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

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Background

Many European offshore windfarm 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 windfarm 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 outwith 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.

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 windfarm 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.

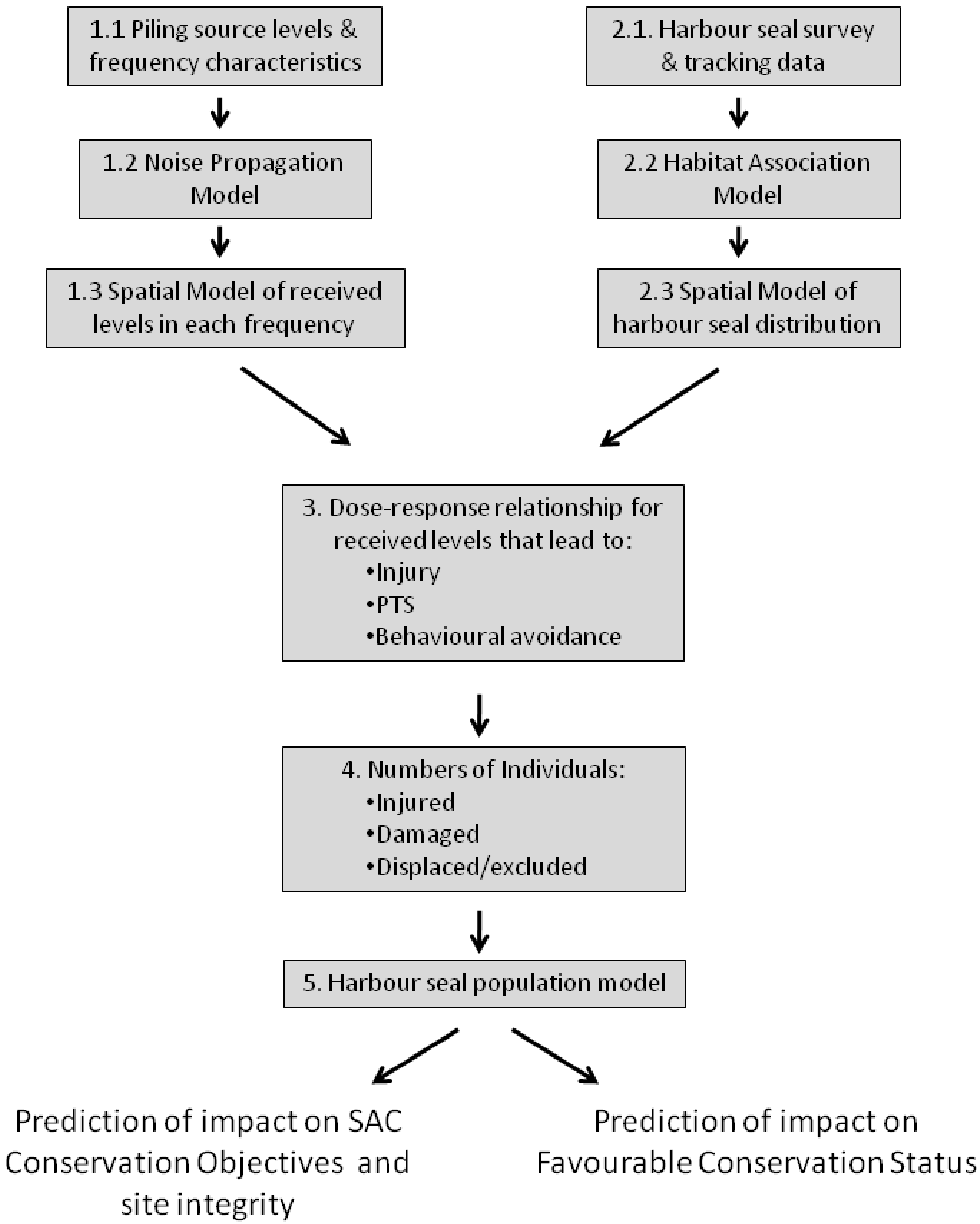


Figure 1. Schematic of the approach proposed for assessing the impact of windfarm construction on the harbour seal SAC and FCS.

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 windfarms 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 windfarm using analogous construction techniques.

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

1. Seal Distribution

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

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).

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.

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 (eg. see Härkönen et al. 1999 and Thompson et al 1998).

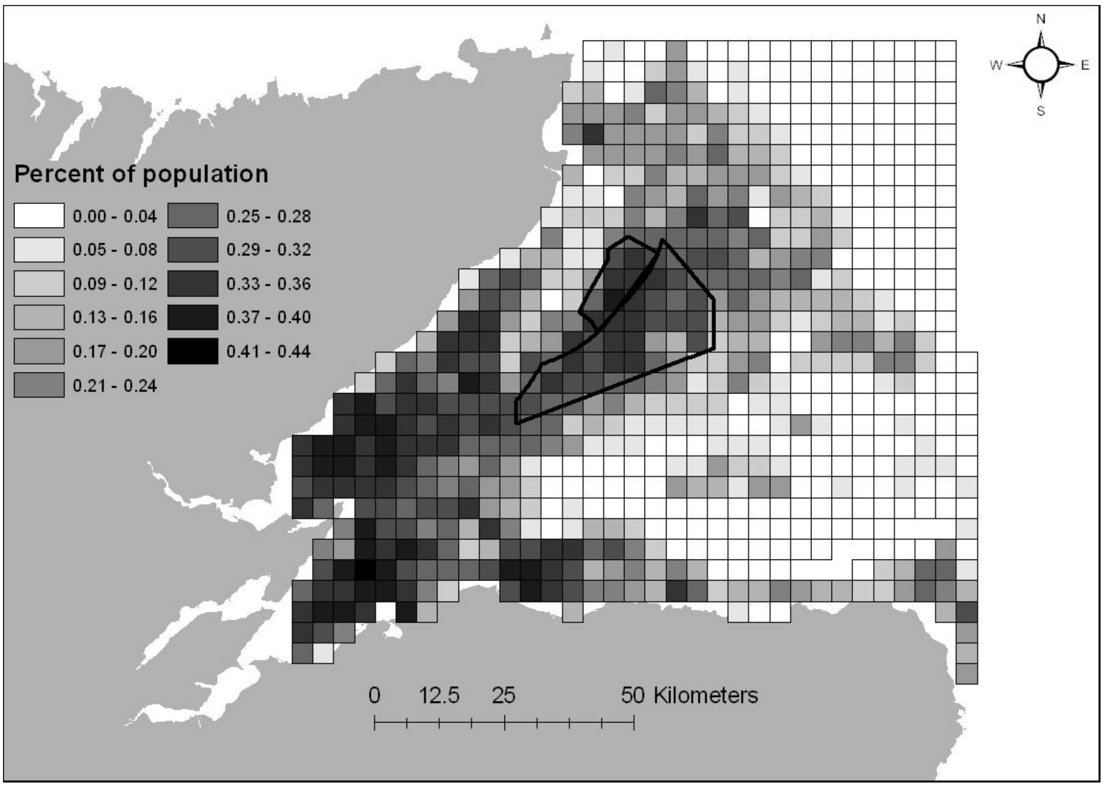


Figure 2. At-sea distribution of harbour seals in the Moray Firth. Data are based on the habitat association analysis in Bailey & Thompson (2011), and show the percentage of the population that is expected to be found in each of the different 4x4 km grid cells.

2. Noise Distribution

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.

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 windfarm 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).

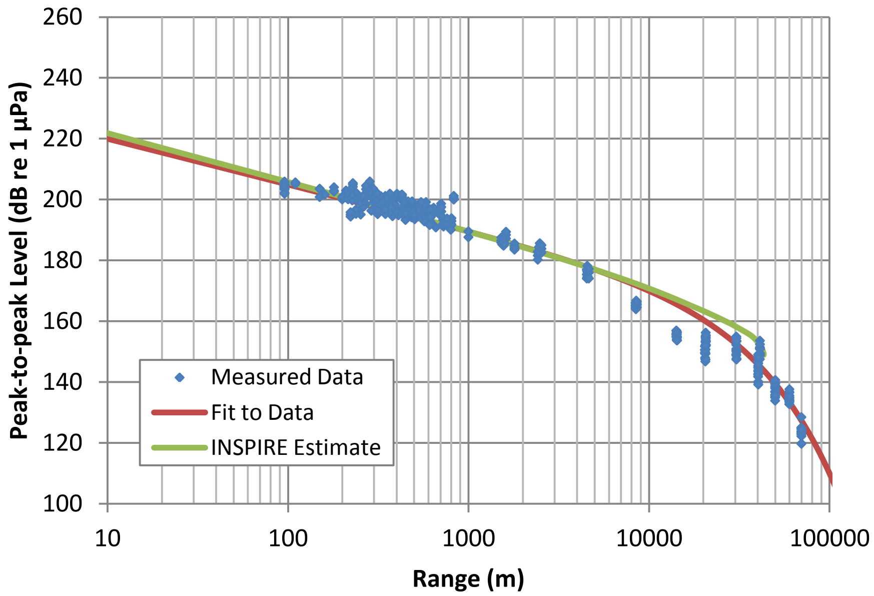


Figure 3. The level of sound in unweighted peak to peak levels as a function of range in meters for a 1.8 m diameter pile at the Beatrice demonstrator site. Predictions from the INSPIRE model are presented alongside measured data from Bailey et al. (2010)

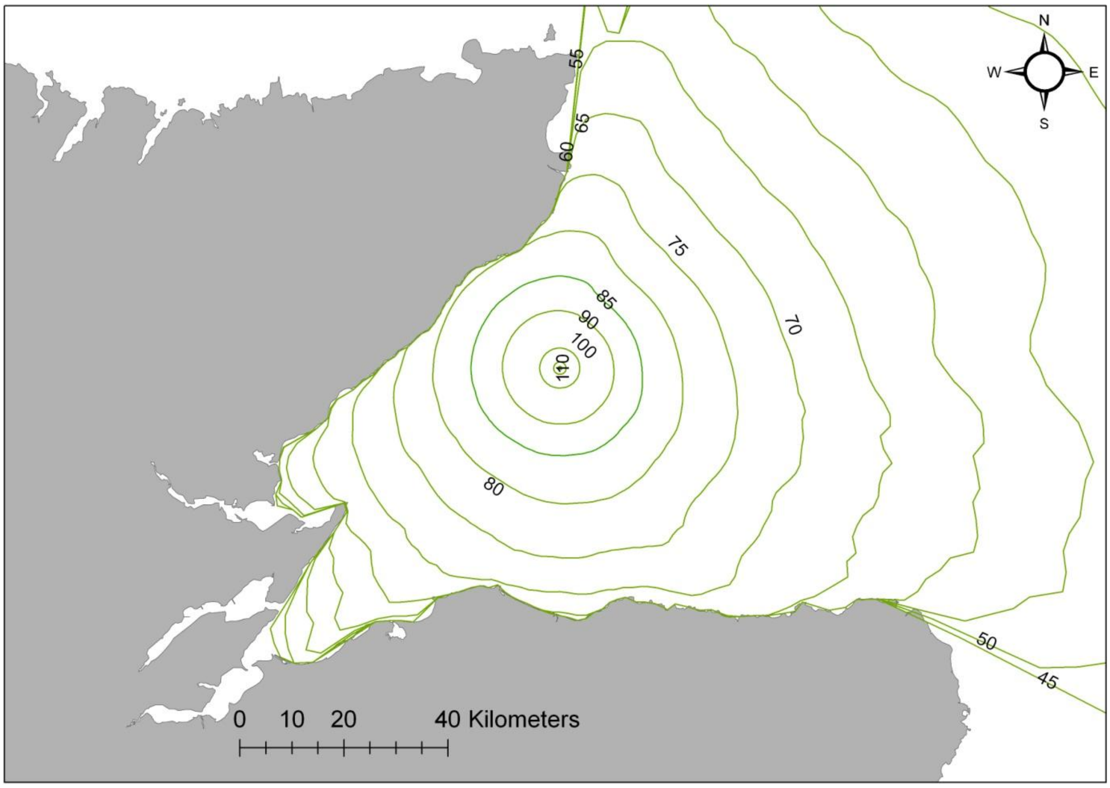
2.3 Spatial Model of received levels

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 windfarm 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.



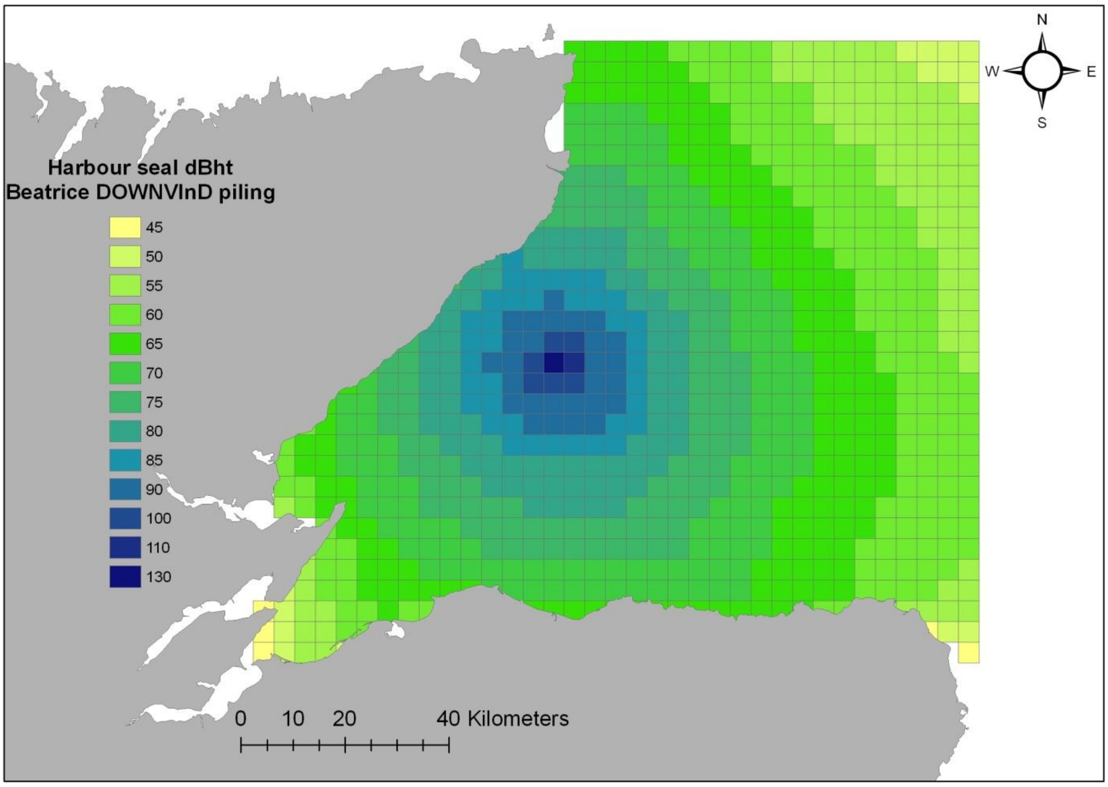


Figure 4. top) contour map showing the output generated from the Subacoustech INSPIRE model, bottom) map showing variation in received levels across the 4x4 km grid cells.

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. 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 windfarm 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

3.1 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 • Physical Injury	Typically associated with rapid compression of air containing structures
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 • Changes in foraging or social behaviour	See Annex II for Southall's et al's more detailed breakdown of behavioural effects

3.1.1 Behavioural avoidance. There is no consensus on the extent to which low level behavioural responses should be considered in environmental assessments. Given the ultimate need to explore the consequences of these effects in the context of long-term population change, we argue that there is only utility in considering a behavioural effect if this can be translated into an impact on demographic rates. For example, whilst an individual may show a startle response or change its swimming speed in relation to a distant pile driving event, it is currently impossible to predict if, or how much, this might affect its reproductive success or survival.

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.

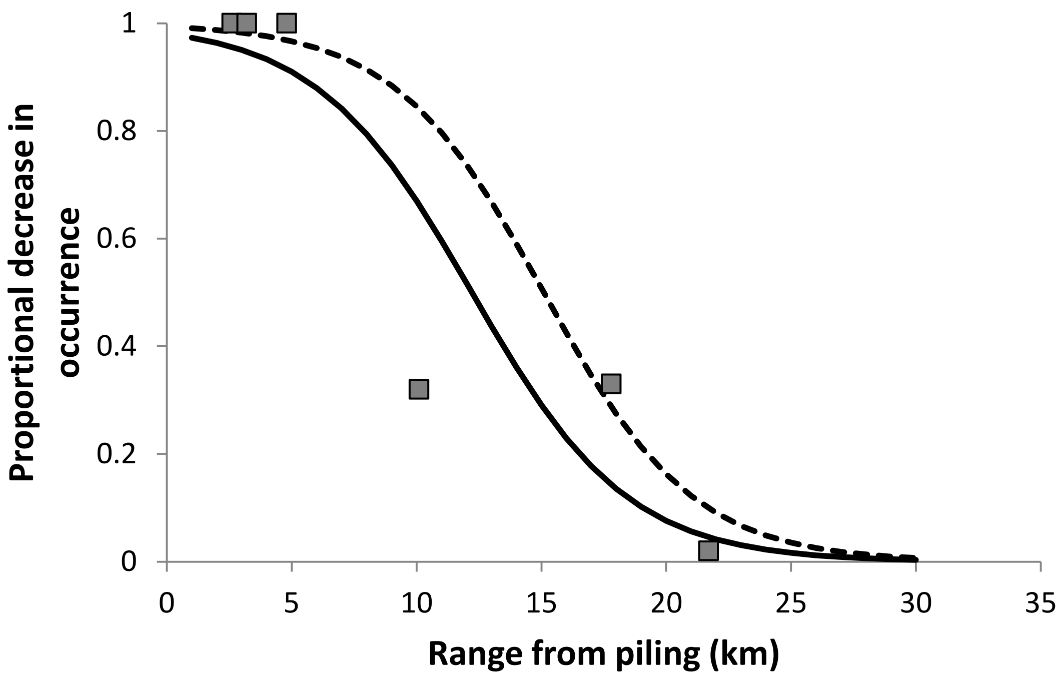


Figure 5: Predicted relationship between range from the Horns Rev piling and the proportional decrease in harbour porpoise occurrence (mean porpoise positive minutes from CPODs (from Brandt et al 2011)) before and in the hour after the event; the figure shows the line of best fit (deviance = 4.19, df=1,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

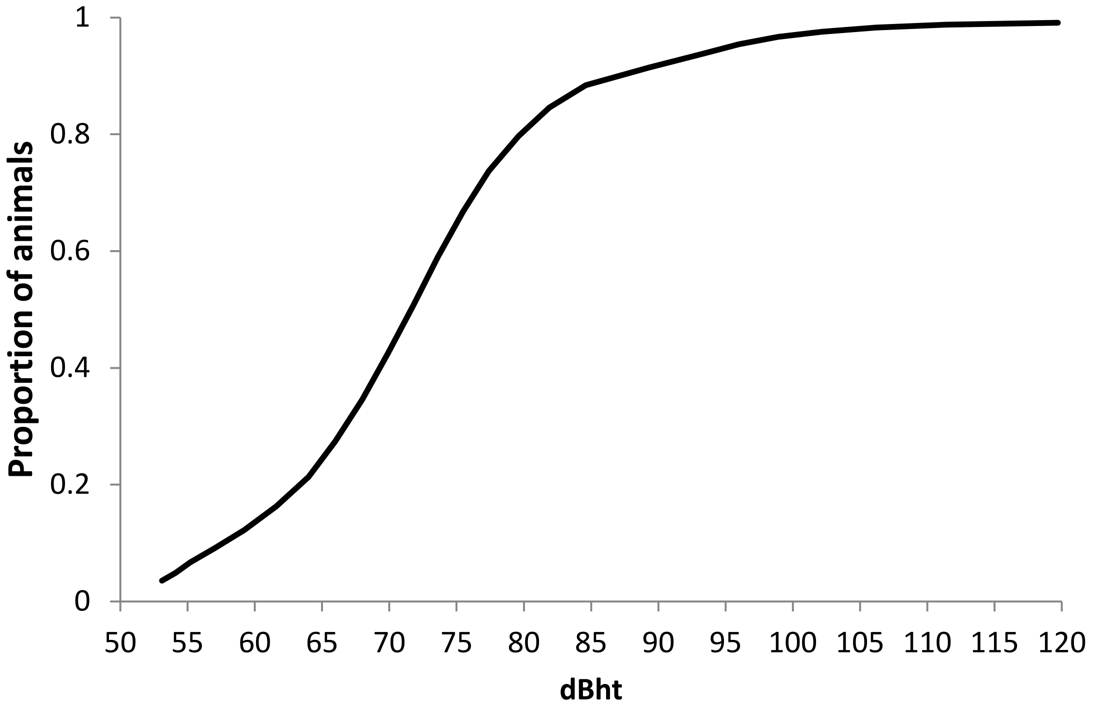


Figure 6. The relationship between dB_{ht} for harbour porpoise and the predicted proportion of animals excluded from the area (using the precautionary relationship from Figure 5).

3.1.2 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 Annex I), 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.

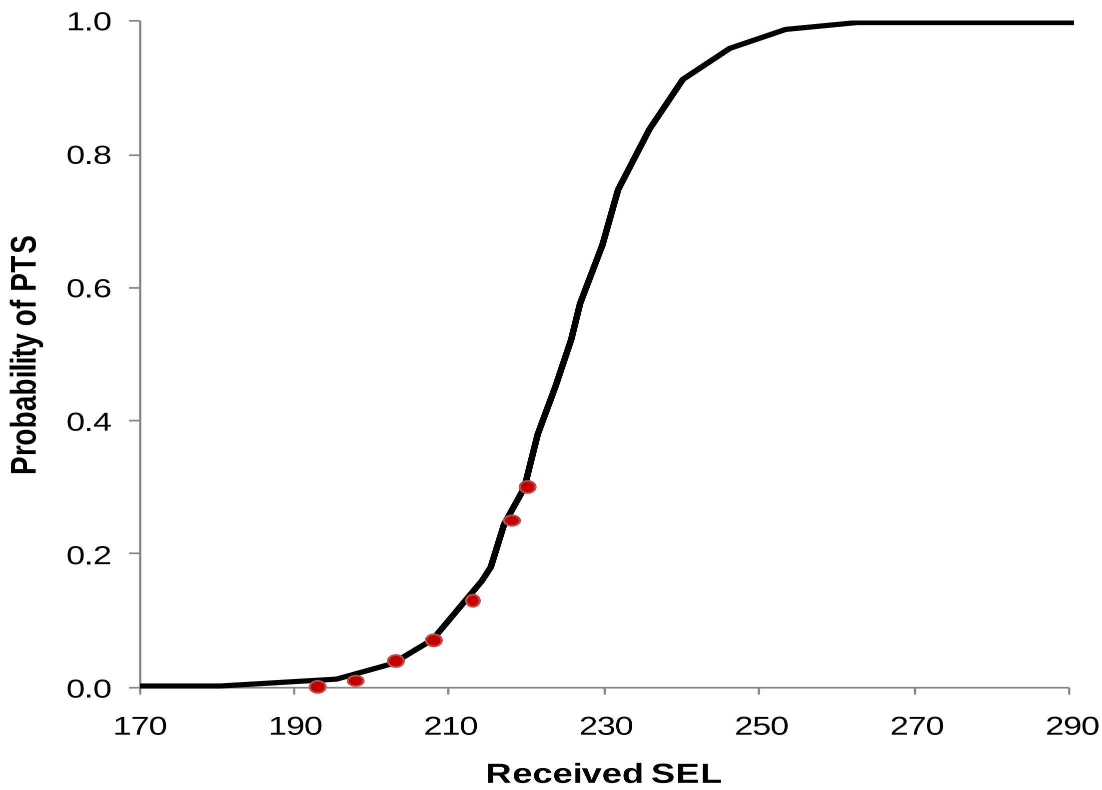


Figure 7 – Generalised PTS dose-response curve for a PTS-onset value of 198 dB, as used within SAFESIMM. Points in red correspond to the SEL contours generated using INSPIRE (see Table 2).

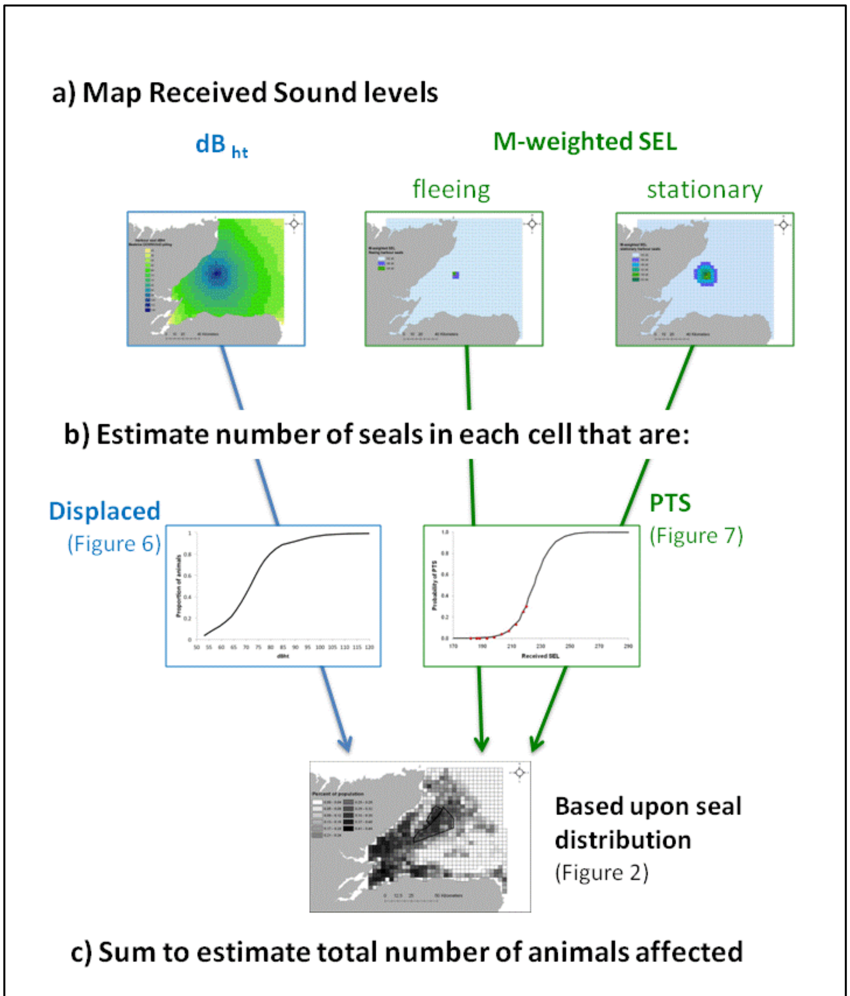
3.2 *Estimate the number of individuals injured, damaged or displaced/excluded*

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.



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

4. Assessment of impacts upon the population

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 periods 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.

4.1.1 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.

4.1.2 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, and 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.

4.1.3 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 (Fig 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

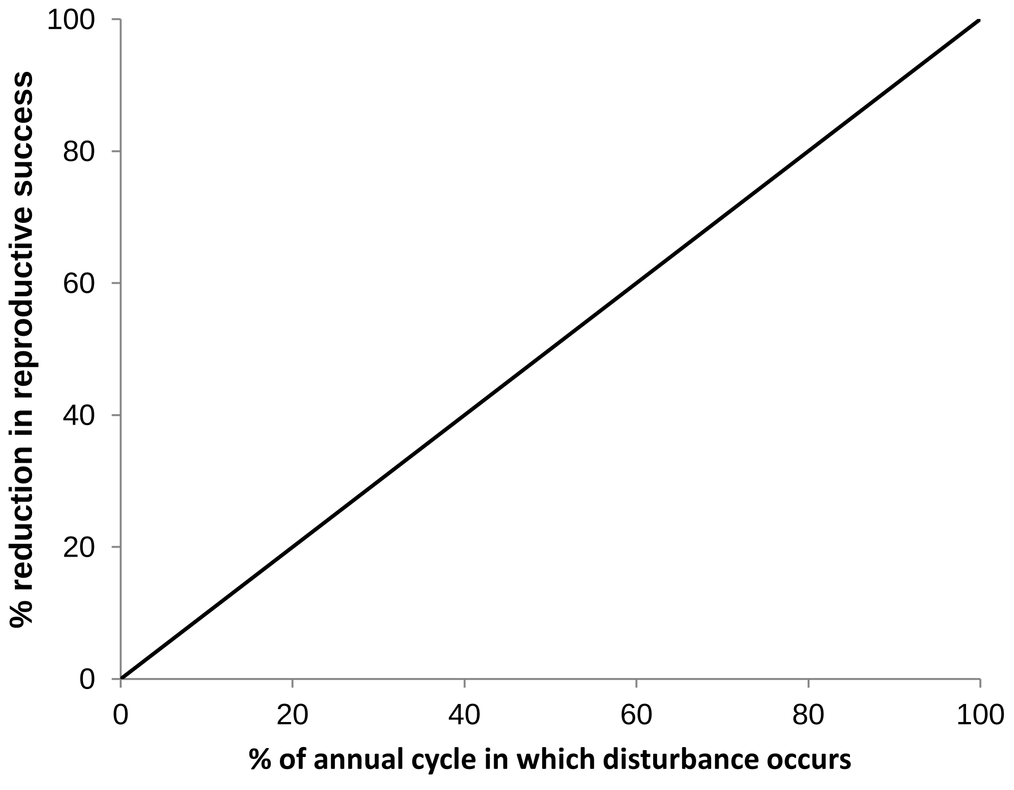
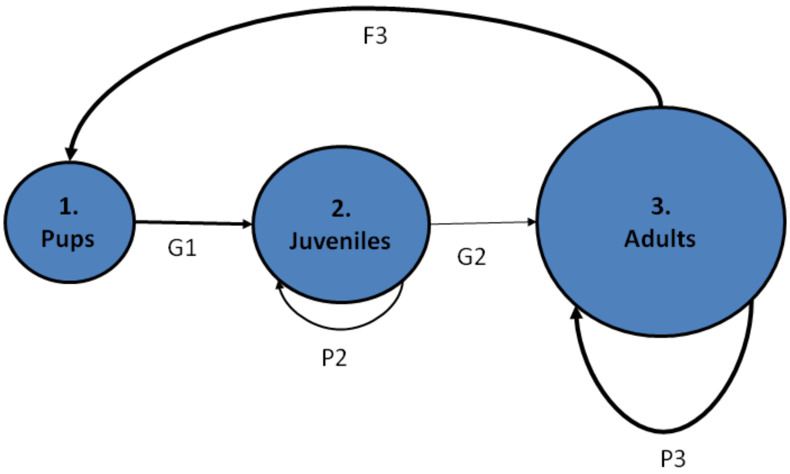


Figure 9. Hypothetical relationship between the amount of the year that individuals were displaced from foraging areas and the consequent reduction in their reproductive success.

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 windfarms, 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)

4.3 Predicted population consequences of displacement & PTS.

Impacts of windfarm 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.

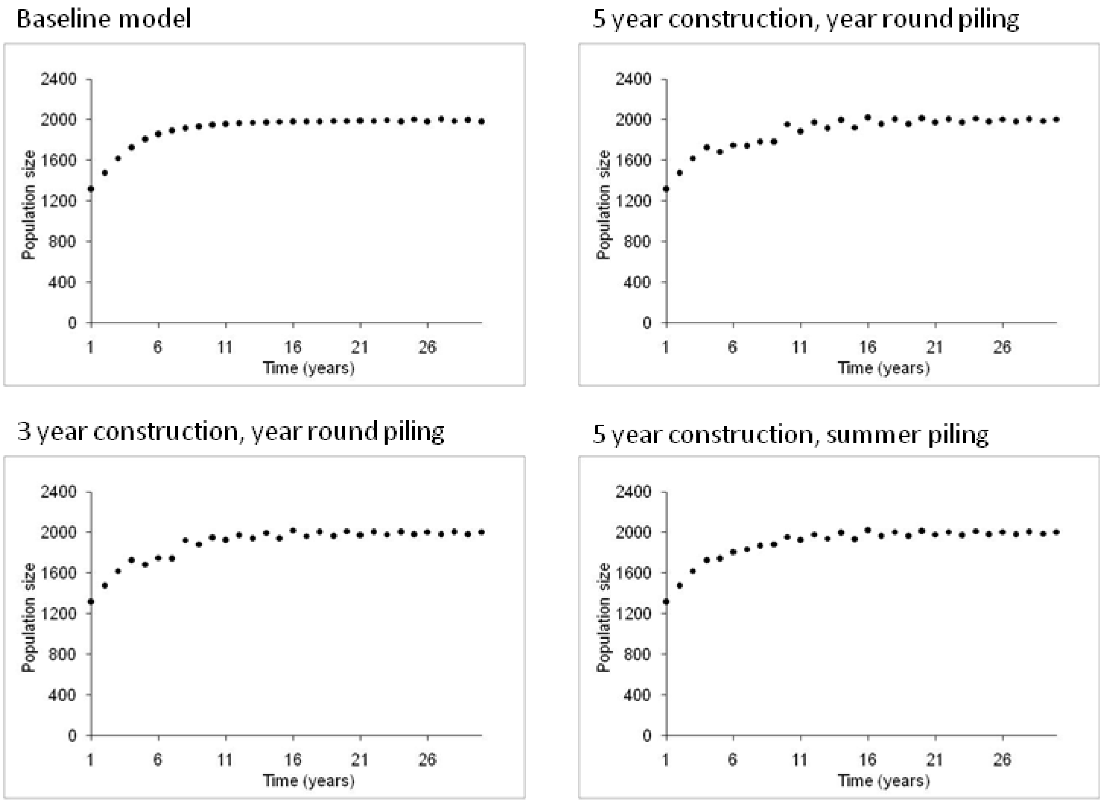


Figure 11. Illustration of how projections of change in total population size can be compared for different construction scenarios. Here the three scenarios vary in duration and intensity within years, and can be compared with the baseline model in which no construction occurs. Seals continue to be shot 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.

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.

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 (eg 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 windfarm 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 understand of how variation in received noise affects PTS. 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).

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.

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 windfarms 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 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 this deterministic framework is its quick operation, which allows us to explore different scenarios and model sensitivity, potentially in workshop situations 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.

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 clearly 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.

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).

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 II. 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 II. 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 windfarm 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.

5. References

Bailey, H., Parvin, S., Senior, B., Simmons, D., Rusin, J., Picken, G. & Thompson, P.M. (2010) Assessing underwater noise levels during pile-driving at an offshore windfarm and its potential effects on marine mammals. Marine Pollution Bulletin 60: 888-897.

Bailey, H. & Thompson P.M. (2011). Technical report on harbour seal telemetry and habitat model. Report to MORL.

Boulva, J., & McLaren, I.A. (1979). Biology of the harbour seal, Phoca vitulina, in eastern Canada. Bulletin of the Fisheries Research Board of Canada 200, 1-24.

Brandt MJ, Diederichs A, Betke K, Nehls G (2011) Responses of harbour porpoises to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. Mar Ecol Prog Ser 421: 205-216

Butler, J.R.A., Middlemas, S.J., McKelvey, S.A., McMyn, I., Leyshon, B., Walker, I., Thompson, P.M., Boyd, I.L., Duck, C., Armstrong, J.D., Graham, I.M., Baxter, J.M., (2008). 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, 18; 1025-1038.

Caswell, H., Brault, S., Read, A.J. & and Smith, T.D. (1998) Harbor porpoise and fisheries: an uncertainty analysis of incidental mortality. Ecological Applications, 8; 1226–1238

Cheney, B., Thompson, P.M., Ingram, S.N., Hammond, P.S., Stevick, P.T., Durban, J.W., Culloch, R.M., Elwen, S.H., Mandleberg, L., Janik, V.M., Quick, N.J., Islas-Villanueva, V., Robinson, K.P., Costa, M., Eisfeld, S.M., Walters, A., Phillips, C., Weir, C.R., Evans, P.G.H., Anderwald, P., Reid, R.J., Reid, J.B. & Wilson, B. (In Press) Integrating multiple data sources to assess the distribution and abundance of bottlenose dolphins (Tursiops truncatus) in Scottish waters. Mammal Review.

Cordes, L.S., Duck, C.D., Mackey, B.L., Hall, A.J., & Thompson, P.M. (2011) Long-term patterns in harbour seal site-use and the consequences for managing protected areas. Animal Conservation 14, 430-438.

Cordes, L.S. (2011). Demography and breeding phenology of a marine top predator. PhD Thesis, University of Aberdeen.

Cunningham, L , Baxter, JM , Boyd, IL , Duck, CD , Lonergan, M , Moss, SE & McConnell, B. (2009) Harbour seal movements and haul-out patterns: implications for monitoring and management , Aquatic Conservation: Marine and Freshwater Ecosystems , 19, 398-407.

Dehnhardt, G., Mauck, B., Hanke, W. & Bleckmann (2001) Hydrodynamic trail-following in harbor seals. Science 293, 102-104.

Deecke, V.B., Slater, P.J.B., & Ford, J.K.B. (2002) Selective habituation shapes acoustic predator recognition in harbour seals. Nature 420, 171-173.

Finneran, J.J., Carder, D.A., Schlundt, C.E. & Ridgway, S.H. (2005) Temporary threshold shift in bottlenose dolphins (Tursiops truncatus) exposed to mid-frequency tones. Journal of the Acoustical Society of America, 118, 2696-2705.

Finneran, J.J., Carder, D.A., Schlundt, C.E. & Dear, R.L. (2010) Temporary threshold shift in a bottlenose dolphin (Tursiops truncatus) exposed to intermittent tones. J. Acoust. Soc. Am. 127: 3267-3272

Fowler, CW (1990) Density dependence in northern fur seals (Callorhinus ursinus). Marine Mammal Science 6: 171–195.

Gotz, T & Janik, V.M. (2010) Aversiveness of sound in phocid seals: psycho-physiological factors, learning processes and motivation. Journal of Experimental Biology 213, 1536-1548.

Harding, K.C., Fujiwara, M., Axberg, Y., & Härkönen, T. (2005). Mass-dependent energetic and survival in harbour seal pups. Functional Ecology 19, 129-135.

Hastie, G.D., Wilson, B., Wilson, L.J., Parsons, K.M. & Thompson, P.M. (2004) Functional mechanisms underlying cetacean distribution patterns: hotspots for bottlenose dolphins are linked to foraging. Marine Biology 144, 397-403.

Härkönen, T., & Heide-Jørgensen, M.-P. 1990. Comparative life histories of east Atlantic and other harbour seal populations. Ophelia 32, 211-235.

Härkönen, T., Hårding, K. C. and Lunneryd, S. G. (1999), Age- and sex-specific behaviour in harbour seals Phoca vitulina leads to biased estimates of vital population parameters. Journal of Applied Ecology, 36: 825–841.

Härkönen, T., Dietz, R., Reijnders, P., Teilmann, J., Harding, K., Hall, A., Brasseur, S., Siebert, U., Goodman, S., Jepson, P., Dau Rasmussen, T. & Thompson, P.M. (2006) A review of the 1988 and 2002 Phocine Distemper Virus seal epidemics in European Harbour Seals. Diseases in Aquatic Organisms, 68; 115-130

Hayes, S.A., Kumar, A., Costa, D.P., Mellinger, D.K., Harvey, J.T., Southall, B.L. & Le Boeuf, B.J. (2004) Evaluating the function of the male harbour seal, Phoca vitulina, roar through playback experiments. Animal Behaviour, 67; 1133-1139.

Loison, A, Festa-Bianchet, M., Gaillard, J-M, Jorgenson, J.T.; Jullien, J-M (1999) Age-Specific Survival in Five Populations of Ungulates: Evidence of Senescence. Ecology, 80, 2539-2554.

Lonergan, M., Duck, C.D., Thompson, D., Mackey, B.L., Cunningham, L. & Boyd, I.L. (2007). Using sparse survey data to investigate the declining abundance of British harbour seals. J. Zoology, 271, 261-269.

Lusseau, D. (2010) Report from the Intersessional LaWE steering group. http://iwcoffice.org/_documents/sci_com/SC62docs/SC-62-WW6.pdf

Maclean, I.M.D., Frederiksen, M & Rehfisch, M.M. (2007) Potential use of population viability analysis to assess the impact of offshore windfarms on bird populations. BTO Research Report No. 480 to COWRIE, BTO, Thetford.

Moore, J.E. & Read, A.J. (2008). A Bayesian uncertainty analysis of cetacean demography and by-catch mortality using age-at-death data. Ecological Applications 18, 1914–1931.

Nedwell J R, Turnpenny A W H, Lovell J, Parvin S J, Workman R, Spinks J A L, Howell D (2007). A validation of the dBht as a measure of the behavioural and auditory effects of underwater noise. Subacoustech Report Reference: 534R1231, Published by Department for Business, Enterprise and Regulatory Reform.

NRC (2003). Ocean Noise and Marine Mammals. National Academies Press, Washington.

NRC (2006).Noise and Military Service; implications for hearing loss and tinnitus. National Academies Press, Washington.

Southall, B.L., Bowles, A.E., Ellison, W.T., Finneran, J.J., Gentry, R.L., Greene, J.C.R., Kastak, D., Ketten, D.R., Miller, J.H., Nachtigall, P.E., Richardson, W.J., Thomas, J.A., & Tyack, P.L. (2007). Marine mammal noise exposure criteria: initial scientific recommendation. Aquatic Mammals, 33, 411-521.

Thompson, P.M., Tollit, D.J., Wood, D., Corpe, H.M., Hammond, P.S. & Mackay, A. (1997). Estimating harbour seal abundance and distribution in an estuarine habitat in N.E. Scotland. Journal of Applied Ecology, 34: 43-52.

Thompson, P.M., Mackay, A., Tollit, D.J, Enderby, S. & Hammond, P.S. (1998). The influence of body size and sex on the characteristics of harbour seal foraging trips. Canadian Journal of Zoology, 76: 1044-1053.

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: 1253-1263.

Thompson, P.M., Mackey, B.L., Barton, T.R., Duck, C. & Butler, J.R.A. (2007) Assessing the potential impact of salmon fisheries management on the conservation status of harbour seals in NE Scotland. Animal Conservation, 10; 48-56.

Van Parijs, S.M., Thompson, P.M., Tollit, D.J. & Mackay, A. (1997). Distribution and activity of male harbour seals during the mating season. Animal Behaviour, 54: 35-43.

Van Parijs, S.M., Hastie, G.D. & Thompson, P.M. (1999). Geographic variation in temporal and spatial vocalisation patterns of male harbour seals, Phoca vitulina, during the mating season. Animal Behaviour, 58: 1231-1239

Wade, P. (1998). Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. Marine Mammal Science 14; 1-37.

Wieskotten, S., Mauck, B., Miersch, L., Dehnhardt, G. & Hanke, W. (2011) Hydrodynamic discrimination of wakes caused by objects of different size or shape in harbour seal. J. Expt. Biol. 214, 1922-1930.

Winship, A. (2009) 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.

Williams, R., Hall, A. & Winship, A. (2008) Potential limits to anthropogenic mortality of small cetaceans in coastal waters of British Columbia. Canadian Journal of Fisheries and Aquatic Sciences, 65:1867-1878

Wilson, B., Hammond, P.S. & Thompson, P.M. (1999). Estimating size and assessing status of a coastal bottlenose dolphin population. Ecological Applications, 9: 288-300.

Annex I. Identification of appropriate noise exposure criteria for assessing pinniped auditory injury during the construction of offshore wind farm sites

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Summary

Many countries now require assessment of the potential impacts of noise upon marine mammals as part of the consenting process for particular activities or developments in the marine environment. However, these assessments are frequently constrained by a lack of data on the nature and extent of potential impacts of noise at both individual and population levels. Understanding of these key areas of uncertainty is crucial, first, to help those evaluating assessments to understand the limitations of predictions about potential impacts and, second, to identify priorities for research that will provide more robust assessments in future. Drawing on the deliberation of a series of inter-disciplinary expert review groups, Southall et al. (2007) made initial scientific recommendations for marine mammal noise exposure criteria. Although driven by the particular needs of the US Marine Mammal Protection Act, this review has provided an important framework for noise assessments, and its findings and recommendations are being used by researchers and regulators across the world. However, our application of Permanent Threshold Shift Onset (PTS-onset) criteria to assessments required under European law has highlighted unexpected inconsistencies in the predicted impact that high levels of pulsed noise resulting from windfarm construction may have on pinniped and cetacean populations. To better understand this issue, we review the basis of these recommendations in Southall et al (2007). Based upon this review, we argue that the evidence-base is insufficient to support Southall et al's (2007) suggestion that there should be different PTS-onset criteria for pinnipeds and cetaceans which are exposed to pulsed noise. Until more appropriate studies have been carried out, we propose that the M-weighted SEL of 198 dB for exposure to multiple pulsed sounds be used for both cetaceans and seals.

Application of noise exposure criteria to assessments of offshore windfarm construction

In the UK, statutory regulators have encouraged the use of Southall et al.’s (2007) criteria to assess environmental impacts of offshore windfarm developments, in particular in relation to the high levels of multiple-pulsed noise produced when piling turbine foundations. Previous consideration of multiple-pulsed sounds from seismic air guns has focussed on the assessment of traumatic injury in the immediate vicinity of these noise sources. Information in Southall et al. (2007), in combination with existing guidance developed to mitigate against such risks (http://jncc.defra.gov.uk/page-1534) can be applied to reduce near-field risks from pile-driving. However, given the extended periods required for large-scale windfarm construction, these assessments require more emphasis on potential far-field impacts through behavioural displacement and more subtle auditory injury that may lead to PTS.

Southall et al.’s (2007) exposure criteria consider different types of noise (single pulsed, multiple pulsed and non-pulsed sounds) and different types of biological impacts (ranging from traumatic injury and death to more subtle behavioural responses). In the absence of species-specific data, they consider relative “M-weighted” sound exposure levels (SEL) at which impacts may be expected for four broad functional groups, defined on the basis of the characteristics of their audiograms (high, medium and low-frequency cetaceans and pinnipeds). The pinniped criteria are further sub-divided for exposure to in-air and underwater noise.

Southall et al.’s (2007) work provides a valuable context for windfarm assessment, but there are practical limitations when using their published noise exposure criteria for these broader-scale purposes. This is recognised for behavioural impacts, as Southall et al. (2007) explicitly state that 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*”. However, the application of Southall et al’s (2007) criteria for PTS-onset results in extremely large predicted zones of impact for seals when compared to cetaceans. This issue is illustrated in Table 1, which presents the predicted areas within which harbour seal, harbour porpoise and bottlenose dolphin would have been at risk of PTS following a pile-driving event during the construction of the offshore wind demonstration site at the Beatrice oilfield, off NE Scotland. This modelled scenario was based on the installation of two 1.8m diameter pin piles over a 24 hour period, representing the first half of the installation of a quadruped turbine base. The piling parameters were recorded during the pile driving activity published by Bailey et al

(2010) and included a strike number of 6,223 per pile, a strike-duration of 600ms, and a broadband source level of 226 dB re 1 µPa at 1m. These data were then used as input parameters in a propagation model (Bailey et al (2010)), which was used to predict third octave band received levels at a series of ranges from the piling (Equation 1); this allowed predictions of the ranges that the Southall PTS thresholds are likely to be exceeded.

Equation 1

$$RL=SL-20 \times \log(R)-0.0004(R)$$

Where:

RL=Received sound pressure level

SL=Source sound pressure level

R=Range in metres

Table 1: Predicted ranges to PTS for each of the functional groups defined by Southall et al (2007). Ranges to PTS for each of the functional groups defined by Southall et al (2007). $M_{(p)}$ = M weighting for pinnipeds in water; $M_{(lf)}$ = M weighting for low frequency cetaceans such as minke whale; $M_{(lf)}$ = M weighting for mid-frequency cetacean such as bottlenose dolphin; $M_{(hf)}$ = M weighting for high frequency cetacean such as harbour porpoise.

Functional group	PTS Threshold	Range (km)	Area (km^2)
$M_{(p)}$	186 dB re 1 lPa2 -s	18.9	1128
$M_{(lf)}$	198 dB re 1 lPa2 -s	2.3	17
$M_{(mf)}$	198 dB re 1 lPa2 -s	2.0	13
$M_{(hf)}$	198 dB re 1 lPa2 -s	1.9	9

The values shown in Table 1 represent the ranges at which SEL are predicted to reach a threshold where there is a risk of PTS-onset for bottlenose dolphin and harbour porpoises (198dB) and for harbour seals (186dB) (see Table 3 Southall et al. (2007)). In this scenario, where we take a conservative approach and assume that animals do not respond behaviourally to the noise, the ranges at which seals were predicted to suffer from PTS was 8-10 times greater than the predicted ranges for cetaceans. Given the need to assess the relative impact of developments on protected populations of both seals and cetaceans, and the clear disparity between the predicted impact ranges, the basis of the difference between the pinniped and cetacean PTS-onset criteria warrants further investigation.

Empirical evidence for PTS-onset criteria

Developing criteria for auditory injury is especially challenging because it is unethical to conduct experiments that directly estimate the noise levels required to cause PTS. Instead, the approach taken has been to base precautionary exposure criteria for PTS-onset upon experimental data on the levels required to cause the onset of Temporary Threshold Shifts (TTS). TTS-onset was, in turn, defined by Southall et al. (2007) as the noise level required to cause a temporary elevation of hearing threshold by 6dB. Southall et al. (2007) recognise that the development of these PTS-onset criteria is constrained by three factors.

- First, the precise relationship between PTS and TTS is not fully understood, even for humans and small mammals that have been the subject of extensive studies.
- Second, that different procedures are required for estimating PTS-onset according to sound type (pulses and non-pulses).
- Third, that experimental data even for TTS-onset is extremely sparse, and is often based on just one or two captive individuals of a very restricted set of species.

Their resulting auditory injury criteria were based on the assumption that PTS-onset occurred under conditions that caused 40dB of TTS (Southall et al. 2007); a level above which the likelihood of PTS becomes increasingly likely in humans (Kryter 1994). Because studies of marine mammals all report lower levels of TTS than 40dB (typically <10dB), the level of exposure to pulsed noise that was predicted to cause 40dB of TTS was estimated from a published relationship between the level of TTS and levels of noise in chinchillas (Henderson & Hammernik 1986). Based upon precautionary analyses of these data, Southall et al. (2007) estimate that "*PTS-onset (40dB TTS) is likely to occur on exposure to an M-weighted SEL 15 dB above that associated with TTS-onset"*.

For cetaceans exposed to pulsed noise, the only published TTS-onset data available to Southall et al. (2007) were from bottlenose dolphins and belugas. Furthermore, their PTS-onset criteria for all cetaceans were based on the study of a single beluga (Finneran et al. 2002) because this represented the most precautionary values. For this individual, TTS-onset from a single pulse occurred at a peak pressure of 224 dB re 1 µPa (peak) and M_{mf} weighted SEL of 183 dB re: 1 µPa^2-s. By adding 15dB to the latter, the M-weighted SEL criteria used for PTS injury from a single pulse was 198 dB re: 1 µPa^2-s. The criteria for multiple pulses were numerically identical to those for a single pulse (Southall et al. 2007).

For pinnipeds in the water, published data on TTS-onset were available to Southall et al. (2007) from three species (harbour seal, California sea lion and northern elephant seal). However, most of these studies used only non-pulsed noise. The exception was Finneran et al's (2003) study of two California sea lions that were exposed to single underwater pulses of up to 183 dB re 1 µPa (peak-to-peak) (SEL: 163 dB re: 1 µPa^2-s). However, no measureable TTS was detected at these levels and there were consequently no experimental data which allowed Southall et al. (2007) to directly estimate TTS-onset for pinnipeds exposed to underwater pulsed noise. In the absence of such data, PTS-onset criteria for pulsed noise were developed by assuming that *"the known pinniped-cetacean difference in TTS-onset upon exposure to non-pulse sounds would also apply (in a relative sense) to pulses. Specifically, with nonpulse sounds, harbor seals experience TTS-onset at approximately 12dB lower received levels than do belugas (ie. 183 vs 195 dB 1 µPa^2-s; Kastak et al. 1999, 2005; Southall et al. 2001; Schusterman et al. 2003 vs Finneran et al. 2000, 2005; Schlundt et al. 2000; Nachtigall et al. 2003, 2004)* (Southall et al. 2007).

Evidence for a difference between cetacean and pinniped TTS-onset levels?

The assertion that there is a consistent difference between pinniped and cetacean TTS-onset levels underpins the proposed difference in criteria for noise exposure levels causing PTS-onset in pinnipeds and cetaceans. There are three factors that lead us to question the basis of this assertion.

First, given the extremely small number of individual subjects used in these studies we question the conclusion that there is a consistent difference in pinniped and cetacean TTS-onset levels. The harbour seal data are based upon experiments on a single captive born male (see www.pinnipedlab.org/animals/) that has been the subject of behavioural psychophysical studies at 4 yrs old (Kastak & Schusterman 1996), 9 yrs old (Kastak et al. 1999), and 14 yrs old (Kastak et al. 2005). The Southall et al. (2001) and Schusterman et al. (2003) studies cited above are both conference abstracts and details are lacking, but the available information indicates that these relate to work on the same individual seal. Similarly, the beluga studies were based upon two individuals (20 and 31 yrs old) held in captivity as part of the US Naval research programmes (Schlundt et al. 2000), with Finneran et al.'s (2000) work being based on just one of these individuals. Bottlenose dolphin subjects within these studies also came from a small pool of five individuals (Schlundt et al. 2000) with several of the critical experiments being carried out on only one or two individual males of 30-35 yrs old (eg. Finneran et al. 2000; Experiment 3 in Finneran et al. 2005). The only other data from cetaceans come from another single male bottlenose dolphin whose hearing was studied at the age of 12 yrs old (Nachtigall et al. 2003) and 13 yrs old (Nachtigall et al. 2004).

Secondly, there were important differences in the experimental designs used in studies cited to support this assertion. Most studies used the same behavioural response paradigm (the exception being Nachtigall et al. 2004), with animals trained using operant conditioning to touch an object or produce a vocalisation in response to different sound levels. One important difference in experimental design was that experiments on pinnipeds were carried out in isolated pools at UC Santa Cruz (Kastak et al. 1999; Kastak et al. 2005) whereas those on belugas and most of the bottlenose dolphins were carried out in floating enclosures in San Diego Bay (Schlundt et al. 2000; Finneran et al. 2000). This difference is particularly pertinent because masking noise had to be employed in the beluga studies due to high and variable levels of ambient noise within San Diego Bay (Schlundt et al. 2000; Finneran et al. 2000). Whilst the role of masking noise in marine mammals remains unclear (Finneran et al. 2000), studies in humans indicate that masking noise can result in elevated hearing threshold (Parker et al. 1976; Humes 1980), potentially decreasing the amount of TTS observed and further constraining comparison between pinniped and cetacean datasets. As recognised by all authors, these behavioural response studies also suffer from alterations in behaviour through the experimental period, with many subjects showing behavioural responses to high noise levels that interfered with experimental protocols and would have affected estimates of received SEL, for example where the harbour seal left the water during experiments (Kastak et al. 1999). Along with individual (or potentially species-specific) variability in the level of false alarms (responses in the absence of a signal) (see eg. Kastak et al. 1999), this constrains the power of these studies to provide directly comparable quantitative measures of TTS-onset.

Thirdly, there appear to be differences in the statistical analysis used, most importantly in the way in which the data from each set of experiments were used to estimate TTS-onset levels of 183 dB 1 μPa^2-s for pinnipeds and 195 dB 1 μPa^2-s for cetaceans. It is the difference in these point estimates that is used to infer the 12dB reduction in TTS-onset in harbour seals. This, in turn, is the critical value that feeds through to produce the extreme differences we found in predicted levels of PTS-onset for harbour seal and small cetacean populations around windfarm sites. Although a series of papers are cited to support the 183 dB 1 μPa^2-s value for pinnipeds, our understanding is that these specific figures result from analyses of data in Kastak et al. (2005) (from harbour seals) and data in Finneran et al. 2005 (using pooled data from beluga and bottlenose dolphins in Finneran et al. (2005) and Schlundt (2000). Kastak et al. (2005) exposed the harbour seal to noise at two different levels (80dB SL and 95 dB SL) with two different durations of exposure (25 mins and 50 mins) at 95 dB SL. When considering overall Sound Exposure Levels (SEL), this therefore resulted in only three different treatments. The TTS-onset of 183 dB re: 1 µPa^2-s was predicted from a non-linear regression of TTS vs SEL, based upon data from individual trials (Fig 7 in Kastak et al. 2005). Whilst significant, the relationship was based on only three SEL levels and the r^2 value was only 0.3. Furthermore, the predicted TTS-onset (ie the point of intercept on the x axis where where TTS = 0) was based on the linear portion of the curve, much of which was outside the range of values used in the experiment. As pointed out by the authors *".. the adapted exponential model used here is limited in terms of predicted power. The limitations arise not through the use of the model itself, but from the highly variable, relatively low TTS values and the small number of sound exposure levels used"* (Kastak et al. 2005). In contrast, Finneran et al. (2005) combined data from their study of two male bottlenose dolphins with those from Schlundt et al's (2000) study of bottlenose dolphins and belugas to assess the level of occurrence of TTS across a broader range of SEL. The resulting estimate of 195 dB 1 µPa^2-s is based on an analysis that demonstrated that significant amounts of TTS were observed above this level (see the lower panel of Fig 9 in Finneran et al. (2005)). This is a completely different approach to the linear extrapolation used on the harbour seal data (where there was no significant difference in the levels of TTS in experiments carried out at different source levels (Kastak et al. 2005 Fig 5) or durations of exposure (Kastak et al. 2005 Fig 4)). Given the differences in the way these values were derived, we therefore question whether the point estimates from the studies of pinnipeds and cetaceans are directly comparable.

In summary, and as highlighted by Southall et al (2007), there are no data available to estimate the onset of TTS in pinnipeds exposed to pulsed noise such as that produced from seismic airguns or pile driving activity. However, contrary to Southall et al (2007), we argue that there are insufficient data to support their assertion that there is a *"known pinniped-cetacean difference in TTS-onset upon exposure to non-pulse sounds.."* (Southall et al. 2007). This conclusion primarily results from the limited number of individuals and species studied within both groups, but also results from methodological differences, especially the contrasting statistical approaches that have been applied to these data to predict TTS-onset levels. Consequently, we suggest that with current data it is not appropriate to use the proposed 12dB difference as a scalar to produce exposure criteria for pinnipeds from Finneran et al's (2002) data on TTS-onset to pulsed sounds in cetaceans.

Proposed criteria for PTS-onset in pinnipeds exposed to windfarm construction noise

Southall et al.'s (2007) review and interim recommendations for exposure criteria provide a useful framework for evaluating how underwater noise from offshore windfarm developments may impact protected marine mammal populations. However, our evaluation of the different PTS-onset criteria highlights the critical need for more experimental data on the levels of different types of noise that cause TTS in a wider range of species, and in a larger number of individual subjects. This need is of course widely recognised, both by the authors of the individual research papers and by Southall et al. (2007).

However, these are challenging research questions that will require significant time and resources to address. It is crucial that efforts are made to develop such studies but, given current policy targets, it must be recognised that many environmental assessments for UK windfarms will need to be submitted by developers and reviewed by regulators during 2012. There is therefore an urgent need for an agreed approach for assessing the extent to which protected seal and cetacean populations may suffer from PTS as a result of exposure to noise from the construction of these proposed offshore windfarms.

Given the arguments above, we do not consider it appropriate for this current round of environmental assessments to use different PTS-onset criteria as proposed by Southall et al. (2007). At the same time, we are aware of no ongoing studies that will produce empirical data that would significantly advance our ability to predict likely levels of PTS-onset within these time-frames. Furthermore, we are not aware of any other published scientific evidence suggesting that pinnipeds are more vulnerable to auditory damage from multiple pulsed sounds than cetaceans.

In the absence of evidence of differences in vulnerability to hearing damage between cetaceans and pinnipeds, and given the lack of any studies of pinnipeds that have demonstrated TTS-onset in response to pulsed sounds, we therefore propose to use Southall et al's (2007) M-weighted SEL of 198 dB re: 1 μPa^2-s as a PTS-onset criteria when comparing potential impacts of pulsed sounds such as pile-driving on both pinnipeds and cetaceans.

References

Bailey, H., Senior, B., Simmons, D., Rusin, J., Picken, G. & Thompson, P. (2010) Assessing underwater noise levels during pile-driving at an offshore windfarm and its potential effects on marine mammals. *Marine Pollution Bulletin,* **60,** 888-897.

Finneran, J.J., Carder, D.A. & Ridgway, S.H. (2002) Low-frequency acoustic pressure, velocity, and intensity thresholds in a bottlenose dolphin (*Tursiops truncatus*) and white whale (*Delphinapterus leucas*). *Journal of the Acoustical Society of America*, 111, 447-456.

Finneran, J.J., Carder, D.A., Schlundt, C.E. & Ridgway, S.H. (2005) Temporary threshold shift in bottlenose dolphins (*Tursiops truncatus*) exposed to mid-frequency tones. *Journal of the Acoustical Society of America*, 118, 2696-2705.

Finneran, J.J., Schlundt, C.E., Carder, D.A., Clark, J.A., Young, J.A., Gaspin, J.B. & Ridgway, S.H. (2000) Auditory and behavioral responses of bottlenose dolphins (*Tursiops truncatus*) and a beluga whale (*Delphinapterus leucas*) to impulsive sounds resembling distant signatures of underwater explosions. *Journal of the Acoustical Society of America*, 108, 417-431.

Henderson, D. & Hamernik, R.P. (1986) Impulse noise – critical review. *Journal of the Acoustical Society of America*, 80, 569-584.

Humes, L.E. (1980) Temporary threshold shift for masked pure tones. *Audiology*, 19, 335-345.

Kastak, D., Schusterman, R.J., Southall, B.L. & Reichmuth, C.J. (1999) Underwater temporary threshold shift induced by octave-band noise in three species of pinniped. *Journal of the Acoustical Society of America*, 106, 1142-1148.

Kastak, D., Southall, B.L., Schusterman, R.J. & Kastak, C.R. (2005) Underwater temporary threshold shift in pinnipeds: Effects of noise level and duration. *Journal of the Acoustical Society of America*, 118, 3154-3163.

Kryter, K.D. (1994) *The handbook of hearing and the effects of noise*. Academic Press, New York.

Nachtigall, P.E., Pawloski, J.L. & Au, W.W.L. (2003) Temporary threshold shifts and recovery following noise exposure in the Atlantic bottlenosed dolphin (*Tursiops truncatus*). *Journal of the Acoustical Society of America*, 113, 3425-3429.

Nachtigall, P.E., Supin, A.Y., Pawloski, J. & Au, W.W.L. (2004) Temporary threshold shifts after noise exposure in the bottlenose dolphin (*Tursiops truncatus*) measured using evoked auditory potentials. *Marine Mammal Science*, 20, 673-687.

Parker, D.E., Tubbs, R.L., Johnston, P.A. & Johnston, L.S. (1976) Influence of auditory fatigue on masked pure-tone thresholds. *Journal of the Acoustical Society of America*, 60, 881-885.

Schlundt, C.E., Finneran, J.J., Carder, D.A. & Ridgway, S.H. (2000) Temporary shift in masked hearing thresholds of bottlenose dolphins, *Tursiops truncatus*, and white whales, *Delphinapterus leucas*, after exposure to intense tones. *Journal of the Acoustical Society of America*, 107, 3496-3508.

Schusterman, R., Kastak, D., Levenson, D., Reichmuth, C. & Holt, M. (2003) Noise-induced temporary threshold shift in pinnipeds: effects of exposure medium, intermittence, duration and intensity. Symposium on Environmental Consequences of Underwater Sound. San Antonio, TX.

Southall, B., Bowles, A., Ellison, W., Finneran, J., Gentry, R., Greene Jnr, C., Kastak, D., Ketten, D., Miller, J., Nachtigall, P., Richardson, W., Thomas, J. & Tyack, P. (2007) Marine mammal noise exposure criteria: initial scientific recommendations. *Aquatic Mammals*, 33, 411-520.

Southall, B., Schusterman, R., Kastak, D. & Kastak, C. (2001) Pinniped hearing and anthropogenic noise. *Journal of the Acoustical Society of America*, 110, 2722.

Annex II. 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