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Environmental Statement

Technical Appendix 7.3 E - Identification of appropriate noise exposure criteria for assessing auditory injury for Pinnipeds using offshore wind farm sites

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

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Document Owner	Nancy McLean
Document Status	Final
File Name	

Revision	Date	Description	Originated By	Checked By	Approved By
A1	November 2011	For review	GH/PT	SJC	NM
A2	May 2012	Final draft	GH/PT	SJC	NM
A3	June 2012	Final			

moray offshore renewables limited

4th Floor
40 Princes Street
Edinburgh
EH2 2BY

APPENDIX 7.3 E

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Contents

Identification of appropriate noise exposure criteria for assessing auditory injury for Pinnipeds using offshore wind farm sites.

Paul Thompson [1] & Gordon Hastie [2]

1. University of Aberdeen, Institute of Biological & Environmental Sciences, Lighthouse Field Station, Cromarty, IV11 8YL, UK
2. SMRU Ltd, New Technology Centre, North Haugh, St Andrews, Fife, KY16 9SR, UK

1. 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 the 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 wind farm 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.

2. Application of noise exposure criteria to assessments of offshore wind farm construction

In the UK, statutory regulators have encouraged the use of Southall et al.'s (2007) criteria to assess environmental impacts of offshore wind farm 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 wind farm 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 (Bailey et al., 2010) and included a strike number of 6,223 per pile, a strike-duration of 600 ms, 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.

3. 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 **d**ata 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).

4. 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 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.

5. Proposed criteria for PTS-onset in pinnipeds exposed to wind farm 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 wind farm 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 wind farms 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 wind farms.

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.

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