (Thompson et al. 2011). This baseline model was compared with different impact scenarios in which the effects of displacement were modelled as a direct impact on reproduction. As for harbour seals, this conservatively assumed that all dolphins that were displaced failed to produce calves in that year. This was implemented in VORTEX by harvesting the appropriate number of 0-1 yr old calves from the population.

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6. Annex I.

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 7. 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	Low
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 198 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
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 form 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 & DeMaster 1993.	Medium	Medium
The carrying capacity is fixed at 2000, 20% above the maximum abundance estimate since 1990	Medium	Medium

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