THRESHOLDS FOR BEHAVIOURAL RESPONSES TO NOISE IN MARINE MAMMALS

Background note to revision of guidelines from the Danish Energy Agency

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Data sheet

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Abstract:	The available scientific literature on behavioural reactions to pile driving noise by Danish species of marine mammals was reviewed in order to provide guidance on generalized thresholds for behavioural reactions (avoidance) to noise from pile driving of wind turbine monopile foundations. The experimental data were from field studies of reactions of harbour porpoises to full-scale pile driving operations and from playback experiments on captive animals with reduced source levels. Based on these results a generalized threshold for onset of behavioural reactions in porpoises for pile driving noise is suggested to be 103 dB re. 1 µPa, calculated as a root-mean-squared level over 125 ms and weighted with an auditory frequency weighting function resembling an inverse audiogram (VHF-weighting function). Insufficient data was available for seals, dolphins and minke whales and generalised thresholds for these groups could not be provided.

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Foreword

The existing Danish guidelines for mitigation of impact from pile driving (Danish Energy Agency, 2016) were based on advice from a working group (Skjellerup et al., 2015). This working group only considered impact in the form of injury to the hearing of marine mammals, as the empirical evidence regarding other forms of impact was considered insufficient at the time. This has changed in the years since the last revision of guidelines and there is now basis for reviewing this evidence and extract recommendations for including disturbance of behaviour in the upcoming revision of guidelines.

The Danish Energy Agency have chaired a working group on the technical aspects of a revision of the guidelines. Other participants in the working groups were from AU / DCE, the Danish Environmental Protection Agency and the private consulting company Vysus Group, commissioned with drafting the text to the guidelines. As part of this process AU / DCE was commissioned to write a number of technical background reports, of which this is the third. The two other reports concerned selection of relevant species for impact assessments (Tougaard et al., 2020) and criteria and thresholds for noise-induced injury to marine mammals (Tougaard, 2021).

This report addresses three key questions: What are the minimum sound levels marine mammals will respond to by behavioural reactions; what is the speed by which they flee from the noise; and how can the cumulative impact from many disturbances be assessed for larger groups of animals (populations) and for distinct geographical areas (marine protected areas of various sorts).

An earlier draft of this report was presented on a working group meeting and discussed by the group. The analysis and conclusions presented in the report is the sole responsibility of the author, however.

Summary

The available scientific literature on behavioural reactions to pile driving noise by Danish species of marine mammals was reviewed. The relevant species groups were phocid seals (harbour seal and grey seal), harbour porpoises, dolphins (whitebeaked dolphin) and mysticete whales (minke whale).

Only for harbour porpoises was the empirical evidence sufficient to provide guidance on a generalized threshold for behavioural reactions (avoidance) to noise from pile driving of wind turbine monopile foundations. The reviewed results came both from field studies of reactions of wild porpoises to full-scale pile driving and studies in captivity involving playback of pile driving noise at greatly reduced levels. When received sound pressure levels at the onset of behavioural reactions were weighted with the auditory frequency weighting function appropriate for porpoises, the results were consistent across studies. Based on these results a generalized threshold for onset of behavioural reactions in porpoises for pile driving noise is suggested to be 103 dB re. 1 µPa, VHF-weighted and calculated as a root-mean-squared level over 125 ms.

Only two relevant studies were available for phocid seals, one for grey seals and another for harbour seals. Both were field studies on wild seals tagged with satellite transmitters and conducted during a period where offshore wind farm pile driving was undergoing. The lowest sound pressure levels where reactions to the sound was observed was 120 dB re. 1 µPa and 138 dB re. 1µPa, for grey seals and harbour seals, respectively, and both weighted with the appropriate phocid seal auditory weighting function. The limited data does not allow more precise guidance on a generalized threshold for avoidance in seals.

Insufficient data is available for dolphins and mysticete whales and no specific guidance on thresholds for behavioural reactions to pile driving noise could be provided.

1 Impact of noise through disturbance

Impact from disturbance is fundamentally different from impact in the form of acute injury, because each disturbance in itself rarely has any direct impact on the well-being of the animal. Instead, the impact from disturbance is cumulative, i.e. the combined effect of numerous, repeated disturbances over time, eventually having a cumulative impact large enough to affect the survival and reproductive success of the individual. This type of cumulative impact has been described as "death by a thousand cuts" (Todd, 2016).

Underwater sound can affect the behaviour of animals in various ways. The reaction to a sound can be positive (attraction), neutral (increased alertness, orientation behaviour), or negative (fleeing, hiding etc.). In any case, the effect of a disturbance is a change in behaviour from one state to another. By repeated disturbances this means that what is affected is in essence the time budget of the individual: more time is spent on reaction to the noise (no matter what kind of behaviour it is) and less time to spend on other behaviours, such as feeding, resting and nursing offspring. In turn, this will have energetic consequences, as less energy is available through foraging and more energy is spent on the reactions to the noise.

All in all this means that an occasional disturbance has no long-term consequences for the animal, but as frequency and duration of disturbances increase, so does the cumulative energetic impact, until some point where it begins to affect the survival of the animal (for small marine mammals most importantly through loss of blubber for insulation) and reproductive success / fecundity (smaller birth weight of offspring and less milk transferred through suckling, leading to decreased chance of survival of offspring through first winter).

Once disturbances reach a level where enough animals are disturbed often and long enough to affect the average survival and fecundity, this will immediately affect the population development, as survival and fecundity directly affects the population growth rate. However, the population effect is likely to be small, even for large disturbances and very difficult to measure directly. Even for species, such as seals, where very good population development data can be obtained, the fact that many other factors (natural and anthropogenic) affect survival and fecundity as well, means that it is close to impossible to disentangle the effects from each other. The consequence of this is that at present the only viable method for assessment of population level effects of disturbance on marine mammals is through mathematical modelling. These are under development under generic frameworks such as the Population Consequences of Disturbance (PCoD, Booth et al., 2020; New et al., 2014) and species-specific models such as the DEPONS model for harbour porpoises (Nabe-Nielsen et al., 2018).

Even though the fundamental mechanisms underlying the way disturbance affects the energetic state of individuals are well known, the knowledge about the fundamental input parameters to the models are most often the limiting factor. This is certainly the case for species relevant to Danish waters, which means that it is not yet possible to use the models to *accurately* predict effects of acoustic disturbances and thereby provide guidance on the most central question: “when are animals disturbed enough to cause population level effects” (National Research Council, 2005).

For these reasons, regulation of impact from underwater noise cannot at present be based on management objections set at population level, as is otherwise the ideal. A target of, for example, a maximum excess annual mortality from disturbance of x% can easily be formulated (such as is done with for example bycatch, see ICES, 2020), but there are currently no methods to translate this target into disturbance of individuals, i.e. to calculate how many animals can be disturbed and for how long, without exceeding the target.

This situation is not unique to marine mammals and a common solution is to perform assessments and set targets based on fractions of the population or the available habitat affected by the disturbance. The central inputs to the assessment are the distances from the source, where animals are disturbed and the duration of the disturbance (i.e. the length of time it takes before normal, undisturbed behaviour is resumed). Additional factors, such as habituation or sensitization with repeated exposures, behavioural and energetic status of the animals when disturbed and individual differences in sensitivity / responsiveness, are important as well, and should be included to the degree possible (Ellison et al., 2012). No matter what, however, the fundamental input to an assessment of impact through disturbance is a map or set of maps that describes the spatial extent of the disturbance. This can be anything from simple circles with a radius equal to the reaction distance, over more complex polygons describing the spatial extent of the disturbance, to graded maps expressing likelihood or severity of a disturbance. In any case, such maps must then be overlaid with maps of where animals are likely to be found and / or maps of important habitats or habitat quality.

The assessment must include time as a factor, as well as number of disturbed animals. If the absolute number of animals cannot be quantified, then disturbed area can be used as a proxy. Disturbance can then be quantitatively described by histograms or probability density functions, along the lines described for porpoises affected by pile driving in the North Sea (Merchant et al., 2018).

Assessments as described above, where spatio-temporal extent of the disturbance(s) is combined with information about animal abundance can be conducted in several ways, depending on the available information about animals and their habitats. These are listed below in order of preference.

1. Optimally, the total number of animals affected by the temporary disturbance is estimated and can then be related to the total size of the population[1]. This requires that accurate density surface maps (absolute or relative) are available for the species, which can then be overlaid with maps of the temporary disturbance. The management target, or permissible disturbance, can then be expressed as a maximum permitted temporal disturbance in percent of the population. A target of x % thus means that x % of the population can be disturbed for one

[1] What constitutes a relevant population is a separate question, the answer to which is beyond the scope of this note. Throughout this note “population” is taken to mean any group of animals that it makes sense to view as a whole during assessment. A more precise term is probably “management unit”.

day, a fraction x / y % of the population can be disturbed for y days, or any other possible combination.[2]

2. If reliable abundance maps are not available, the assessment of impact can be based on habitat of the target species and the target then expressed as x % of habitat. As above, this means that a maximum of x % of the habitat can be disturbed for one day, a fraction of x / y % of the habitat disturbed for y days, or any other possible combination. Mathematically, this approach only considers animals within habitats assuming an even distribution of animals. Animals outside the habitats are not included in the model.

3. If reliable habitat information is not available either, the assessment of impact can be based on total potential habitat, which for example could be the entire Kattegat, or the Western Baltic[3]. Expression and evaluation of the target is the same as for the two other approaches. Mathematically, this approach is equivalent to assuming that the key species is evenly distributed throughout the entire assessment area.

1.1 Temporary impact in Marine Protected Areas

Often there is an interest in evaluating impact on animals within a well-defined geographical area, such as a Natura2000 area or other marine protected area (MPA). In this case, assessment of impact should consider both impact on individual animals (how many and what type of impact) and the temporary loss of habitat due to disturbance from the pile driving.

An example of how such an assessment of impact from pile driving could be performed is presented below. The only impact considered is disturbance, which is assumed to lead to displacement of porpoises from the area of impact around the pile driving operation and thereby cause a temporary loss of habitat. The magnitude of this temporary habitat loss should be considered in space (put in relation to the total area of the MPA) and in time (the duration of the habitat loss). The absolute number of animals impacted should also be included in the assessment, if possible.

In the schematic example illustrated in Figure 1, four pile driving operations are planned, two inside an MPA, one partly inside and one outside. The parameters required for the assessment are given in Table 1.

[2] The fundamental time unit is of some importance and especially whether one is allowed to operate with fractions of this unit. As an example, if the target is set to 1% per day, this could be interpreted as allowing 100% of the population to be disturbed, as long as the disturbance does not last more than 1 / 100 of a day, or roughly 15 minutes. This can be prevented by setting a lower limit to the duration of disturbances, i.e. requiring that all disturbances must be assigned a duration of at least z hours, irrespective that the actual duration in some cases may be smaller.

[3] Selection of these areas is not trivial, but could, to the degree possible, be aligned with information about overall distribution of subpopulations, or management units.

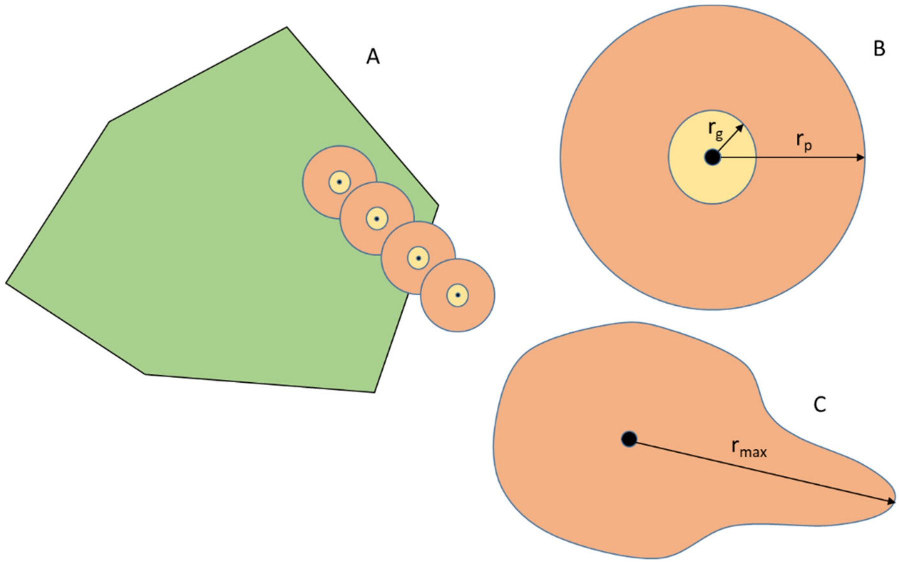


Figure 1. Example of assessment of temporary habitat loss due to pile driving of four foundations inside and outside a marine protected area (green in A). Around each foundation are indicated two disturbance zones (B): one related to pile driving (orange), characterized by the impact range r_p, and zone for general disturbance (from jack-up rig etc; yellow), characterized by the impact range r_g. Impact areas can be modelled as circles, or as more complex polygons, as in (C), by means of appropriate propagation modelling.

Table 1. Input parameters used in the schematic assessment example in the text.

Parameter	Explanation	Value in example
A	Total area of MPA	6,000 km^2
$a_{p,i}$	Impacted area i'th pile driving	380 km^2
Φi	Fraction of i'th impact area inside MPA	[1,1,0.5,0]
a_g	Impacted area for jack-up etc.	0.8 km^2
n	Number of foundations	4
D	Density of porpoises in MPA	1.2 km^{-2}
d_{piling}	Duration of disturbance from piling	0.5 day
d_{total}	Total duration of construction period	5 days

The impact area for each pile driving, $a_{p,i}$, can be found from measured or modelled impact ranges of porpoises to pile driving noise. If they are modelled based on sound propagation models, they can be irregular polygons, such as Figure 1C, which represents maximum distance where the behavioural response threshold is exceeded (see Chapter 3) in different directions from the foundation and based on an appropriate sound propagation model that takes local hydrography, bathymetry and sediment properties into consideration. They can also be found from empirical studies of reaction distances to pile driving noise, in which case they should be represented as circles with a radius equal to the largest distance where reactions were observed.

In the example below the impact ranges are assumed to be the same in all directions for each pile driving and the same for all pile drivings. They are thus calculated based on an assumed maximum reaction distance of 11 km (Brandt et al., 2018):

$$a_p = \pi r_p^2 = 380\ km^2$$

The general disturbance area around the construction vessel is in the same way based on an assumed maximum reaction distance of 500 m (Bas et al., 2017):

$$a_g = \pi r_g^2 = 0.8\ km^2$$

Impact on the MPA can be assessed both in terms of how large a fraction of the MPA is affected and the absolute number of animals affected. The maximum affected fraction of the MPA ($F_{MPAaffcted}$) is found as the ratio between the impacted area of the pile driving with the largest impact area and the total area of the MPA:

$$F_{MPA\,affected} = \frac{\max\left(\Phi_i \cdot a_{p,i}\right)}{A} \cdot 100\ \% = 6.3\ \%$$

The factor Φ expresses the fraction of the impact area associated with each pile driving that falls inside the MPA and can be between 0 (completely outside) and 1 (completely inside).

In the same way as for area, the maximum number of animals impacted at any one time can be estimated by multiplication with the animal density (1.2 porpoise/km2, Hammond et al., 2017):

$$N_{affected} = \mathrm{D} \cdot \max\left(\Phi_i \cdot a_{p,i}\right) = 456\ porpoises$$

As harbour porpoises are known to be highly mobile, it is unlikely that it is the same porpoises that are affected by the first and the last pile driving. This means that it is reasonable to assume a linear trade-off between time and area, within reasonable limits. This means that we assume that the cumulative impact on the MPA and the animals inside the MPA for a large, but short disturbance is equivalent to a small, but longer disturbance, as long as the product of number of disturbed animals and duration of disturbance is constant. By this assumption we can assess the temporary habitat loss (THL) relative to the maximal habitat loss (entire MPA unavailable to animals for the entire duration of the construction, d_{total}). In this calculation it is assumed that each pile driving will cause a disturbance lasting d_{piling} and that there will be a smaller disturbed area around the construction vessel for the rest of the time, characterized by the impact range $r_{general}$. For clarity, the temporary loss caused by pile driving and general disturbance, respectively, are calculated separately:

$$THL_{piling} = \frac{d_{piling} \sum_n \Phi_i a_{p,i}}{A \cdot d_{total}} \cdot 100\% = 1.58\%$$

$$THL_{general} = \frac{\left(d_{total} - n \cdot d_{piling}\right) \cdot a_g \cdot \frac{1}{n} \sum_n \Phi_i}{A \cdot d_{total}} \cdot 100\% = \ 0.005\%$$

It is evident that in this particular case, the temporary habitat loss caused by the general presence of the construction vessel is insignificant compared to the loss caused by the pile driving and can be ignored in the combined assessment.

By multiplying by the density of animals the disturbance can also be expressed in the unit of porpoise×days (PD). This unit should be understood as an equivalent unit. A disturbance with a magnitude of 10 porpoise×*days could represent ten porpoises disturbed for one full day, one porpoise (not necessarily the same porpoise) being disturbed for ten days (not necessarily consecutive days), or any other combination resulting in the same product. As noted above, this assumption of a linear trade-off between animals and duration of disturbance is likely to be reasonable within limits, but unlikely to hold for comparisons between very short, very large disturbances, and very long, but very small disturbances.

$$PD = D \cdot d_{piling} \sum_{n} \Phi_i a_{p,i} = 570\, porpoise \times days$$

The value of the THL and PD parameters is that they allow for relative comparisons such as comparing impacts of different construction scenarios on the same MPA, or the effect of changing construction period to a time of the year with fewer porpoises in the MPA or with hydrographical conditions less favourable for long range transmission of sound.

All of the above metrics are difficult to interpret in an absolute way and any assessment based on them must factor in local conditions and specifics of the particular project and area. A large temporary habitat loss (THL) could for example be assessed as an overall minor impact, if the impacted MPA is located adjacent to one or more additional MPAs, perhaps of significantly larger size, that are not impacted by the project. In contrast, even a small temporary habitat loss could be considered significant if the MPA is considered of particular importance to a porpoise population which is not in favourable conservation status. In the same way, a substantial impact quantified as porpoise×days could be assessed as insignificant, if it occurs in a region where porpoises are abundant and in good conservation status.

What cannot be done is to present general criteria and thresholds regarding temporary habitat loss and number of impacted animals in marine protected areas (nevertheless, see JNCC, 2020a; JNCC, 2020b). At present there are no methods available that will allow derivation of a maximum limit to habitat loss based on first principles, i.e. from population parameters (growth rates, survival and fecundity rates). Some models, such as the conceptual PCAD model (National Research Council, 2005) and the parametrized agent-based models PCoD (Booth et al., 2020) and DEPONS (Nabe-Nielsen et al., 2014; Nabe-Nielsen et al., 2018) represent promising developments, but do not yet have sufficient accuracy or even validation against independent empirical data to be useful in assessment of individual projects.

2 Definitions

Before reviewing thresholds for behavioural disturbances some definitions useful for the review are provided below. Terminology follows ISO 18405 (ISO, 2014) as closely as possible.

2.1 Sound pressure and energy

Sound is pressure fluctuations around the ambient pressure and thus a function of time, *t*, measured in seconds. In the following we denote this as *p(t)*. In most contexts it is relevant to quantify the sound pressure level, L_P, which is found from the root-mean pressure-squared (rms), computed over some interval T:

Equation 2

$$L_p = 20\log_{10}\left(\frac{\sqrt{\frac{\int p^2(t)dt}{T}}}{p_0}\right)$$

Where p_0 is the reference pressure, by convention 1 µPa for underwater sound. The unit of sound pressure level is thus dB re. 1 µPa.

The choice of the duration T over which the rms averaging is performed is central. Often it is selected to match the duration of the sound (see Madsen, 2005 for thorough discussion), but can also be matched to the integration time of the mammalian ear (see discussion in Tougaard et al., 2015). Such a match to the auditory integration time results in a measure, which correlates well with the perceived loudness of the sound. The loudness of short sounds increases with duration of the sound up to the auditory integration time, but remains constant for durations above it. The auditory integration time of mammals are on the order of a few hundred milliseconds and for humans set to 125 ms, the value used in sound level meters (see for example ANSI, 1983).

Often a second measure is used to characterize the magnitude of a sound, the cumulated acoustic energy, also referred to as Sound Exposure Level, SEL.

Equation 3

$$SEL = L_{E,p} = 10\log\int_0^{\tau}\frac{p^2(t)}{p_0^2}dt$$

SEL can be thought of as a normalisation to a duration of 1 second. The SEL of a signal of duration T can be expressed as the 1-second equivalent L_p, or in other words the L_p of a 1-second sound with the same total energy as the sound of duration T. This also means that it is simple to convert from L_p to SEL and vice versa

Equation 4

$$SEL = L_p - 10log_{10}(T), \qquad L_p = SEL + 10log_{10}(T)$$

This means that for sounds shorter than the auditory integration time (125 ms) there is a fixed relationship between SEL and $L_{p,125ms}$ of the sound:

Equation 5

$$L_{p,125\,ms} = \mathrm{SEL} - 10\log_{10}(0.125) = SEL + 9\ dB$$

2.2 Auditory weighting functions

Frequency weighting is the process where the frequency content of a sound is weighted according to some weighting curve, usually roughly expressing the audibility of the sound (loudness). Such frequency weighting is related to the audiograms of animals and has been reviewed in general by Southall et al. (2019) and specifically for porpoises by Tougaard et al. (2015). Southall et al. (2019) offers generic underwater audiograms for five different functional hearing groups, based on review of all available audiogram data for marine mammals. The group audiograms for HF and VHF cetaceans as well as phocid seals are shown in Figure 2.

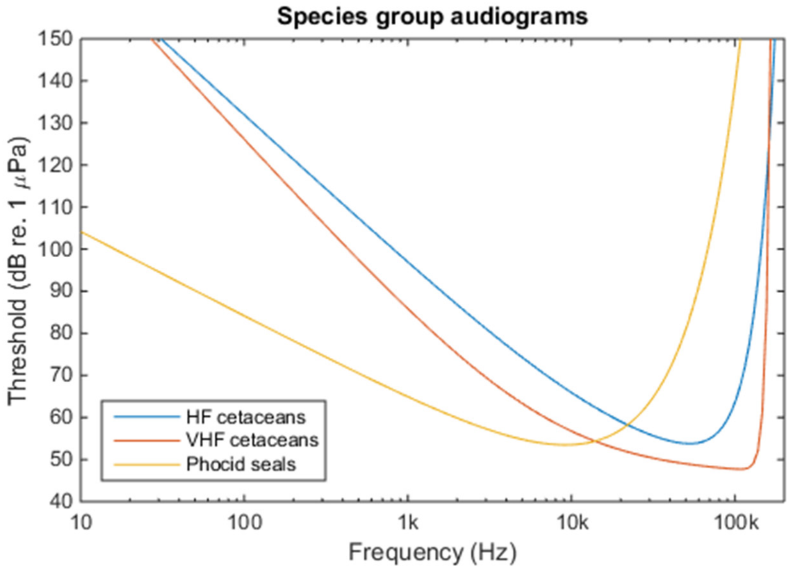


Figure 2. Group audiograms from Southall et al. (2019), based on aggregation of available audiogram data for the different species groups.

The group audiograms give a rough indication of the functional hearing range of the different groups. Table 2 shows the approximate lower and upper frequencies corresponding to the point on the audiogram curves where the threshold is elevated 40 dB relative to the threshold at frequency of best hearing, taken as rough indications of the upper and lower limits to the hearing range[4]. No empirical data is available on hearing in LF cetaceans (baleen whales), so the hearing range must be estimated indirectly. This is done by Southall et al. (2019), based on anatomy of the inner ear and the frequency range of the animals' own vocalisations. In table 2 only the estimates for the minke whale (*Balena acutorostrata*) are given, as it is the only common baleen whale in Danish Waters (Tougaard et al., 2020).

Table 2. Indicative lower and upper limits for hearing range of marine mammal groups. Based on Southall et al. (2019).

	Lower 40 dB point	Upper 40 dB point
Minke whale	10 Hz	34 kHz
HF cetaceans	1 kHz	120 kHz
VHF cetaceans	1 kHz	150 kHz
Phocid seals	40 Hz	50 kHz

[4] Defining the hearing range is very difficult, especially for the gently sloping low frequency part and it is not of particular importance in this context. The indicative limits, based on the 40 dB points on the audiograms, should only be used as a rough guidance for selection of appropriate recording bandwidth and other parameters for recording and analysis.

Application of auditory weighting functions to behavioural response data has only been done in a few studies, such as Tougaard et al. (2015) and Kastelein et al. (2021) (see further chapter 3 below). This is in marked contrast to the case for noise induced hearing loss (temporary and permanent threshold shift, TTS and PTS), where the substantial empirical evidence has been reviewed several times (Finneran, 2015; Finneran and Jenkins, 2012; National Marine Fisheries Service, 2018; Southall et al., 2007; Southall et al., 2019; Tougaard, 2021) and weighting functions have been proposed based on best fit to empirical data (Southall et al., 2019). The applicability of the current weighting functions in assessment of risk of permanent hearing loss for pile driving noise was recently supported by Tougaard (2021).

Thus, rather than proposing new weighting curves tailored to fit the scarce empirical data on behavioural responses, the approach in the review in chapter 3 has been to utilize the weighting functions of Southall et al. (2019) as a first approximation to auditory weighting for behavioural reactions.

2.3 Frequency weighting of pile driving sounds

Application of auditory weighting functions to broadband sounds such as from pile driving and airguns requires knowledge on the frequency spectrum of the noise. A very large number of measurements have been accumulated over recent years for pile driving noise, in particular from offshore wind farms in the German Bight. Most of this data has been summarized and reviewed by Bellmann et al. (2020). Of particular relevance are the average 1/3-octave band spectra obtained at various distances from the pile driving (Figure 3). Each of the four spectra can be frequency weighted with either the VHF or the PCW weighting function (see Southall et al., 2019), appropriate for harbour porpoises and seals, respectively. The weighting factor W, as a function of distance, r (in km), can then be calculated as the difference between the summed power of the unweighted spectrum and the weighted spectrum, consisting of n 1/3-octave bands

Equation 6

$$W_r = 10\log_{10}\left(\frac{\sum_n I_{i,r}}{\sum_n I_{i,r,w}}\right)$$

where $I_{i,r}$ is the unweighted intensity at distance r in the i^{th} band with centre frequency f_i

Equation 7

$$I_{i,r} = 10^{\left(L_{P,1/3-octave}(f_i)/10\right)}$$

and $I_{i,r,w}$ is the corresponding weighted intensity at distance r.

Equation 8

$$I_{i,r,w} = 10^{\left(\left(L_{P,1/3-octave}(f_i)+W(f_i)\right)/10\right)}$$

$W(f_i)$ is the appropriate weighting function (HF, VHF or PCW) at the centre frequency of the i'th 1/3-octave band (equation 2 in Southall et al., 2019).

The resulting weighting factors at increasing distance from the pile driving site and for HF, VHF and PCW weighting are shown in figure 3, right, together with best fitting straight lines. The empirical data points are very well described by straight lines (on a log-x axis), which means that the weighting factors at distance r can be estimated from the following equations

Equation 9 $$W_{HF}(r) = 7.35 \cdot \log_{10}(r) + 33.0\ dB$$

Equation 10 $$W_{VHF}(r) = 8.85 \cdot \log_{10}(r) + 36.5\ dB$$

Equation 11 $$W_{PCW}(r) = 2.24 \cdot \log_{10}(r) + 18.7\ dB$$

where range, r, is given in km.

With these equations it is possible to convert broadband levels to weighted levels and thereby convert estimated reaction distances from empirical studies to weighted levels, as is done in the following sections.

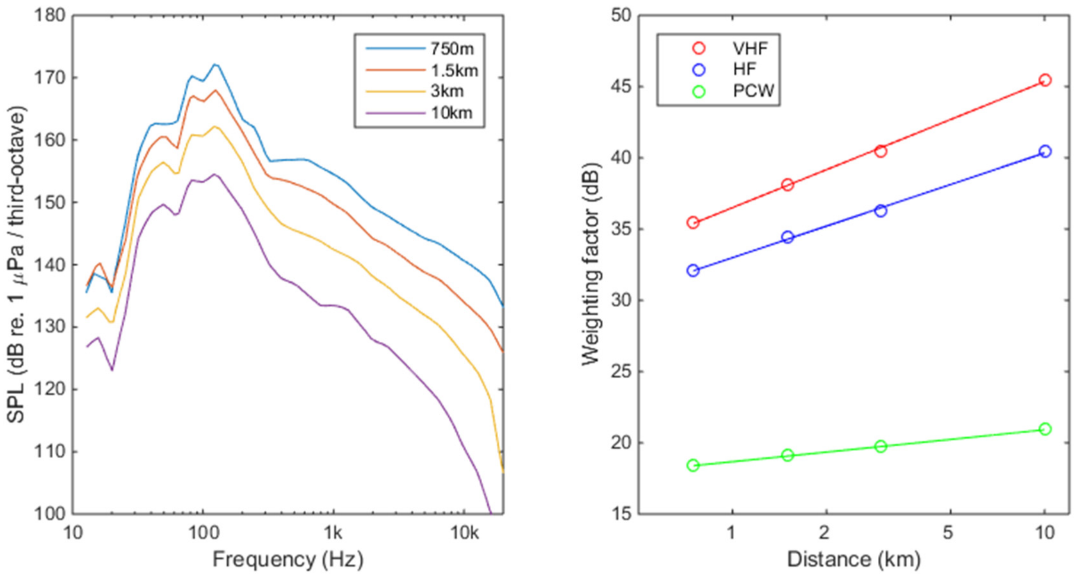


Figure 3. Left: Average frequency spectra of pile driving noise at four different distances from the piling site. From Bellmann et al. (2020). Right: Weighting factor with increasing distance from the piling site for HF (dolphin), VHF (porpoise) and PCW (phocid seal) weighting. The weighting factor is the difference (in energy) between the unweighted and the weighted power density spectrum. The difference in slopes of the three different weighting types is a reflection of the difference in emphasis of higher frequencies, which are absorbed faster with distance than lower frequencies.

3 Behavioural responses to noise

A large number of studies have documented behavioural reactions (disturbance) of marine mammals during exposure to underwater noise. See for example Gomez et al. (2016) for a recent review and Southall et al. (2007) for a comprehensive, yet no longer updated review. There is little consensus, however, with respect to methodology for determining dose-response relationships and deriving generalized response thresholds for behavioural reactions. In particular, the question of frequency weighting is unresolved, i.e. how the different auditory sensitivity across the frequency spectrum can be incorporated into a generalized (species-group specific) threshold, in the same way as for hearing loss and it is beyond the scope of this note to undertake a full review of the literature. Instead, the focus is on the empirical evidence regarding responses to pile driving noise, which is reviewed below for the relevant groups and species of marine mammals.

3.1 Harbour porpoises

Tougaard et al. (2015) reviewed the studies available at the time, where reactions of wild porpoises to different types of sound had been studied and where the received level of sound at the maximum reaction distance was measured or could be estimated. This is shown in **Error! Reference source not found.** top, where $L_{p,\ 125ms}$ at threshold for behavioural reactions (fleeing) is plotted as a function of the dominant frequency of the sound. The thresholds seem to track the porpoise audiogram, offset on the y-axis, indicating that the threshold of response is not an absolute sound pressure level, but rather a level above the lowest audible sound, also referred to as sensation level or loudness[5], indicating that some form of frequency weighting is appropriate. **Error! Reference source not found.** bottom shows the same data, but with sound pressure levels adjusted by the VHF-weighting factor at the respective frequency, following Southall et al. (2019) and Tougaard and Beedholm (2019). Although there is considerable scatter in the data, the plot is nevertheless suggestive of a generalized threshold for behavioural response (fleeing) to noise at around 95 dB re 1µPa, VHF-weighted. The validity of this conjecture, specifically in connection to pile driving noise, is assessed below by evaluation of recent studies.

[5] Sensation level is the difference (in dB) between the sound pressure level of some sound and the hearing threshold for the same sound. In human audiology this is expressed in the unit Son. Strictly, loudness is a psychophysical metric, expressing how loud a sound is *perceived* to be and thereby different from the sensation level. In human audiology, loudness is expressed in the unit Phon, which expresses the sound pressure level (in dB re. 20 µPa) of a 1 kHz tone perceived to be as loud as the test sound. As the threshold of human hearing is close to 0 dB re. 20 µPa at 1 kHz, the Phon-scale is almost identical to the Son-scale for low sound pressures. Due to the different reference pressure in water (1 µPa rather than 20 µPa) the comparison is slightly more complicated for underwater hearing. Nevertheless, sensation level serves as a good first approximation for loudness also for marine mammals.

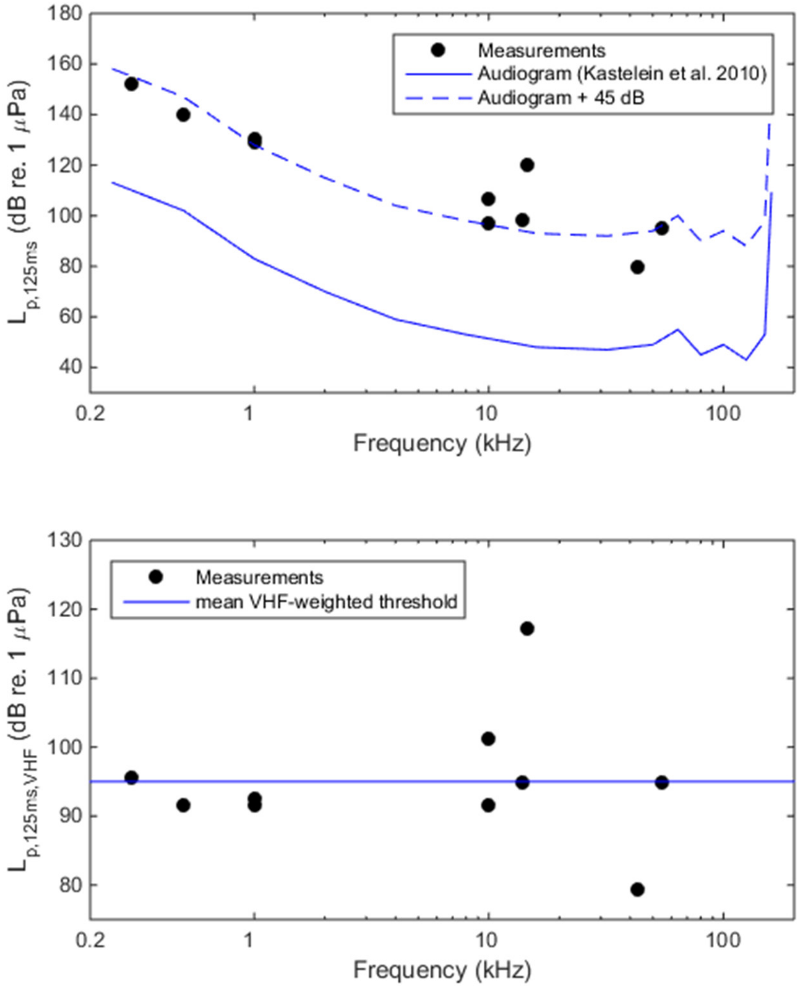


Figure 4. Top: Estimated sound pressure level rms-averaged over 125 ms ($L_{p,\ 125ms}$) at threshold for behavioural responses (fleeing) when exposed to various types of underwater noise pulses (pile driving, seal scarers and gillnet pingers). Blue curve is the harbour porpoise audiogram. Replotted from Tougaard et al. (2015). Bottom: Same data, but sound pressure levels weighted by the VHF-weighting function of Southall et al. (2019).

Response to playback in captivity

Kastelein et al. (2021) tested whether responses of captive porpoises to playback of pile driving noise were best predicted by weighted or unweighted sound pressure levels. This was done by recording the response to a series of six different signals (**Figure** , left), with decreasing amount of energy at high frequencies. The unweighted level was adjusted to be constant across all signals, but the VHF-weighted level decreased with decreasing cut-off frequency of the low-pass filter. VHF-weighted single pulse SEL is reported by Kastelein et al. (2021), which was converted into sound pressure level, $L_{p,125ms,VHF}$, by use of equation 5.

Response to the playback was quantified by two metrics: respiration rate and distance to transducer. Both metrics showed increased response to higher VHF-weighted exposures (Figure 5, right), supporting that sensation level is a more appropriate predictor of response than unweighted sound pressure level. Respiration rate was significantly increased compared to baseline for sounds with VHF-weighted levels 115 dB re. μPa and higher, whereas the mean distance to the transducer was increased for all sounds down to the lowest VHF-weighted sound pressure level of 100 dB re. 1μPa. A threshold as such cannot be determined from the experiment and extrapolation to full-scale pile driving noise and wild porpoises is not straight-forward. Nevertheless, a VHF-weighted threshold below 110 dB re. 1 μPa is clearly suggested.

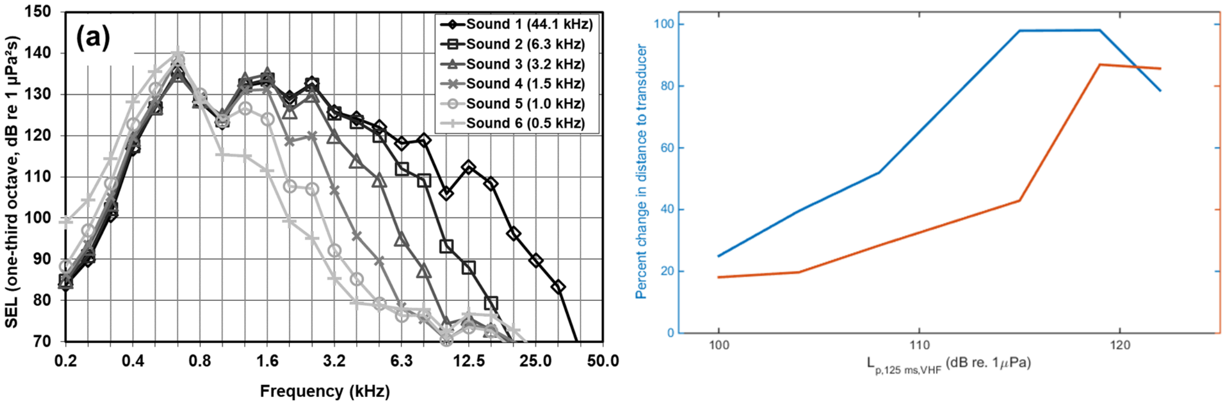


Figure 5. Left: Low-pass filtered pile driving sounds used by Kastelein et al. (2021). See Table 3 for details. Right: Behavioural responses to low-pass filtered pile driving noise and; from Kastelein et al. (2021) and Table 3. The increase in respiration rate at the two lowest levels were not statistically significant.

Table 3. Pile driving sounds used in playback experiment by Kastelein et al. (2021). Compare with Figure 5, left.

Sound no.	Low-pass filter kHz	SEL dB re. 1µPa²s	SEL_{VHF} dB re. 1µPa²s	$L_{p,125\ ms,\ VHF}$ dB re. 1 µPa
1	44.1	135	113	122
2	6.3	136	110	119
3	3.2	135	106	115
4	1.5	135	99	108
5	1.0	135	95	104
6	0.5	133	91	100

An earlier experiment (Kastelein et al., 2013) also with playback of pile driving sounds to a porpoise in captivity includes levels sufficiently low to estimate a threshold. The respiration rate increased significantly at L_p above 136 dB re. 1 µPa, but not at 130 dB re. 1 µPa (figure 6, left). As the 1/3-octave spectrum of the pile driving sound used for playback is available (figure 6, right), it is possible to perform a VHF-weighting by calculating the difference between the unweighted and the weighted spectra (equation 6). The difference (weighting factor) for the pile driving sound used by Kastelein et al. (2013) is 35 dB, which means that the VHF-weighted threshold for behavioural response in that study is between 95 and 101 dB re. 1µPa.

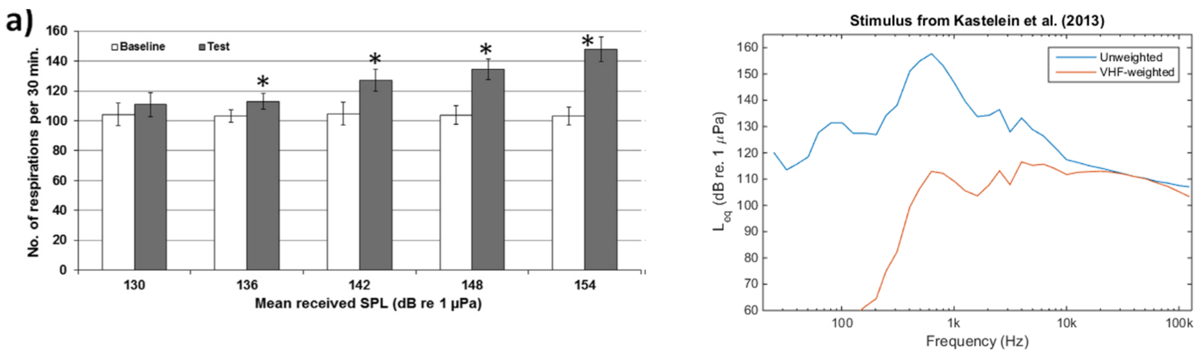


Figure 6. Left: Change in respiration rate of a porpoise during playback of pile driving sound (test) at five different levels. Asterisk indicate a statistically significant change. From Kastelein et al. (2013). Right: 1/3-octave frequency spectrum of the pile driving sound used by Kastelein et al. (2013), both unweighted (blue) and VHF-weighted (orange).

Responses to pile driving at Alpha Ventus

Dähne et al. (2013) studied reactions of wild porpoises to full-scale pile driving at the Alpha Ventus offshore wind farm and were able to couple reactions to estimated received levels of noise. Reactions quantified by passive acoustic monitoring of porpoise echolocation show statistically significant responses to pile driving at a distance of 10 km, but not at 25 km (no intermediate distances were included). The estimated received SEL (unweighted) of single pile driving sounds was 139-145 dB re. 1 μPa^2s at 25 km and 146-152 dB re. 1 μPa^2s at 10 km. This translates into weighted sound pressure levels, $L_{p,\,125ms,\,VHF}$, of 99-105 dB re. 1µPa (25 km) and 110-116 dB re. 1 µPa (10 km), by application of equation 5 and equation 10. These figures indicate that wild porpoises react to pile driving noise above a VHF-weighted threshold in the range 100 – 115 dB re. 1µPa.

Generalized response to pile driving in German Bight

Brandt et al. (2018) performed a unified statistical analysis of passive acoustic monitoring data on porpoises from construction of seven different offshore wind farms in the German Bight. The porpoise monitoring was complemented by acoustic recordings of the pile driving noise, which made it possible to derive a generalized threshold for porpoise reactions to the pile driving noise of 143 dB re. 1 μPa^2s single pulse SEL, reached at distances between 20 and 30 km from the pile driving site. By application of equation 5 and a weighting factor at 25 km of 49 dB (from equation 10) a weighted threshold $L_{p,\,125ms,\,VHF}$ of 103 dB re. 1µPa is obtained.

Responses to pile driving at Beatrice offshore wind farm

Graham et al. (2019) studied reactions to pile driving at the Beatrice offshore wind farm in Moray Firth, Scotland, by passive acoustic monitoring and modelling of received sound exposure levels at different distances. Based on this they derived a 50% probability of response at an unweighted single pulse SEL of approximately 145 dB re. 1 μPa^2s (figure 7, left) for the first pile driving event, consistent with the results of Dähne et al. (2013) and Brandt et al. (2018). As construction went on, the response threshold increased, indicating a diminishing response with time (habituation). Graham et al. (2019) also performed a weighting of received levels with an inverted porpoise audiogram as weighting function (figure 7, right). This procedure is comparable to weighting with the VHF-function of Southall et al. (2019), with the exception of the offset parameter T_0 in Southall et al. (2019), which normalizes the weighting curve to 0 dB at frequency of best hearing, rather than to the actual threshold in dB re 1 µPa. The 50% reaction threshold of Graham et al. (2019) of approximately 55 dB can be converted to $L_{p,\,125ms,\,VHF}$ by adding T_0 (46 dB) plus 9 dB (conversion from single pulse SEL to $L_{p,\,125ms}$, equation 5), which results in an estimated VHF-weighted reaction threshold of 110 dB re. 1µPa.

The two weighting curves are compared in figure 8 by subtracting T_0 from the curve of Graham et al. (2019). The comparison shows that the weighting of Graham et al. (2019) grossly underestimates the contribution of energy at frequencies below 500 Hz, somewhat overestimates the contribution around 1 kHz and somewhat underestimates the contribution above 2 kHz, all when compared to the VHF-weighting function of Southall et al. (2019). As the energy at frequencies below 500 Hz contribute very little to the total VHF-weighted level (due to the poor hearing of porpoises in this range), the deviation between the two curves below 500 Hz likely has no influence for a comparison between the two weightings, whereas the differences above 500 Hz add some dB of uncertainty to the VHF-weighted threshold[6].

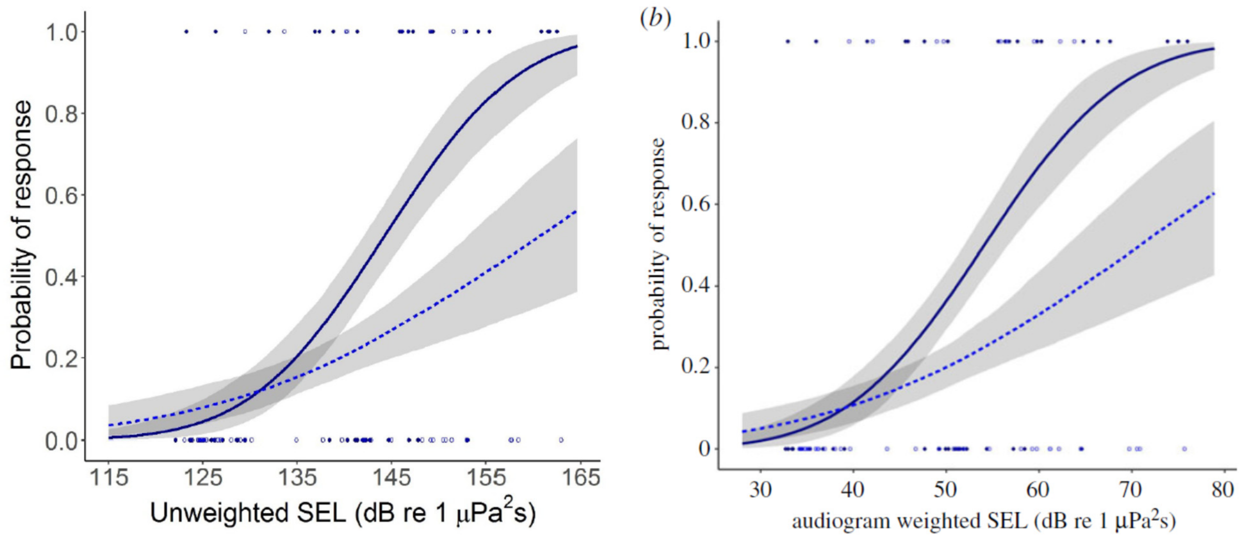


Figure 7. Probability of response to pile driving noise at Beatrice offshore wind farm, Scotland. Solid lines indicate responses to piling of first foundation, dashed line response to piling of last foundation. Left shows responses as function of unweighted, single strike SEL, right shows responses as function of audiogram-weighted levels. See text for further explanation. From Graham et al. (2019).

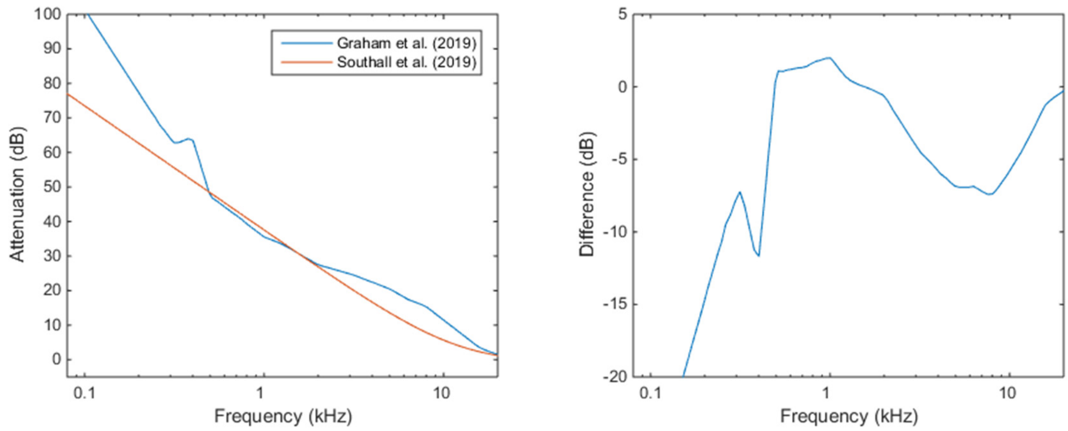


Figure 8. Weighting function used by Graham et al. (2019) compared to VHF-weighting function from Southall et al. (2019). The difference between the two curves is small above 500 Hz. A positive difference, such as in the range around 1 kHz, means that Graham et al. (2019) underestimates the weighted level, whereas a negative value, such as in the range 2-10 kHz means that Graham et al. (2019) overestimates the weighted level, compared with the weighting function of Southall et al. (2019).

Summary of all porpoise thresholds

When all the empirical studies described above are taken together, as done in table 4, it can be seen that there is a large degree of consistency in VHF-weighted thresholds across experiments. In conclusion, the results of the six different studies are consistent with a threshold for behavioural responses of porpoises to pile driving noise at a VHF-weighted level in the range 95-110 dB re. 1 μPa. If a single number rather than a range is desired, it appears

[6] It is beyond the scope of this report to address which of the weighting functions (Southall et. al, 2019 vs. Graham et al. 2019) is the more appropriate. The weighting function of Southall et al. (2019) was selected for the present analysis based on the generic nature of the curve and the fact that if is based on substantially more empirical evidence than the single audiogram used by Graham et al. (2019).

appropriate to use the threshold of 103 dB re. 1 µPa VHF-weighted from Brandt et al. (2018), as this threshold is the modelled average of six different studies of full-scale pile driving operations (including Dähne et al., 2013) and thereby represents the largest amount of empirical data.

Table 4. Summary of VHF-weighted thresholds for behavioural responses to pile driving noise derived from five different studies. See text for further explanation.

Study	Threshold (VHF-weighted)	Comments
Dähne et al. (2013)	100-115 dB re. 1µPa	Based on reaction distance between 10 and 25 km at Alpha Ventus OWF
Kastelein et al. (2013)	95-101 dB re. 1 µPa	Playback of pile driving noise in captivity
Tougaard et al. (2015)	95 dB re. 1µPa	Generalized threshold based on data from pile driving and ADDs.
Brandt et al. (2018)	103 dB re. 1µPa	Modelled threshold based on six OWFs in the German Bight
Graham et al. (2019)	110 dB re. 1µPa	Audiogram-weighted threshold from pile driving at Beatrice OWF
Kastelein et al. (2021)	<100 dB re. 1 µPa	Playback of low-pass filtered pile driving noise in captivity

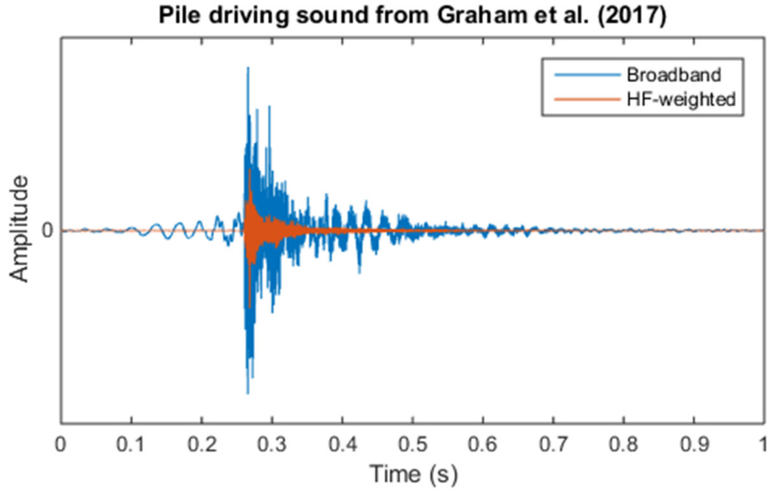
3.2 White-beaked dolphin

No studies have addressed the response of white-beaked dolphins to pile driving noise and only a single study has studied the response of white-beaked dolphins to sound at all (Rasmussen et al., 2016). In this study, different pure-tone and frequency modulated signals of high frequency (100, 200 and 250 kHz) was played to wild dolphins at relatively high received levels and the dolphins reacted to all playbacks. As no thresholds could be derived from the playbacks and frequencies of the sounds were well above those relevant for pile driving, the results cannot be applied to pile driving noise.

Stone et al. (2017) studied a number of cetacean species from seismic survey vessels and the reactions of the animals to airgun arrays. No received levels were estimated, but white-beaked dolphins reacted negatively to the airgun noise (as did most other cetaceans), indicating sensitivity to low frequency noise.

A few studies have looked at reactions of bottlenose dolphins (*Tursiops truncatus*) to impulsive sounds. Although the two species are not closely related, the results from bottlenose dolphins are likely the most relevant data available. Graham et al. (2017) studied reactions of bottlenose dolphins (and porpoises) to impact and vibratory pile driving noise of small-diameter monopiles. Dolphins did not flee the study area, but stayed away from the vicinity of the construction site. Received sound exposure levels (single pulse SEL) were estimated to be between 129 and 133 dB re. 1 µPa^{2}s. As a sample recording of a pile driving sound is supplied by Graham et al. (2017) (figure 9), it is possible to estimate the frequency weighted exposure level.

Figure 9. Sample pile driving sound from Graham et al. (2017), both as unweighted (blue) and weighted (red).



From the signal in figure 9 the conversion factor between single pulse SEL and $L_{p,125ms}$ is determined to be 6.5 dB (difference between maximum of a calculation of $L_{p,125ms}$ with a running window and SEL of the pulse) and the weighting factor W determined as the difference between $L_{p,\,125\,ms}$ for the HF-weighted and the unweighted signal (12.5 dB). Dolphins in Graham et al. (2017) were thus estimated to have been exposed to HF-weighted sound pressure levels between 123 and 126 dB re. 1 µPa.

Fernandez-Betelu et al. (2021) also studied the response of bottlenose dolphins to pile driving noise, but from larger piles at the Beatrice and Moray East offshore wind farms. Dolphin activity was monitored by passive acoustic monitoring (C-PODs) moored 40-70 km away from the wind farms. The dolphins remained in the area, but some changes in their behaviour was noted. Maximum received single pulse SEL was estimated to be 128 dB re. 1 µPa^{2}s, unweighted. No spectrum of the signal is given. Although the environmental conditions in Moray Firth may not be comparable to the German Bight, conversion by means of equation 9 from SEL to $L_{p,125ms,\,HF}$ can be applied as a first approximation. The HF-weighting factor at 40 km distance is 45 dB and after application of equation 5 to convert to $L_{p,\,125\,ms}$ is 9 dB, the estimated maximum received level ($L_{p,125ms,HF}$) for the dolphins is therefore 92 dB re 1 µPa.

With only two available studies on bottlenose dolphins and with large differences in received levels there is not sufficient empirical evidence to derive a threshold for behavioural reactions in white-beaked dolphins exposed to pile driving noise.

3.3 Minke whale

Few studies are available on reactions of minke whales to impulsive sounds. Most studies on mysticete reactions to noise has been done on other species, in particular humpback whales (*Megaptera novaeangliae*), grey whales (*Eschrichtius robustus*) and bowhead whales (*Balaena mysticetus*). Several studies indicate that mysticetes can react to seismic surveys at very large distances, tens of kilometres (all early studies reviewed by Richardson et al., 1995) and also that they may react to pile driving noise at considerable distance. Borsani et al. (2008) thus show reactions by fin whales (*Balaenoptera physalis*) more than 200 km away from a presumed pile driving operation in the Mediterranean Sea.

A few studies have exposed minke whales to low frequency sounds. Sivle et al. (2015) exposed a single minke whale to simulated sonar sounds between 1 and 2 kHz. The minke whale started responding (by swimming away) at received levels of 146 dB re. 1 µPa. As the duration of signals was exactly 1 second, the SEL also equals 146 dB re. 1 µPa^{2}s. Furthermore, as the sonar signals are right in the centre of the LF-cetacean frequency weighting curve, the weighted levels equal the unweighted level (weighting factor ~0 dB).

In a similar fashion, Kvadsheim et al. (2017) exposed two minke whales to simulated sonar, either 1.3-2 kHz or 3.5-4.1 kHz. Negative reactions to the sound were observed at received levels of 146 dB re. 1µPa for the 3.5-4.1 kHz sonar (distance to source 1-2 km) and 156 dB re. 1 µPa for the 1.3-2 kHz sonar (distance to source 6 km). These unweighted levels are also essentially identical to the LF-weighted levels (weighting factor ~0 dB).

As for white-beaked dolphins, it is difficult to extract robust response thresholds for minke whales from these few and diverging results. However, response thresholds appear to be considerably higher than for harbour porpoises (by some 40-50 dB) indicating a lower sensitivity to the noise than porpoises. Nevertheless, due to the (presumed) better hearing of minke whales at low frequencies, the actual reaction distances of minke whales and porpoises could be comparable, i.e. tens of km in the case of pile driving noise.

3.4 Harbour seal and grey seal

Two studies on harbour seals and grey seals, respectively, are directly concerned with reactions to full-scale pile driving noise and can be used as starting point for estimating a generalized response threshold.

Russell et al. (2016) studied how harbour seals equipped with satellite transmitters used the waters around an offshore wind farm in the English North Sea during pile driving and found that seals avoided the wind farm area up to 20-30 km away during pile driving. Based on sound propagation modelling the received level (single strike SEL) where a 50% reduction in area use occurred was between 142 and 151 dB re. 1 µPa^{2}s, unweighted (figure 10). If conversion factors from SEL to $L_{p,\ 125\ ms}$ (+ 9 dB, equation 5) and frequency weighting is applied, the corresponding reaction thresholds are between 129 and 138 dB re. 1 µPa ($L_{p,\ 125\ ms,\ PCW}$), using a weighting factor of 22 dB (equation 11 at 25 km).

In a similar fashion Aarts et al. (2017) studied reactions of tagged grey seals to pile driving in the Dutch North Sea and saw responses (change in swimming speed) up to 30 km away from the wind farm area. Based on sound propagation modelling, Aarts et al. (2017) estimated the threshold for onset of reactions to occur at a received single strike SEL of 133 dB re. 1 µPa^{2}s (figure 11). This corresponds to a weighted level, $L_{p,\ 125\ ms,\ PCW,}$ of 120 dB re. 1 µPa, if conversion from single pulse SEL is applied in the same way as above.

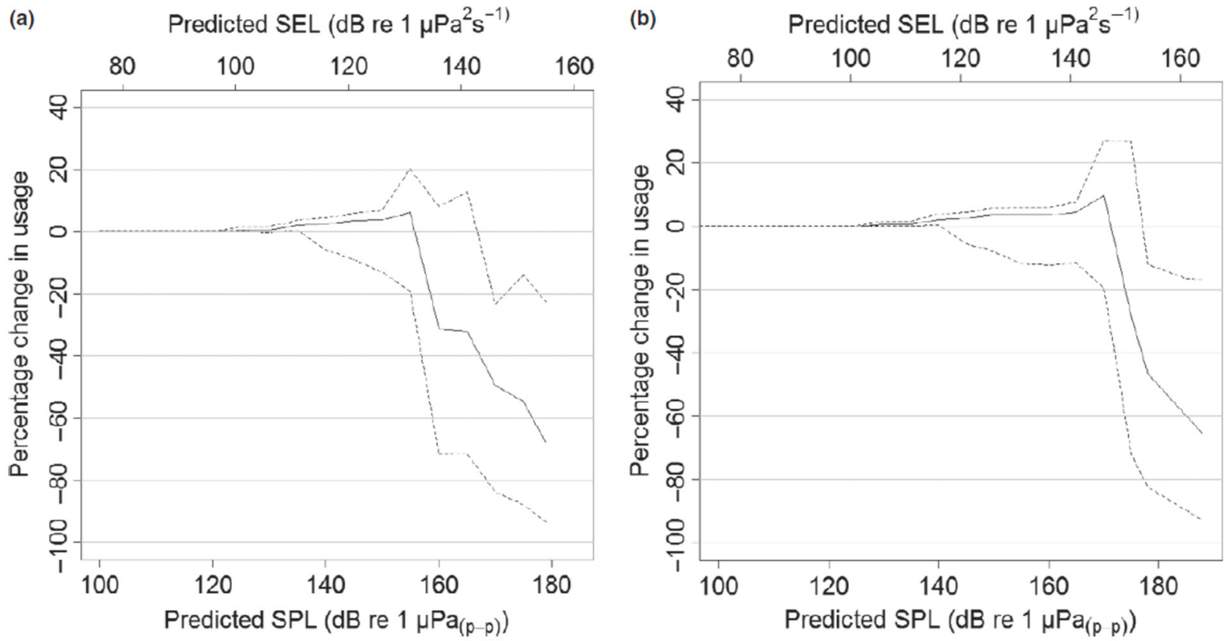
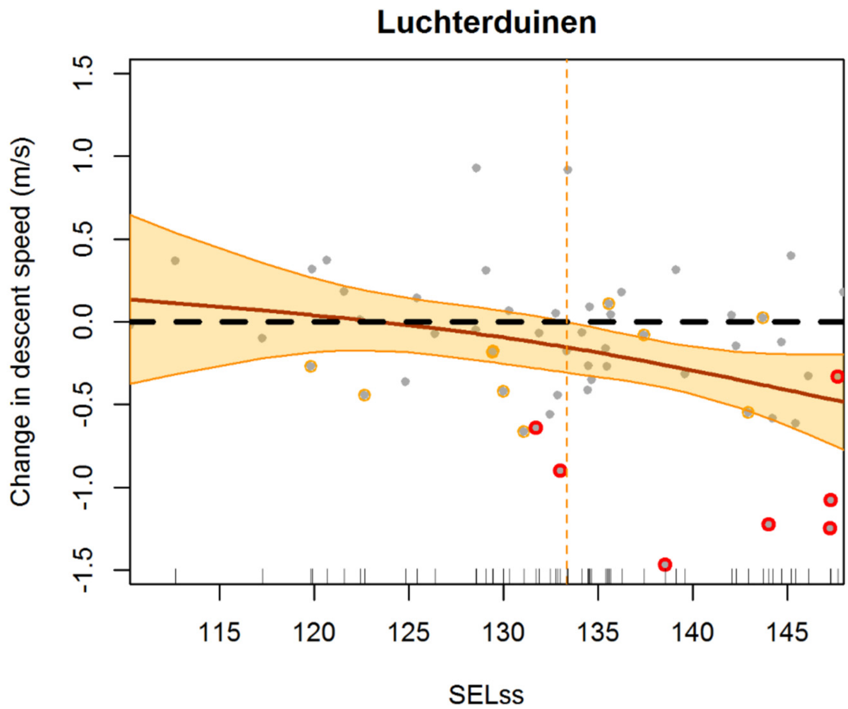


Figure 10. Response in occurrence of harbour seals as function of minimum (left) and maximum (right) modelled noise exposure around a pile driving site. A 50% decrease in usage is found at single strike sound exposure levels between 142 and 151 dB re. 1µPa2s. Dashed lines indicate 95% confidence interval. From Russell et al. (2016).

Figure 11. Response of grey seals to pile driving noise. The response is quantified as a relative change in decent speed during diving and expressed as function of predicted received single strike SEL. The change in descent speed is a reflection of the behaviour of the seals in response to the pile driving noise: slower, shallower dives and then remaining in the surface throughout the duration of the pile driving. From Aarts et al. (2017).



The empirical evidence for seals when it comes to reaction thresholds for pile driving noise is thus considerably better than for white beaked dolphins and minke whales, but not nearly as solid as for harbour porpoises with only one study available per seal species. A few points are worth noting, however. Taken together, the two studies (recalling that they were on different species of seals) are indicative of a PCW-weighted behavioural response threshold for pile driving noise somewhere around the range 120 to 138 dB re. 1 µPa, considerably higher than the threshold for harbour porpoises, indicating a lower responsiveness to noise exposure in seals compared to porpoises. Expressed in terms of loudness (level above the hearing threshold), the results indicate that seals may tolerate exposure to higher loudness levels than porpoises. However, as the hearing of harbour seals (and presumably grey seals) is considerably better than porpoise hearing at the low frequencies dominating pile driving noise, the lower responsiveness to noise is balanced out by the seals' better hearing. Seals and porpoises therefore respond to pile driving noise at roughly the same distances from the construction site (20-30 km).

3.5 Note on possible different effects of noise abatement in seals and porpoises

All of the thresholds derived above relate to unabated pile driving, i.e. without the use of air bubble curtains or similar noise abatement techniques. As the noise abatement techniques now employed in many countries are extremely efficient in removing the higher frequency components of the pile driving noise (above a few kHz), but less efficient at lower frequencies (Dähne et al., 2017; Tougaard and Dähne, 2017), the noise abatement techniques are likely to be more efficient in reducing impacts on harbour porpoises than on seals.

4 Speed of fleeing for different species

When modelling responses of behavioural reactions to underwater noise, for example when estimating the cumulated SEL that an animal receives as it is swimming away from a construction site during pile driving, the assumed swimming speed is critically important. There are very few direct measurements of swimming speed in wild animals during evasion from a noise source, whereas there are more studies that have estimated undisturbed swimming speeds. It is a fair assumption that an animal actively fleeing from a noise source will have a higher swim speed than during undisturbed travelling, but little is known about how long time such higher swim speeds can be sustained. In judging this, the swimming speed where cost of transportation has a minimum can be useful. Cost of transportation is the energy requirement for the animal per travelled distance (equivalent to litres of petrol used per 100 km in a car). While swimming faster is always energetically more expensive (measured as energy expenditure per unit time), the lower cost of transportation means that the total energy spent to travel a fixed distance is lower. It is therefore expected that animals that move over large distances (as most marine mammals) are capable of swimming at the speed with minimum cost of transportation for considerable periods of time.

Some relevant measures of swimming speeds for the relevant species are listed in table 5. A generalized speed of fleeing of 1.5 m/s across all species appears a reasonable precautionary first approximation.

Table 5. Swimming speeds of different groups of marine mammals, derived from the literature.

Reference	Speed	Comments
Harbour porpoise		
Otani et al. (2001)	1.3-1.5 m/s	Speed at minimum cost of transportation
Scottish Natural Heritage (2016)	1.4 m/s	Derived from Westgate et al. (1995).
Kastelein et al. (2018)	1.9 m/s	Swimming speed in captivity during exposure to pile driving noise
Whitebeaked dolphin and other odontocetes		
Fish (1998)	2.5-5 m/s	Bottlenose dolphin in tank
Minke whale		
Williams (2009)	2.1 m/s	Mean swimming speed
Christiansen (2014)	2.5-7 m/s	Speed at minimum cost of transportation
McGarry et al. (2017)	2 m/s	Evading seal scarer
Kvadsheim et al. (2017)	5 m/s	Fleeing from sonar, 1-4 kHz
Harbour seal and grey seal		
Gallon (2007):	1.5 m/s	Harbour seal foraging in tank
Band et al. (2016)	1.6 m/s	Harbour seals in tidal current

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THRESHOLDS FOR BEHAVIOURAL RESPONSES TO NOISE IN MARINE MAMMALS

Background note to revision of guidelines from the Danish Energy Agency

The available scientific literature on behavioural reactions to pile driving noise by Danish species of marine mammals was reviewed in order to provide guidance on generalized thresholds for behavioural reactions (avoidance) to noise from pile driving of wind turbine monopile foundations. The experimental data were from field studies of reactions of harbour porpoises to full-scale pile driving operations and from playback experiments on captive animals with reduced source levels. Based on these results a generalized threshold for onset of behavioural reactions in porpoises for pile driving noise is suggested to be 103 dB re. 1 μPa, calculated as a root-meansquared level over 125 ms and weighted with an auditory frequency weighting function resembling an inverse audiogram (VHF-weighting function). Insufficient data was available for seals, dolphins and minke whales and generalised thresholds for these groups could not be provided.

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