Aquatic Mammals 2015, *41*(4), 375-387, DOI 10.1578/AM.41.4.2015.375

A Simulation Framework to Evaluate the Efficiency of Using Visual Observers to Reduce the Risk of Injury from Loud Sound Sources

Russell Leaper,[1] Susannah Calderan,[2] and Justin Cooke[3]

[1]*International Fund for Animal Welfare, 87-90 Albert Embankment, London, SE1 7UD, UK*
E-mail: r.c.leaper@abdn.ac.uk
[2]*Canal House, Banavie, Fort William, UK*
[3]*CEMS, Höllenbergstrasse 7, 79312 Windenreute, Germany*

Abstract

Mitigation measures to reduce the risk of injury to whales from loud sound sources are often based on shutting down the sound source if whales are detected visually within a certain safety zone. Visual detection will only detect a proportion of the whales that enter such a zone, and the likely risk reduction achieved has rarely been quantified. A general simulation model is presented which uses data from sighting surveys and diving behaviour to estimate the probability of detection of a surfacing cue. This can be combined with simple assumptions about sound propagation to estimate the proportion of animals that would be subject to sound exposure levels above a certain threshold, with and without mitigation measures in place. This gives an indication of the mitigation efficiency or the level of risk reduction that can be achieved. Results indicate that there will be many cases where using visual observers results in only a very small risk reduction, but these situations may not always be immediately apparent. Without an adequate quantified assessment of the risk reduction, mitigation measures may often be applied inappropriately or result in regulators granting approval for activities on the basis of measures that do little to reduce risk. The simple simulation model is easy to apply but does need to be performed on a case-by-case basis using input data that correspond as closely as possible to the scenario being investigated.

Key Words: underwater noise, MMO, mitigation efficiency, seismic survey

Introduction

Regulatory authorities in many nations recognise the risk of injury to whales from loud sound sources and particularly seismic surveys. For example, Australian national policy makes a strong case for the need for seismic surveys to avoid whales. It states, "Do not program seismic surveys in areas where and when whales are likely to be breeding, calving, resting, or feeding" and "When planning seismic surveys, avoid where possible areas where and when whales are known or are likely to be migrating" (Department of the Environment, Water, Heritage, and the Arts [DEWHA], 2008, p. 9). New Zealand adopted a new code in 2013, which notes the uncertainty in the effectiveness of measures to reduce impacts and that "the best course of action is simply to avoid conducting seismic surveys in sensitive areas" (New Zealand Department of Conservation [DOC], 2013, p. 5)

Nevertheless, for the majority of seismic surveys, it will not be possible to avoid areas with whales, and so there will inevitably be a level of disturbance and risk of injury. As a result, regulations and guidelines have been devised by many countries where seismic surveys occur which require operators of seismic airguns to implement mitigation measures involving shutdown of the source in response to whales being detected within a specified zone. Other guidelines, such as the widely used Joint Nature Conservation Committee (JNCC) (2010) guidelines, do not require any shutdown. Some operators of military sonar also follow shutdown procedures in the presence of whales (Barlow & Gisiner, 2006). Various decision processes for responding to whale sightings have been employed in different guidelines and regulations. These are generally based around a safety zone of a distance from the source at which shutdown would occur if a cetacean was seen within that zone or thought likely to enter the zone. These distances are often, but not always, based on expected received sound levels.

Quantifying the effectiveness of such mitigation strategies has rarely been attempted; however, this is critical if the best overall mitigation strategy is to be determined and to assess whether the use of mitigation measures is likely to reduce risk to an acceptable level. There has been considerable discussion of possible noise exposure criteria for marine mammals, such as Southall et al. (2007), but there is also a need to quantify the effectiveness of measures designed to reduce the number of animals exposed to levels in excess of the specified criteria. There have been previous reviews of global seismic guidelines such as Weir & Dolman (2007) and Compton et al. (2008). Those papers focused on many of the practical details of implementing the guidelines. There certainly are challenges in this implementation. For example, there is generally a requirement for training of marine mammal observers (MMOs) for seismic mitigation, but even trained MMOs may have very limited experience at sea. Mori et al. (2003) found that the overall sighting rates of Antarctic minke whales (*Balaenoptera bonaerensis*) by observers categorised as having limited experience (fewer than four survey seasons) were 42% lower than experienced observers. Nevertheless, even observers classified in that study as having limited experience still spent several months at sea and so would be much more experienced than many MMOs.

Even if guidelines are followed as intended by experienced personnel, an overall assessment of the likelihood of reducing impacts by responding to sightings from MMOs is still lacking. Parsons et al. (2009) note that for many species, only a small proportion of animals within mitigation zones are likely to be detected by visual observers, but that this reality is generally not acknowledged within seismic guidelines. They also suggest that visual surveys alone as a mitigation measure may be little more than a "public relations exercise" by "giving management authorities, oil and gas companies and the public a false sense of security that seismic survey impacts are being mitigated" (p. 649).

Despite these concerns, and the lack of formal quantification of risk reduction, some regulators have made strong claims about the effectiveness of their guidelines. In the United Kingdom, the relevant guidelines are described in JNCC (2010) and state that "It is considered that compliance with the recommendations in these guidelines will reduce the risk of injury to [protected species] to negligible levels" (p. 3). Although "negligible" is not quantified by JNCC, the implication is that following the recommendations should result in a substantial (well over 50%) reduction in risk since any smaller reduction could not be considered to alter a situation of concern into one of negligible impact, in spite of there being no requirement in the guidelines for a shutdown of the source in the event of marine mammals being detected within the mitigation zone. Although the effectiveness of current mitigation guidelines has not been assessed, there is still an assumption that they are, at best, efficient and appropriate, and, at worst, "better than nothing," in that following shutdown procedures will at least mitigate some impacts on some animals. Confidence in guidelines that are largely untested may result in a reluctance to investigate alternative mitigation options, including reduction of the noise at source, which might prove more effective in decreasing exposure risks.

This study makes a start towards addressing the uncertainties with current measures through a simulation that allows some quantitative assessment of current mitigation strategies. The simulation considers a situation where an operator of a seismic survey has decided that a specific sound source is required to achieve the goals of the project, and it is not feasible to schedule the operations at a time when marine mammals are unlikely to be present. Under these circumstances, a commonly used mitigation measure is to specify a safety zone around the source, based on the source output levels, and to reduce power output or shut down the source entirely if visual observers detect animals within the zone or likely to enter it. This is the approach taken by regulators for seismic surveys in many countries (Weir & Dolman, 2007). Given that the JNCC guidelines do not provide for such shutdowns, the potential effectiveness of the JNCC guidelines is not considered here. Where there are a specified set of mitigation actions in response to information from visual observers, the effectiveness of mitigation can be investigated through simulation. The basic components of the simulation are as follow:

- The visual detection process, including factors that affect detectability (e.g., sighting conditions) and availability (e.g., whale diving behaviour and whale movement)
- Whale aggregation patterns (a shutdown for one individual may affect others that were not detected)
- The cumulative exposure of a whale to the sound source
- The response of the operator to a visual sighting, including response time and rules for shutdowns

These components can be incorporated into a relatively simple simulation that can give an estimate of the mitigation efficiency for a particular situation. For the purposes of this study, mitigation efficiency (M_e) is defined as the proportion of animals that would have been exposed to sound levels above a specified level that are no longer exposed due to the mitigation response.

Methods

Visual Detection Process

The visual detection process has been studied in detail during numerous analyses of sighting surveys for cetaceans. The probability of an animal or group being seen depends on the frequency with which it comes to the surface (its availability) and the strength and duration of the visual cues—for example, size and persistence of blows, splashes, or the amount of body that is visible at the surface (detectability). Cue detection probability will also be a function of weather sea and swell conditions, the radial distance of the cue from the observer, observer height, and the efficiency of the individual observer in detecting the cue. The detection probability for animals that come to the surface within the range of detection has been estimated based on experiments with independent observers (e.g., Hammond et al., 2013); however, these methods cannot fully account for availability for long-diving species, which may not surface during the period that the vessel is within sighting range. Correcting for availability bias requires some model of diving behaviour.

The simulation considered herein needs to take into account the differences in vessel speed between sighting surveys and observations carried out onboard seismic vessels (seismic vessels typically travel at about 2.5 ms^{-1} compared to 5 ms^{-1} for many surveys for abundance estimation), the number of observers (there is often only a single MMO at any one time), and weather conditions (often seismic surveys and, therefore, observations by MMOs, continue in sighting conditions that would not be considered suitable for sighting surveys—e.g., sighting surveys for harbour porpoises are generally only conducted in sea Beaufort state 3 or less [Hammond et al., 2013]). The simulation process involved whales that come to the surface at a specified interval and emit a number of cues during a specified surface time. Each cue had a probability, $P(r)$, of being detected by an individual observer, where r is the radial distance from the vessel to the whale given by

(1) $$P(r) = \frac{e^z}{1+e^z}$$

and

(2) $$z = a_0 + a_1 r + a_2 r^2 + a_3 r^3$$

Multiple observers were assumed to be independent such that $P_T(r)$, the cue detection probability for the observer team as a whole, is

(3) $$P_T(r) = 1 - (1 - P(r))^n$$

for n observers within the team.

For simplicity, whales were assumed to travel in straight lines with constant speed, v, and visual observers were assumed to cover the 180° sector ahead of the vessel equally. The process for ensuring that the distribution of whale headings generated by the simulation was unbiased in relation to whale speed was the same as described by Leaper et al. (2010) based on Hiby (1982). Simulated whales entered a box 6 km ahead of and 6 km to either side of the vessel trackline. Thus, 6 km was considered the maximum distance at which a whale could be detected by any method; and in all scenarios, cue detection probability was set to 0 at distances of greater than 6 km.

Sighting surveys targeting different whale species in different areas were reviewed to gather a range of estimates of overall cue detection probabilities. These are usually expressed in terms of g(0) (the proportion of animals directly on the trackline that are detected) and effective strip half width (eshw) (Table 1). Where g(0) is assumed to be 1, eshw is the distance from the trackline at which the number of whales missed within the strip is, on average, equal to the number of whales seen outside it. Distance analysis generally involves the fitting of a function to observed perpendicular distance data. Commonly fitted functions are a hazard rate or half-normal (Buckland et al., 1993). For these functions, the overall detection probability within the eshw is around 0.8g(0). For the half-normal which is quite peaked, this value is 0.79; and for a flatter hazard function with a shape parameter b = 3 (see (Buckland et al., 1993), it is approximately 0.85.

It can be seen from Table 1 that there is considerable variation in estimates of g(0) and eshw between surveys, even of the same species under similar conditions. Therefore, the cue detection probabilities used in the simulation were not initially conditioned on any one dataset but selected to be broadly representative of naked eye searching. Initial parameter values a_0 to a_3 were based on analyses of Norwegian minke whale survey data searching with the naked eye (Cooke & Leaper, 1998). These parameters gave a detection function most closely fitted by a half-normal. The initial parameters were then iteratively adjusted by a single multiplicative factor k applied to a_1 to a_3 to reflect the sightability of different species with distance from observer.

Table 1. Some examples of estimates of g(0) and effective strip half width (eshw) from sightings surveys. In these analyses, g(0) and eshw were estimated independently—that is, the expected number of whales N detected along transect length L is given by where D is the density.

Species	Region	g(0)	eshw	Survey reference
Fin whale (*Balaenoptera physalus*)	NE Atlantic	0.81	1.1-2.4 km	Víkingsson et al., 2009
Fin whale	Antarctic		2.5-3.4 km	Branch & Butterworth, 2001
Fin whale	West Greenland		0.9 km	Heide-Jørgensen et al., 2007
Blue whale (*Balaenoptera musculus*)	California coast	0.90	2.2-3.2 km	Calambokidis & Barlow, 2004
Blue whale	NE Atlantic		2.1-3.4 km	Pike et al., 2004
Blue whale	Antarctic		2.9-3.9 km	Branch & Butterworth, 2001
Blue whale	Sri Lanka		1.3 km	Priyadarshana et al., 2014
Sperm whale (*Physeter macrocephalus*) (long-diving males)	Antarctic	0.32[1]	3.5 km	Kasamatsu & Joyce, 1995
Beaked (single)	Antarctic	0.27[1]	0.5 km	Kasamatsu & Joyce, 1995
Beaked (≥ 4)	Antarctic	0.27[1]	1,000 m	Kasamatsu & Joyce, 1995
Sperm whale (mainly female groups; 25 min dive followed by 5 min at surface). Two observers searching with Big Eye 25× binoculars	Eastern Tropical Pacific	0.87	3,600-4,600 m	Barlow & Taylor, 2005
Sperm whale	Antarctic		0.13-0.36 km	Branch & Butterworth, 2001
Harbour porpoise (*Phocoena phocoena*)	North Sea	0.34	126-358 m	Hammond et al., 2002
Harbour porpoise	North Sea	0.22		Hammond et al., 2013
Minke whale (*Balaenoptera acutorostrata*)	North Sea	0.82	0.23-0.42 km	Hammond et al., 2002
Minke whale	NE Atlantic	0.43-0.51		Schweder et al., 1997
Minke whale	NE Atlantic	0.54		Hammond et al., 2013
Minke whale	NW Pacific	0.82[2]		Okamura et al., 2009
Antarctic minke whale (*Balaenoptera bonaerensis*)	Antarctic		0.7-1.1 km	Branch & Butterworth, 2001
Antarctic minke whale	Antarctic	0.42-0.59	0.44-0.65 km	Okamura & Kitakado, 2007

[1]Model based estimate based on three dedicated observers searching with binoculars

[2]The estimates of g(0) were 0.754 (CV = 0.33) for top barrel, 0.668 (CV = 0.45) for IO platform, 0.447 (CV = 0.77) for upper bridge, and 0.822 (CV = 0.26) for top barrel and upper bridge

To fit to a particular species scenario (i.e., data from a sighting survey), the simulation was first run with vessel speed set to survey speed and the number of observers set to what was used in the survey (usually two for a single platform with observers searching either side of the vessel). k was then adjusted to match the reported eshw. The estimated g(0) from the simulation was used as a validation check where this could be compared to the g(0) estimated from the actual survey. Once parameters were chosen to fit the sighting data, then the vessel speed and number of observers were adjusted to suit the mitigation scenario. In addition, simulation of mitigation was conducted for different whale swim speeds since these can have a substantial effect on mitigation efficiency but little effect on the sighting detection function.

Most MMOs on seismic vessels search with the naked eye. Searching with binoculars increases the average distance of detected cues but results in a higher fraction of nearby cues being missed because at any one moment the observer's field of view spans a narrower angle. Depending on the observer search behaviour, the sighting conditions, and the diving behaviour, g(0) and eshw can be either higher or lower with binoculars compared to the naked eye. Sometimes both g(0) and eshw are highest for naked eye; sometimes both are highest with binoculars (either with 7× or for 25× magnification big eye binoculars); and sometimes g(0) is higher with naked eye, but eshw is higher with binoculars. Similar considerations apply to comparing different powers of binoculars. For these simulations, the method of searching was assumed to be the same as the closest equivalent survey data.

Sound Exposure

Quantifying the exposure of an animal to sound requires a number of assumptions. Even if the characteristics of the source across the relevant range of frequencies are accurately known, it is difficult to predict the received level for an animal at a certain depth and distance. Certain types of injury may depend on the maximum received sound pressure level; whereas other impacts will depend on the cumulative effect of sound exposure integrated over time. These issues have been extensively reviewed by Southall et al. (2007) who proposed noise exposure criteria for cetaceans and pinnipeds. They classified sounds into three basic types: (1) single pulses, (2) multiple pulses, and (3) nonpulses. For this study, it is assumed that the sound sources being used would be classified as multiple pulses. Southall et al. (2007) suggest a Sound Exposure Level (SEL) for multiple pulses calculated by

(4) $$SEL = 10\log_{10}\left\{\frac{\sum_{n=1}^{N}\int_0^T p_n^{\,2}(t)dt}{(p_{ref})^2}\right\}$$

where instantaneous sound pressure, *p,* is measured for *n* exposures and p_{ref} is 1 µPa. This is essentially an equal energy criterion in that the SEL for two pulses is equivalent to the SEL of a single pulse with twice the energy. Most noise standards now use the equal energy rule (Starck et al., 2003). If it is assumed that pulses are identical and that received pressure level for any pulse can be expressed as a fraction of the source level, and if the energy within each pulse is expressed as E_{pulse}

(5)

$$E_{pulse} = \frac{\int_0^T p_n^{\,2}(t)dt}{(p_{ref})^2}$$

then

$$SEL = 10\log_{10}(E_{pulse}) + 10\log_{10}\left\{\sum_{n=1}^{N} s_n\right\}$$

where s_n is the spreading loss for each pulse $n = 1$ to N in terms of a multiplicative factor of P_n. For example, in the case of spherical spreading for a pulse at a received distance of 100 m, S_n = 1/10,000.

The effectiveness of mitigation measures can thus be expressed as the difference in SEL with mitigation compared to no action relative to the SEL of a single pulse. In the case of a shutdown, the s_n would be 0 for pulses $n = i$ to $n = j$ during which shutdown occurred. Within a limited range of pulse intervals, the effect of pulse interval on SEL is effectively an additive term in equation 3. Thus, the effect of mitigation on SEL is largely independent of pulse interval. For example, changing from a 7.5-s interval to 15-s interval would decrease SEL by around 3 dB across all individuals—both with and without mitigation.

For a stationary whale, equation 5 can be used to estimate the cumulative SEL from a pass by a seismic vessel for any closest approach distance. This can be expressed relative to the SEL of a single shot (Figure 1). Assuming a 25-m shot spacing and either 20log(r) or 15log(r) propagation loss gives the estimates for different distances of closest approach (Table 2). These estimates give an indication of whether it is worth considering visual observers as a mitigation option. This is very dependent on propagation conditions and the source level.

The assessment of risk and what might constitute an appropriate safety zone will depend on the properties of the source, the propagation conditions, and the exposure criteria. Hildebrand (2009) suggests a pulse energy for a 2,000 psi and 8,000 in^3 airgun array of 241 dB re 1 µPa^2-s. More typical industry standard airgun arrays comprise about 28 airgun elements totalling around 3,100 in^3 of volume. The New Zealand guidelines refer to measurements from a 3,250 in^3 airgun array with reported levels around 174 dB re 1 µPa^2-s at 200 m and a maximum of 165 dB re 1 µPa^2-s at 1 to 1.5 km. Measurements from a 3,090 in^3 volume source averaged over the azimuth in the horizontal plane suggested an equivalent energy level for a single shot of 228 dB re 1 µPa^2-s (Parnum & Duncan, 2012). Modelling of this source also suggested 20log(r) spreading loss to be an appropriate overall assumption for ranges

Table 2. Distances at which total received energy during a pass would be at a certain level relative to a single pulse

SEL relative to single pulse at 1 m (dB re 1 µPa^2-s)	-20 dB	-25 dB	-30 dB	-35 dB	-40 dB	-45 dB
Closest approach during a pass (m), 20log(r) propagation loss	18	46	132	389	1,118	2,860
Closest approach during a pass (m), 15log(r) propagation loss	312	1,630	5,270	12,750	28,150	61,000

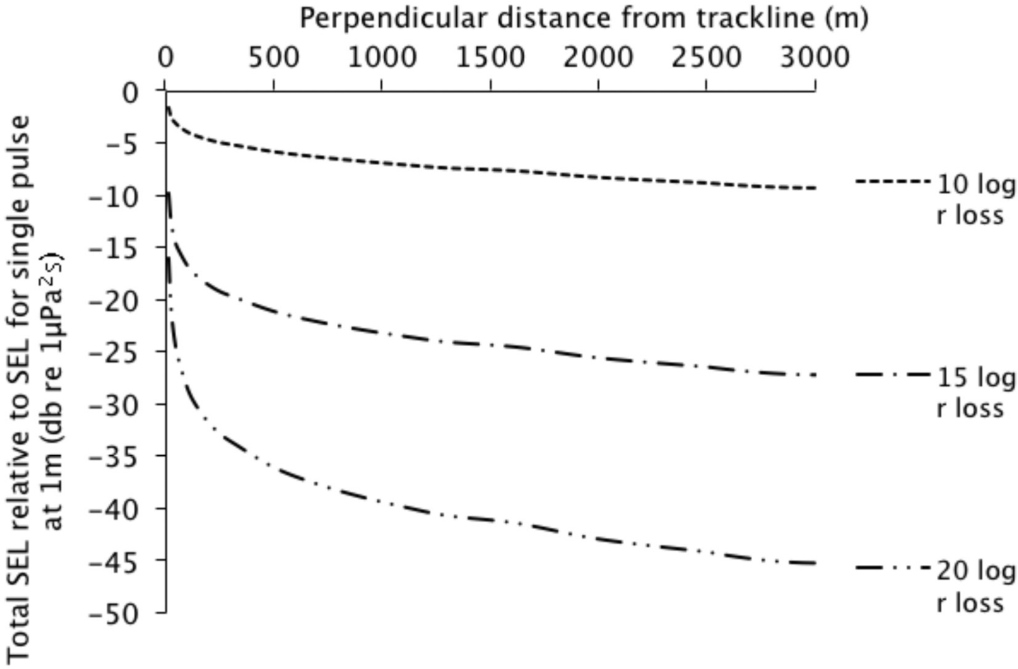

Figure 1. Change in total SEL for a pass with perpendicular distance from the trackline for three different assumptions of propagation loss

up to 1 km for several different scenarios of depth and bottom type (Parnum & Duncan, 2012).

Mitigation Response

Mitigation options in response to sightings of cetaceans may involve either reducing source power or shutting down altogether. In both cases, there will be a time delay between a sighting and a mitigation response. However, for the purposes of this paper, it was assumed that there would be an instant response. To allow testing by simulation, it was also assumed that the response was a total shutdown according to two types of decision rule:

(i) Instant shutdown following any sighting of species considered at risk
(ii) Instant shutdown following any sighting within a specified "safety zone" based on perpendicular distance from the trackline—that is, any whale seen within the specified perpendicular distance of the trackline would trigger a shutdown even if it was still greater than that distance ahead of the vessel (distance from trackline was used to simulate a response to whales that would be expected to come within the shutdown zones; whereas in practice, a whale often has to be seen within a specified radial distance from the source to trigger a shutdown).

Option (i) was included in the simulation to investigate cases in which whales occur in aggregations and mitigation efficiency could be improved by shutting down in response to any sighting, which might then protect other whales in the area. However, this does increase the likelihood of shutdowns when the closest whales are outside the designated safety zone.

For a specific scenario, the simulation output included the proportion of animals within 6 km of the trackline exposed to cumulative levels relative to the SEL of a single pulse. The mitigation efficiency, M_e can then be estimated as the proportion of the number of animals no longer exposed (M1 to M0) as a fraction of the number that would be exposed without mitigation (M1) for any cumulative SEL (Figure 2). The assumption of instant shutdown will not be achieved in practice however assiduous the observers and operators. In addition, shutdowns may only be initiated if a whale is observed within the safety zone radius rather than within a perpendicular distance of the vessel's track. Both of these factors will contribute a positive bias to M_e within the simulation results.

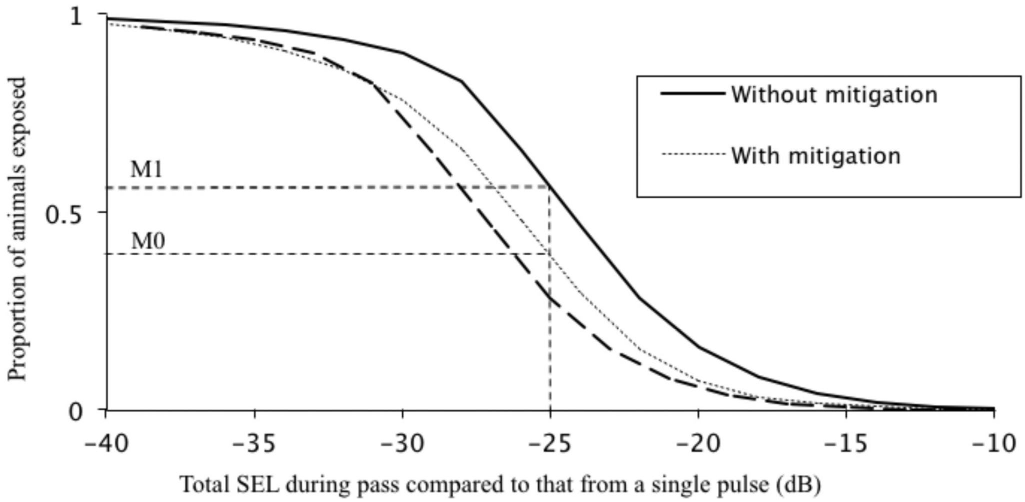

Figure 2. Example output from simulation illustrating calculation of M_e for a specified SEL (in this case -25 dB relative to the SEL of a single pulse); dashed line illustrates the proportion of animals exposed if source level was reduced by 3 dB throughout the seismic operations.

Results

The value of g(0) achieved by the MMOs will essentially set an upper bound on M_e so a set of simulations was conducted to investigate the effect of vessel speed on g(0) estimates for different species. For the most conspicuous species, g(0) increased by about 30% when lowering speed from typical survey speeds of 10 kts to typical seismic survey speeds around 5 kts. For less conspicuous or longer diving species, the increase was around 100% (Table 3). These results gave a good fit ($R^2 > 0.99$ in all cases) to the theoretical relationship in equation 4 with values of *a* and *b* for each scenario shown in Table 3:

$$g(0) = 1 - ae^{-(b/v)}$$

where *v* is the vessel speed in ms^{-1}. In some mitigation scenarios such as for military sonar, vessel speeds may be greater than typical survey speeds. Hence, vessel speeds up to 10 ms^{-1} (20 kts) were investigated in the simulations.

Blue whales were selected as a case study because they represent one of the most conspicuous species for which there are estimates of g(0) and strip width from sighting surveys. Visual observations from MMOs have been proposed for mitigation during seismic surveys in some areas where blue whales were the main species of concern (e.g., Great Australian Bight). Most sightings surveys for blue whales assume g(0) = 1, and typical eshws for blue whale surveys with observers searching with binoculars are between 2 to 3 km (e.g., Branch & Butterworth, 2001; Calambokidis & Barlow, 2004). For average group sizes of blue whales comprising fewer than 1.5 individuals, Calambokidis & Barlow (2004) estimated an eshw of 2.2 km and g(0) of 0.9 pooling data across Beaufort sea states 0 through 5. These sea states are probably also representative of seismic survey conditions. Their survey speed was greater than a typical seismic vessel, but they also had three observers: one searching with naked eye and 7× binoculars and two with 25× binoculars. For the purposes of simulation with the naked eye or only 7×50 binoculars, a strip width of 1,500 m was selected, and parameters were adjusted to achieve this. This strip width falls between 2 to 3 km for observers using binoculars and 1.3 km for naked eye observers during surveys off Sri Lanka (Priyadarshana et al., 2014). Dive times of blue whales vary, but dive times for blue whales off Chile used to estimate g(0) for aerial surveys ranged from 149 to 487 s (Galletti Vernazzani, 2012). Croll et al. (2001) report results from time depth recorders on Balaenopterid whales with a mean dive duration of 6.8 min. Off Sri Lanka, De Vos et al. (2011) found mean times for long dives of around 10 min. Dive time will have a substantial impact on mitigation efficiency, and so sensitivity tests were carried out with dive times of 300 and 600 s (Table 4).

This case study illustrates some general characteristics. With increasing whale movement speeds, fewer animals will be exposed to the same

Table 3. Variation in g(0) with vessel speed from simulation results, assuming a single observer and good (survey equivalent for the species in question) sighting conditions. Surfacing and diving behaviours are examples for that species or group and may vary between areas and populations.

Scenario	BW	MW	PP	SW	BK
	Blue whale: 5 quick surfacings followed by 300 s dive[1]	Minke whale: 3 quick surfacings followed by 150 s dive[2]	Harbour porpoise: 1 surfacing every 60 s[3]	Sperm whale: 40 blows then 40 min dive[4]	Beaked whale: 4 quick surfacings followed by 20 min dive[5]
g(0) at 2.5 ms^{-1} (5 kts)	0.95	0.77	0.59	0.44	0.16
g(0) at 5 ms^{-1} (10 kts)	0.75	0.50	0.34	0.23	0.08
Ratio of g(0) at 5 kts to g(0) at 10 kts	1.27	1.54	1.71	1.94	2.03
A	1.30	1.12	1.03	1.04	1.00
B	-8.16	-4.00	-2.29	-1.48	-0.43

[1] Examples of blue whale diving behaviour off California in Croll et al. (2001) and off Sri Lanka in De Vos et al. (2011)
[2] Review of data on surfacing rates for minke whales in the North Atlantic in Øien et al. (2009)
[3] Observations of diving behaviour of harbour porpoise in Teilmann et al. (2007)
[4] Data on sperm whale respiration rates in Gordon & Steiner (1992)
[5] Examples of beaked whale diving data in Barlow & Gisiner (2006) and Baird et al. (2006)

Table 4. *Blue whale case study:* Simulation parameters were selected to give a strip width of 1,500 m at vessel survey speed of 10 kts with two observers. This resulted in a g(0) estimate of 0.85 from the simulation (top row). Estimates of M_e were then made for vessel speeds of 5 kts with a mitigation shutdown distance of 1,000 m and whale swim speeds of 0 to 2 ms^{-1}. Propagation loss was assumed to be 20log(r).

Vessel speed (ms^{-1})	Whale speed (ms^{-1})	Dive duration (s)	No. obs.	Number of surfacings in each sequence	Shut down distance (m)	g(0)	eshw (m)	M_e -20 dB	M_e -30 dB	M_e -40 dB
5.0	0	300	2	5	--	0.85	1,500	--	--	--
2.5	0	300	2	5	1,000	0.96	1,600	0.96	0.96	0.59
2.5	1	300	2	5	1,000	--	--	0.94	0.93	0.60
2.5	2	300	2	5	1,000	--	--	0.91	0.87	0.56
2.5	2	300	1	5	1,000	0.80	1,300	0.80	0.76	0.43
2.5	0	600	2	9	1,000	0.91	1,600	0.91	0.90	0.53
2.5	1	600	2	9	1,000	--	--	0.86	0.85	0.51
2.5	2	600	2	9	1,000	--	--	0.78	0.72	0.45

SEL, but mitigation can be less effective because of whale movement (Figure 3). The influence of whale speed is also illustrated in Figure 4. In this case, for moderate exposure levels around -35 dB, assumptions of slower whale travel speeds result in higher M_e with just one observer compared to faster travel speeds with two observers.

The variability in M_e in Figure 5 at high values of SEL reflects the small sample sizes within the simulation of whales subject to these received levels. Even for this large, conspicuous species and the assumption of greatest propagation loss (spherical spreading), the estimates of M_e drop sharply for levels less than -40 dB. This is mainly influenced by the probability of cue detection at greater distances, which is not changed much by number of observers. However, adding an additional observer does increase M_e for higher noise levels by 10 to 15%.

Table 5 gives some examples of simulation runs to estimate M_e based on the scenarios in Table 3 for other species and 15log(r) spreading loss. These are provided only to illustrate a range of different outcomes and are not an alternative to a full set of trials for a particular mitigation option. For inconspicuous species such as the harbour porpoise, M_e, even for levels as high as -20 dB relative to a single pulse, is less than 50%. This

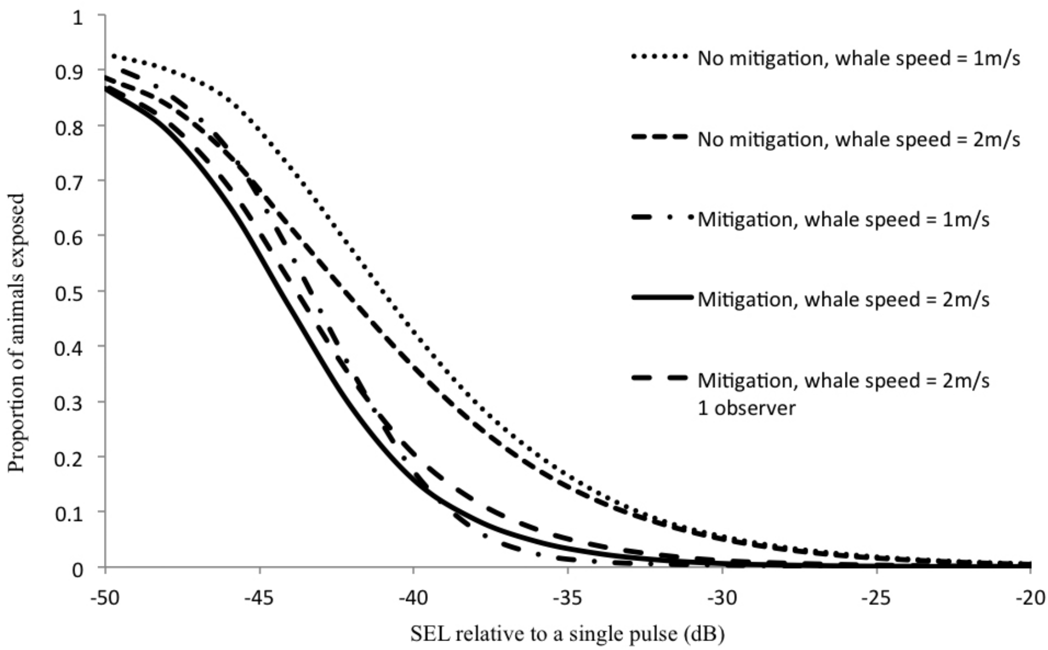

Figure 3. Proportions of animals within 6 km of the trackline affected by SEL over a certain level with and without mitigation for blue whale case study with eshw ~1.8 km, long dive time (D = 300 s), cues for each surfacing (C = 5), and 1 or 2 observers. g(0) from simulation = 0.94; ship speed = 2.5 ms^{-1}. Propagation loss assumed to be 20log(r).

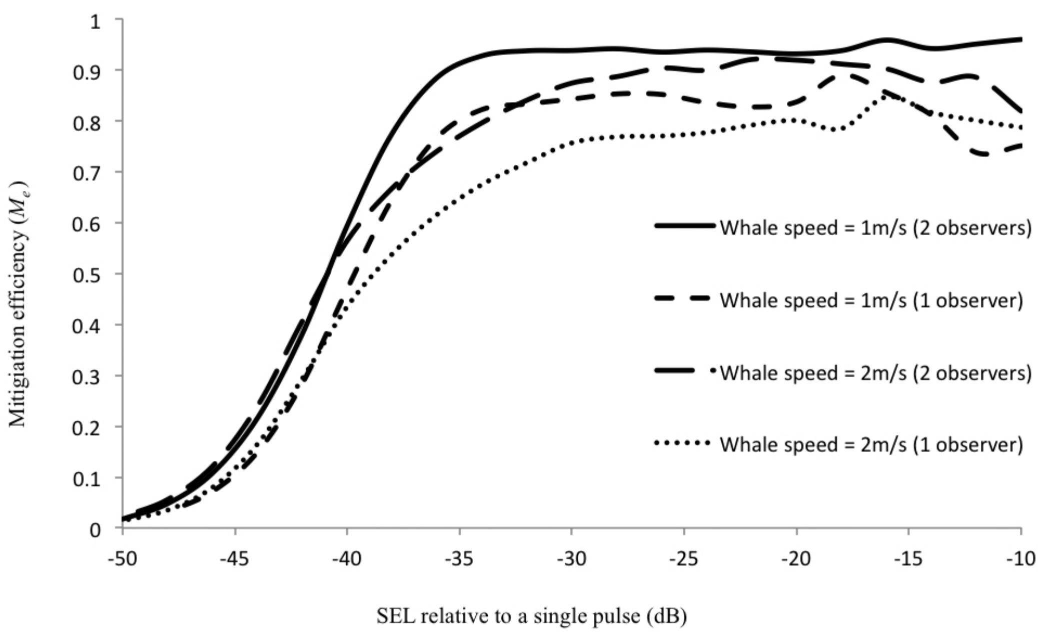

Figure 4. Mitigation efficiency for blue whale case study. Parameters as for Figure 3.

Table 5. Estimates of M_e for scenarios with parameters from Table 3 and 15log(r) propagation loss

Scenario	M_e -20 dB
Blue whale, whale speed = 0	0.93
Blue whale, whale speed = 1 ms^{-1}	0.92
Blue whale, whale speed = 2 ms^{-1}	0.84
Single harbour porpoise	0.43
Aggregation of 10 harbour porpoises (surfacing independently)	0.68
Single beaked whale	0.11
Aggregation of 10 beaked whales (surfacing independently)	0.26

will rapidly drop to very low values if mitigation is required in higher sea states than Beaufort sea state 2, which is the maximum sea state for which survey data on detection probability were available. However, if animals do occur in aggregations (see Skov & Thomsen, 2008) and shutdowns occur when any individual is sighted, then M_e can be substantially greater.

Discussion

These results essentially put an upper bound on M_e for good conditions, assuming experienced, fully alert observers and an instant shutdown response. The difficulties of practical implementation of mitigation procedures discussed in Weir & Dolman (2007) and Compton et al. (2008) will inevitably result in lower M_e. In addition, seismic surveys continued at night will generally reduce M_e by around 50% depending on daylight hours. Poor sightings conditions due to wind (sea state) or poor visibility will further reduce this but are difficult to quantify. Barlow & Gisiner (2006) estimated that when weather and daylight considerations were taken into account, mitigation monitoring would detect fewer than 2% of beaked whales that were directly in the path of the ship. The results of this simulation for similar situations with difficult-to-see species (Table 5) are consistent with Barlow and Gisiner's findings. Sightings rates for inconspicuous species drop very rapidly with increasing sea state. For example, for harbour porpoise sightings, rates fall substantially between Beaufort sea states 1 and 2 (Teilmann, 2003; SmartWind, 2013). Teilmann (2003) found that sighting rates for harbour porpoise in Beaufort sea states 2 and 3 were only 11% that of Beaufort sea states 0 and 1. For a large dataset in the North Sea, SmartWind (2013) found the equivalent ratio was 15%; however, in 50,000 km of surveys that were only conducted in Beaufort sea states 4 or less, sea states 0 and 1 were only encountered for 14% of the time.

Some of the most detailed evaluations of mitigation efficiency have been undertaken by the Western Gray Whale Advisory Panel (WGWAP) (2007) with respect to the effects of seismic surveys on western gray whales (*Eschrichtius robustus*) on their feeding grounds off Sakhalin Island. Mitigation efficiency was evaluated for criteria of a shutdown in response to a whale sighted within 1.5 km, or on a course to come closer than 1.5 km from the airgun array. For cumulative exposure levels in the range 195 to 215 dB_{SEL}, mitigation was estimated to reduce the expected number of such exposure cases by 44 to 99%. However, the mitigation efficiency dropped to 2 to 10% for exposure levels of 180 dB_{SEL}.

For conspicuous species in good sighting conditions, use of MMOs may provide a useful level of risk reduction against injuries from exposures to SELs greater than -20 to -30 dB relative to a single pulse (Table 4). However, if propagation loss was lower (e.g., 15log[r]), then use of MMOs may no longer be effective at these sound levels (Table 5). The main conclusion, therefore, is that simulations of the effectiveness of mitigation measures need to be performed on a case-by-case basis using input data that correspond as closely as possible to the scenario being investigated. This study considered exposure in terms of SEL. In some cases, the greatest concern may derive from the maximum received sound pressure level (SPL). Evaluating the effectiveness of mitigation measures to reduce risks of received levels exceeding a given SPL would require a different approach. There will be many cases for which using visual observers results in only a very small risk reduction, and these situations are not always immediately apparent. In such situations, alternatives to mitigation based on visual observers may be required. Passive Acoustic Monitoring (PAM) using towed hydrophones is increasingly used in conjunction with MMOs as a way of detecting animals. It may be possible to simulate the mitigation efficiency associated with PAM in a similar way to using MMOs. However, there are currently limited data on the probability of detection associated with PAM compared to data on the visual detection process. Acoustic detection probability is also strongly influenced by noise levels and vocal behaviour (Leaper et al., 2001).

Without an adequate quantified assessment of the risk reduction, mitigation measures may often be applied inappropriately or result in regulators granting approval for activities on the basis of measures that do little to reduce risk. The simulation framework used here is simple but does give a useful indication of the likely risk reduction that may be achieved. Estimates of g(0) and eshw along with predictions of propagation loss do provide a useful first indication of whether it is worth considering using visual observers for mitigation purposes. Estimates of g(0) provide an upper bound on M_e if adjusted for vessel speed. For safety zones with a radius of less than around half eshw, M_e may be around 90% of g(0). The overall estimates of M_e are sensitive to all the variables that affect eshw and g(0), and estimates of these for surveys of similar species in similar areas can vary considerably. Although seismic activity is used as the main case study here, the same approach could be applied to sonar use or pile driving. However, overall results are most influenced by assumptions about propagation loss. Many Environmental Impact Assessments do include detailed predictions of propagation and so this information is often available for input into a simulation model, including the visual detection process (Parnum & Duncan, 2012; SmartWind, 2013).

In addition to whether the risk reduction achieved by shutdowns is a useful contribution to mitigation, it is also necessary to consider the impacts of additional noise input due to shutdowns. Completing a seismic survey that has been interrupted by a shutdown will inevitably require additional source deployments to resurvey a section of a line. In some circumstances, the whole survey line needs to be repeated following a shutdown. There may also be a requirement for a soft start following the interruption. The resulting additional noise needs to be taken into account in any assessment of the risk reduction that is likely to be achieved, including implications for disturbance and behavioural responses. However, unless the likely actions needed to complete the seismic survey following a shutdown are specified, it is difficult to predict how many additional pulses are likely to be generated.

The proportion of whales that would have been at risk of injury without mitigation and remain at risk of injury with mitigation in place will be $(1-M_e)$. So, if N whales were at risk of injury without mitigation, then $N(1-M_e)$ will be at risk with mitigation. If the mitigation actions increase the total length of seismic lines surveyed by a factor of Q, then $NQ\,(1-M_e)$ whales will be at risk assuming whale density within the survey region has not been influenced by the survey. In areas with more than one cetacean species, shutdowns may occur for the more visible species, but the additional noise generated will also impact the less visible species. In these circumstances, it is possible that $Q\,(1-M_e) > 1$ will result in greater risk for some species as a result of the mitigation measures.

In conclusion, quantifying the effectiveness of proposed mitigation measures is necessary to allow regulators to make informed decisions on whether to grant a licence to allow an activity to take place and to select mitigation options that most effectively reduce risk. In particular, the risk reduction associated with technologies that allow for reduced source levels can be compared with current mitigation practices. Based on the results of this study, there will be very few instances where mitigation using visual observers can achieve a greater risk reduction than would be achieved by a 3 dB reduction in source level throughout the survey.

Acknowledgment

Funding for this study was provided by the International Fund for Animal Welfare.

Literature Cited

Baird, R. W., Webster, D. L., McSweeney, D. J., Ligon, A. D., Schorr, G. S., & Barlow, J. (2006). Diving behaviour of Cuvier's (*Ziphius cavirostris*) and Blainville's (*Mesoplodon densirostris*) beaked whales in Hawaii. *Canadian Journal of Zoology*, *84*, 1120-1128. http://dx.doi.org/10.1139/z06-095

Barlow, J., & Gisiner, R. (2006). Mitigating, monitoring and assessing the effects of anthropogenic sound on beaked whales. *Journal of Cetacean Research and Management*, *7*(3), 239-249.

Barlow, J., & Taylor, B. L. (2005). Estimates of sperm whale abundance in the northeastern temperate Pacific from a combined acoustic and visual survey. *Marine Mammal Science*, *21*, 429-445. http://dx.doi.org/10.1111/j.1748-7692.2005.tb01242.x

Branch, T. A., & Butterworth, D. S. (2001). Estimates of abundance south of 60° S for cetacean species sighted frequently on the 1978/79 to 1997/98 IWC/IDCR-SOWER sighting surveys. *Journal of Cetacean Research and Management*, *3*(3), 251-270.

Buckland, S. T., Anderson, D. R., Burnham, K. P., & Laake, J. L. (1993). *Distance sampling: Estimating abundance of biological populations*. New York: Chapman and Hall. xii + 446 pp. http://dx.doi.org/10.1007/978-94-011-1572-8; http://dx.doi.org/10.1007/978-94-011-1574-2

Calambokidis, J., & Barlow, J. (2004). Abundance of blue and humpback whales in the eastern North Pacific estimated by capture-recapture and line-transect methods. *Marine Mammal Science*, *20*(1), 63-85. http://dx.doi.org/10.1111/j.1748-7692.2004.tb01141.x

Compton, R., Goodwin, L., Handy, R., & Abbott, V. (2008). A critical examination of worldwide guidelines for minimising the disturbance to marine mammals during seismic surveys. *Marine Policy*, *32*, 255-262. http://dx.doi.org/10.1016/j.marpol.2007.05.005

Cooke, J. G., & Leaper, R. (1998). *Results of simulation studies of a hazard probability approach to estimation of whale density and effective strip width* (Paper SC/50/RMP20). Presented to the IWC Scientific Committee. (Unpublished).

Croll, D. A., Acevedo-Gutierrez, A., Tershy, B. R., & Urban-Ramírez, J. (2001). The diving behavior of blue and fin whales: Is dive duration shorter than expected based on oxygen stores? *Comparative Biochemistry and Physiology Part A*, *129*, 797-809. http://dx.doi.org/10.1016/S1095-6433(01)00348-8

De Vos, A., Christiansen, F., Pattiaratchi, C., & Harcourt, R. (2011). *Submergence times and abundance estimation of blue whales off Sri Lanka* (Paper SC/63/WW7). Presented to the IWC Scientific Committee, Tromsø, Norway. (Unpublished). 4 pp.

Department of the Environment, Water, Heritage, and the Arts (DEWHA) (Ed.). (2008). *EPBC Act policy statement 2.1 – Interaction between offshore seismic exploration and whales*. Canberra, Australia: DEWHA.

Galletti Vernazzani, B. (2012). *Abundance estimates of Chilean blue whales by mark-recapture and line transect techniques* (Paper SC/64/SH19). Presented to the IWC Scientific Committee, Panama City, Panama. (Unpublished). 11 pp.

Gordon, J., & Steiner, L. (1992). Ventilation and dive patterns in sperm whales, *Physeter macrocephalus*, in the Azores. *Reports of the International Whaling Commission*, *42*, 561-565.

Hammond, P. S., Berggren, P., Benke, H., Borchers, D. L., Collet, A., Heide-Jørgensen, M. P., . . . Øien, N. (2002). Abundance of harbour porpoise and other cetaceans in the North Sea and adjacent waters. *Journal of Applied Ecology*, *39*, 361-376. http://dx.doi.org/10.1046/j.1365-2664.2002.00713.x

Hammond, P. S., Macleod, K., Berggren, P., Borchers, D. L., Burt, L., Cañadas, A., . . . Vázquez, J. A. (2013). Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. *Biological Conservation*, *164*, 107-122. http://dx.doi.org/10.1016/j.biocon.2013.04.010

Heide-Jørgensen, M. P., Simon, M. J., & Laidre, K. L. (2007). Estimates of large whale abundance in Greenlandic waters from a ship-based survey in 2005. *Journal of Cetacean Research and Management*, *9*(2), 95-104.

Hiby, A. R. (1982). The effect of random whale movement on density estimates obtained from whale sighting surveys. *Reports of the International Whaling Commission*, *32*, 791-793.

Hildebrand, J. A. (2009). Anthropogenic and natural sources of ambient noise in the ocean. *Marine Ecology Progress Series*, *395*, 5-20. http://dx.doi.org/10.3354/meps08353

Joint Nature Conservation Committee. (2010). *JNCC guidelines for minimising the risk of injury and disturbance to marine mammals from seismic surveys*. Aberdeen, UK: JNCC.

Kasamatsu, F., & Joyce, G. G. (1995). Current status of odontocetes in the Antarctic. *Antarctic Science*, *7*(4), 365-379. http://dx.doi.org/10.1017/S0954102095000514

Leaper, R., Gillespie, D., & Papastavrou, V. (2001). Results of passive acoustic surveys for odontocetes in the Southern Ocean. *Journal of Cetacean Research and Management*, *2*(3), 187-196.

Leaper, R., Burt, L., Gillespie, D., & MacLeod, K. (2010). Comparisons of measured and estimated distances and angles from sighting surveys. *Journal of Cetacean Research and Management*, *11*(3), 229-238.

Mori, M., Butterworth, D. S., Brandao, A., Rademeyer, R. A., Okamura, H., & Matsuda, H. (2003). Observer experience and Antarctic minke whale sighting ability in IWC/IDCR-SOWER surveys. *Journal of Cetacean Research and Management*, *5*(1), 1-11.

New Zealand Department of Conservation (DOC). (2013). *Code of conduct for minimising acoustic disturbance to marine mammals*. Wellington, New Zealand: DOC.

Øien, N., Bøthun, G., & Kleivane, L. (2009). *Summary of available data on northeastern Atlantic minke whale surfacing rates* (Paper SC/61/RMP7). Presented to the IWC Scientific Committee, Madeira, Portugal. (Unpublished). 6 pp.

Okamura, H., & Kitakado, T. (2007). *Abundance estimates of Southern Hemisphere minke whales from the IDCR-SOWER surveys using a hazard probability model* (Paper SC/59/IA14). Presented to the IWC Scientific Committee, Anchorage, AK. (Unpublished). 29 pp.

Okamura, H., Miyashita, T., & Kitakado, T. (2009). *Revised estimate of g(0) for the North Pacific minke whale* (Paper SC/61/NPM5). Presented to the IWC Scientific Committee, Madeira, Portugal. (Unpublished). 7 pp.

Parnum, I., & Duncan, A. (2012). *Sound exposure modelling for the bight 3D seismic survey in the eastern Great Australian Bight, South Australia* (Report C2012-36). Prepared by CMST. Retrieved 22 July 2015 from www.bightpetroleum.com.

Parsons, E. C. M., Dolman, S. J., Jasny, M., Rose, N. A., Simmonds, M. P., & Wright, A. J. (2009). A critique of the UK's JNCC seismic survey guidelines for minimising acoustic disturbance to marine mammals: Best practise? *Marine Pollution Bulletin*, *58*, 643-651. http://dx.doi.org/10.1016/j.marpolbul.2009.02.024

Pike, D. G., Víkingsson, G. A., & Gunnlaugsson, T. (2004). *Abundance estimates for blue whales (*Balaenoptera musculus*) in Icelandic and adjacent waters* (Paper SC/56/O6). Presented to the IWC Scientific Committee, Sorrento, Italy. (Unpublished). 10 pp.

Priyadarshana, T., Randage, R., Alling, A., Calderan, S., Gordon, J., Leaper, R., & Porter, L. (2014). *Preliminary results of surveys to investigate overlap between* *shipping and blue whale distribution off southern Sri Lanka* (Paper SC/65b/HIM06). Presented to the IWC Scientific Committee, Bled, Slovenia. 11 pp.

Schweder, T., Skaug, H. J., Dimakos, X. K., Langaas, M., & Øien, N. (1997). Abundance of northeastern Atlantic minke whales, estimates for 1989 and 1995. *Reports of the International Whaling Commission*, *47*, 453-484.

Skov, H., & Thomsen, F. (2008). Resolving fine-scale spatio-temporal dynamics in the harbour porpoise *Phocoena phocoena*. *Marine Ecology Progress Series*, *373*, 173-186. http://dx.doi.org/10.3354/meps07666

SmartWind. (2013). *Hornsea offshore wind farm project one: Environmental statement: Volume 5. Offshore annexes* (Annex 5.4.1, Marine Mammal Technical Report). London: SMart Wind Limited.

Southall, B., Bowles, A. E., Ellison, W. T., Finneran, J. J., Gentry, R. L., Greene, C. R., Jr., . . . Tyack, P. (2007). Marine mammal noise exposure criteria: Initial scientific recommendations. *Aquatic Mammals*, *33*(4), 411-509. http://dx.doi.org/10.1578/AM.33.4.2007.411

Starck, J., Toppila, E., & Pyyko, I. (2003). Impulse noise and risk criteria. *Noise Health*, *5*(20), 63-73. http://dx.doi.org/10.4103/1463-1741.56217

Teilmann, J. (2003). Influence of sea-state on density estimates of harbour porpoises (*Phocoena phocoena*). *Journal of Cetacean Research and Management*, *5*(1), 85-92.

Teilmann, J., Larsen, F., & Desportes, G. (2007). Time allocation and diving behaviour of harbour porpoises (*Phocoena phocoena*) in Danish and adjacent waters. *Journal of Cetacean and Research Management*, *9*(3), 201-210.

Víkingsson, G. A., Pike, D. G., Desportes, G., Øien, N., & Gunnlaugsson, T. (2009). Distribution and abundance of fin whales (*Balaenoptera physalus*) in the Northeast and Central Atlantic as inferred from the North Atlantic Sightings Surveys 1987-2001. *NAMMCO Scientific Publications*, *7*, 49-72. http://dx.doi.org/10.7557/3.2705

Weir, C. R., & Dolman, S. J. (2007). Comparative review of the regional marine mammal mitigation guidelines implemented during industrial seismic surveys, and guidance towards a worldwide standard. *Journal of International Wildlife Law & Policy*, *10*, 1-27. http://dx.doi.org/10.1080/13880290701229838

Western Gray Whale Advisory Panel (WGWAP). (2007). *Report of the seismic survey task force*. Retrieved 22 July 2015 from http://cmsdata.iucn.org/downloads/seismic_tf___report_final_20_09_07_with_caution_20_05_08_1.pdf.