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Responses of cetaceans to anthropogenic noise

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ABSTRACT

1. Since the last thorough review of the effects of anthropogenic noise on cetaceans in 1995, a substantial number of research reports has been published and our ability to document response(s), or the lack thereof, has improved. While rigorous measurement of responses remains important, there is an increased need to interpret observed actions in the context of population-level consequences and acceptable exposure levels. There has been little change in the sources of noise, with the notable addition of noise from wind farms and novel acoustic deterrent and harassment devices (ADDs/AHDs). Overall, the noise sources of primary concern are ships, seismic exploration, sonars of all types and some AHDs.

2. Responses to noise fall into three main categories: behavioural, acoustic and physiological. We reviewed reports of the first two exhaustively, reviewing all peer-reviewed literature since 1995 with exceptions only for emerging subjects. Furthermore, we fully review only those studies for which received sound characteristics (amplitude and frequency) are reported, because interpreting what elicits responses or lack of responses is impossible without this exposure information. Behavioural responses include changes in surfacing, diving and heading patterns. Acoustic responses include changes in type or timing of vocalizations relative to the noise source. For physiological responses we address the issues of auditory threshold shifts and ‘stress’, albeit in a more limited capacity; a thorough review of physiological consequences is beyond the scope of this paper.

3. Overall, we found significant progress in the documentation of responses of cetaceans to various noise sources. However, we are concerned about the lack of investigation into the potential effects of prevalent noise sources such as commercial sonars, depth finders and fisheries acoustics gear. Furthermore, we were surprised at the number of experiments that failed to report any information about the sound exposure experienced by their experimental subjects. Conducting experiments with cetaceans is challenging and opportunities are limited, so use of the latter should be maximized and include rigorous measurements and or modelling of exposure.

Keywords: acoustic deterrent, acoustic harassment, behaviour, device, exposure, hearing, man-made, marine mammals, sound

INTRODUCTION

Richardson *et al.* (1995) provided a thorough and useful review of the contemporary knowledge of the sources and effects of anthropogenic noise on marine mammals. Since that time, a proliferation of research and improved abilities to study marine mammal behaviour has added significant information to our understanding of this issue. While documented disturbance reactions of individual animals are becoming more common in the literature, we still lack any understanding of whether or how anthropogenic noise may affect populations of marine mammals (NRC, 2005). The assessment of population-level effects requires linking noise exposure with some change in abundance or demographic parameters (e.g. birth rate or survival). Inherent in such an assessment is a need for controlled experiments linking measured sound levels to responses by individual animals (Tyack, Gordon & Thompson, 2004).

This review provides an update on the documented behavioural, acoustic and some physiological responses of cetaceans to anthropogenic noise. While most of the response data in the literature focuses on motor behaviour, we also review several studies that have quantified acoustic and/or physiological responses to sound, such as type and timing of vocalizations, hearing threshold shifts and effects on heart rate. Studies of terrestrial animals are helpful in this regard, though we use them primarily to illustrate what can and should be done with cetaceans. We focus in detail on those studies that report quantitatively on both the sound field as well as some indicator of response, i.e. received sound characteristics associated with behavioural or physiological response(s). Studies that did not report sound exposure at the location(s) of the animal(s) are not specifically reviewed, because results are almost impossible to interpret; i.e. measured or modelled characteristics of the received sounds ('received level' or RL) are critical to the interpretation of animal(s) responses or lack thereof. Furthermore, our review is limited to anthropogenic sources of sound, and we considered only work reported in the peer-reviewed literature, except in rare cases where we think it important to pursue a new or pressing topic, e.g. effects of sonar on beaked whales. As a result of these criteria, our review is dominated by response of cetaceans to acoustic deterrent and harassment devices (ADDs and AHDs) where significant recent research has occurred and RLs are much more consistently reported.

Dynamics and measurement of sound in the ocean: a short primer

Throughout this review we use the terms 'noise' and 'sound' interchangeably with respect to the introduction of acoustic energy by anthropogenic sources. Some of this acoustic energy is unwanted or unintentional (e.g. vessel propulsion), while some sound is produced with a specific purpose, such as sonar or seismic exploration. Here, we do not aim to assess the intentions of producers of anthropogenic sound, only to review the reports of the response(s) to them by cetaceans. With regard to sound levels, we report measures used by the original authors. Sound pressure magnitude in water is normally described by sound pressure on a decibel (dB) scale relative to a reference root mean square (rms) pressure of 1 μ Pa (micro-Pascal) (i.e. dB re 1 μ Pa), but there are other measures reported, for example peak-to-peak or 0-peak; the latter two referring to the measurement of amplitude using the peaks and troughs of sound energy as measured in volts. Often the total energy to which an animal is exposed is of interest, and energy flux density is normally the unit for this measurement. Use of an inappropriate metric can lead to very different results. For example, results for one metric (e.g. peak-to-peak) compared with another (e.g. rms) can vary significantly depending on the duration of the signal. We refer readers to two recent papers (Madsen, 2005; Madsen *et al.*,

2006a) for a thorough discussion of sound pressure measurements, specifically with reference to signal type and marine mammal hearing.

We have categorized our review by noise source and then by taxon, with examples from non-cetacean species providing insight where appropriate. The sound sources reviewed by Richardson *et al.* (1995) still represent the primary noise contributors, and we considered these sources together with some that have become more prevalent in the last decade, e.g. wind farm construction and operation. Again, we have reviewed only those studies that reported, measured or modelled RL along with data on response(s) or lack thereof. Following this criterion severely limited the number of studies we formally reviewed because many studies reported response data without any information about the characteristics or levels of sound received by the animals. We have, however, compiled a reference list that includes all of the studies we considered (see Appendix S1, Supplementary material).

Sound is an important sensory modality for many marine animals, mammals included. In the opaque and viscous marine environment, other sensory modalities such as vision, touch, smell or taste have severe limitations in effective range and speed of signal transmission. Among marine mammals, the cetaceans (whales and dolphins) utilize a wide band of acoustic frequencies. The massive blue whale *Balaenoptera musculus* produces low-frequency sounds down to ~15 Hz, and on the other end of the spectrum several species of porpoises (e.g. harbour porpoises – *Phocoena phocoena*) emit echolocation signals at 120–150 kHz. This broad range of frequencies intersects with many of the sounds humans introduce into the water, including ship noise, sonars of various types and seismic exploration signals. Some sounds produced by humans are well above the range used by marine mammals, e.g. high-frequency echo sounders. It is in the shared frequency range that we are concerned about the effects that anthropogenic sound may have on cetaceans.

In the ocean, acoustic energy propagates efficiently, travelling fast and potentially over great distances. Sound travels almost five times faster through sea water than through air, and low frequencies can travel hundreds of kilometres with little loss in energy (Urlick, 1983). Sound propagation can be affected by many factors, the most influential of which are: (i) frequency of the sound; (ii) water depth; and (iii) density differences within the water column, which vary primarily with temperature and pressure (Urlick, 1983). So, the sound arriving at an animal is subject to propagation conditions that can be quite complex, which can in turn significantly affect the characteristics of arriving sound energy. This variability is one of the reasons that we decided not to thoroughly review those studies that did not measure or model the RL.

There are at least four primary concerns for animals exposed to elevated noise levels: permanent threshold shifts (PTS), temporary threshold shifts (TTS), acoustic masking and behavioural disturbance. The hearing threshold is the amplitude necessary for detection, and the threshold varies depending on frequency across the hearing range of a given individual. Both PTS and TTS represent actual changes in the ability of an animal to hear, usually at a particular frequency, whereby it is less sensitive at one or more frequencies as a result of exposure to sound. Masking can occur when an extraneous sound covers or ‘masks’ a desired signal, making the latter more difficult to detect. Finally, behavioural responses are a demonstrable change in the activity of an animal in response to a sound. These effects can be difficult to detect due to the cryptic and variable nature of cetacean behaviour, but they can indeed be demonstrated with careful data collection. Examples of behavioural effects include the abandonment of an important activity (e.g. feeding, nursing) or location in response to some sound, and the repeated abandonment of such vital activities can lead to detrimental consequences for the animal(s) affected.

SOURCES OF NOISE

Aircraft

Analysis of exposure of marine mammals to aircraft noise is complicated because the sound is generated in air and can travel over considerable distances before entering the water. Angle is also important: at angles >13 degrees from the vertical, most sound is reflected and does not penetrate the water, except in rough seas (see Richardson *et al.*, 1995, p. 13). These factors and animal-generated variables, such as whale depth, make it difficult to estimate exposure at whales when they are submerged.

Only one study since Richardson *et al.* (1995) has documented responses to aircraft. Patenaude *et al.* (2002) observed bowhead whales *Balaena mysticetus* during spring migration and recorded their short-term responses to both helicopter and fixed-wing aircraft activity. They included an analysis of the noise from the two aircraft recorded at 3 and 18 m depth. The helicopter (Bell 212) was 7–17.5 dB noisier than the fixed-wing aircraft (De Havilland Twin Otter) with a peak RL of ~126 dB re 1 μ Pa, and helicopter noise consisted primarily of tones from the main and tail rotors. The Twin Otter noise spectrum contained fewer prominent tones than the helicopter and reached levels of 120 dB re 1 μ Pa. The sound levels were predictably and inversely related to aircraft altitude, with measurements made for the helicopter at 110, 150, 300, 450 and 650 m and for the fixed-wing aircraft at 150 and 450 m. The most common responses the authors observed included short surfacing durations, abrupt dives and percussive behaviour (e.g. breach, tail slap). Of the 63 bowhead groups observed, 14% reacted to the helicopter, with the majority of the responses occurring when the helicopter was at altitudes of \leq 150 m and lateral distances of \leq 250 m. Interestingly, one whale also responded with a briefer than normal surfacing when the helicopter was stationary on the ice with its engines running, at a horizontal distance of 230 m. Such an event could indicate startle or could be the result of curiosity, it is impossible to know with one observation.

Patenaude *et al.* (2002) had a unique opportunity to observe a mother-calf pair exposed to three close (50–150 m lateral distance), low-altitude (15–30 m) helicopter passes and one more distant (500 m) pass. The mother dived every time it was at the surface during a pass, and the calf dived once. While the authors did report RL, these measurements were taken directly under the aircraft, and the reported levels were related to aircraft altitude. No effort was made to model RL laterally away from this track, though the authors reported ‘reaction’ and ‘no reaction’ out to 1600 m lateral displacement from the aircraft trackline. The authors reported no statistical difference in reaction at \leq 250 vs. >250 m lateral distance, though the sample sizes were small in the >250 m category. Additionally, the authors did not elaborate on the choice of these range categories. Without more information about the received sound characteristics at the locations of whales that reacted vs. those that did not, it is difficult to interpret these results in a larger context.

Whale-watching

Erbe (2002) used an acoustic exposure model developed by Erbe & Farmer (2000a) to investigate the acoustic effects of whale-watching vessels on killer whales *Orcinus orca* in southern British Columbia and northern Washington State. Erbe (2002) used a broadband 120 dB re 1 μ Pa rms level as the threshold necessary for a response, though this represents a significant extrapolation from the source on which this threshold is based, i.e. Richardson *et al.* (1995). A whale-watching zodiac with twin 150-hp engines at 51 km/hour in a sea state of 1–2 would (i) be audible to approximately 16 km; (ii) mask a chosen killer whale call to approximately 14 km; (iii) elicit a behavioural response at ranges of approximately 200 m

using the 120 dB responsiveness threshold; (iv) cause a TTS of 5 dB if a whale spent 30–50 minutes within 450 m of the boat; and (v) cause a PTS of 2–5 dB if the whale was within 1 km of the boat for 8 hours per day, 5 days a week for 50 years. These levels of exposure could occur based on whale-watching boat densities in the study area. A boat with twin 225-hp engines at a typical ‘circling speed’, 10 km/hour, would result in audibility and masking potential at approximately 1 km, a behavioural response at 50 m, and a TTS of 5 dB in 50 minutes of exposure at the 20 m range.

Williams *et al.* (2002) studied killer whale behaviour to investigate impacts of ‘leapfrogging’, a practice commonly used by whale watch operators to put themselves in the predicted path of killer whales without approaching whales closer than 100 m as outlined in whale-watching guidelines. Movement paths of killer whales were significantly less direct and less predictable during encounters with a leapfrogging vessel than during control periods where no approaches were made by boats. RLs were measured 100 m from the leapfrogging boat and peaked at ~115 dB at ~100 Hz, though the measure (e.g. rms, peak-peak) was not provided. Given this level and using simple spherical spreading loss, which is a first-order approximation of propagation loss (Urick, 1983), the authors conclude that leapfrogging vessels would have to be 500 m away from whales to produce the same RL as a boat travelling slowly and paralleling a whale’s path 100 m away. Williams *et al.* (2002) oversimplified the received sound characteristics by providing a single dB value for a ~5 octave band. Moreover, while they did document a response, the 116 dB level they cite as the threshold for response was likely dominated by energy at frequencies well below the whale’s best hearing range. The response they observed may have been triggered by lower amplitude levels at frequencies more audible to the whales.

Buckstaff (2004) examined changes in bottlenose dolphin *Tursiops truncatus* whistle behaviour in response to noise from recreational vessels. Mean whistle rate was significantly different during different stages of vessel approaches: whistle rate was higher at the onset of vessel noise than during exposure and was lowest after exposure. Boat noise was measured at the observation vessel, which maintained a distance of ~20 m from the focal dolphin. Buckstaff (2004) reported estimates of RL for instances when the focal dolphin was between the observation vessel and the approaching/passing boat, so the reported RLs represent a minimum estimate. RLs ranged from 115 to 138 dB re 1 μ Pa rms for planing boats, 114–121 dB re 1 μ Pa rms for plowing boats, and 113–116 dB re 1 μ Pa rms for idling boats (Table 1). Furthermore, Buckstaff (2004) reported that RLs for personal watercraft (‘jet skis’) were 9 dB quieter than a boat travelling at similar speed. This quieter underwater signature confirms that less noise is introduced into the marine environment; for animals that could use acoustic cues to avoid approaching boats, these quieter watercraft may actually represent a greater threat for collision than louder boats. The increase in whistle rate Buckstaff (2004) observed suggests an effort to decrease signal degradation due to acoustic disturbance (i.e. masking). Alternatively, the whistle increase may indicate that the dolphins heard the boat before they entered the zone being monitored, and this increase in whistle rate could actually mediate the behavioural responses reported by Nowacek, Wells & Solow (2001), who found that dolphins took longer dives, decreased interanimal distance, changed heading and increased speed in response to approaching boats. Thus, the changes in group spread could be mediated by the change in whistle rate reported by Buckstaff (2004).

Au & Green (2000) investigated RLs of vessel noise from whale-watching boats in Hawaii and their effects on humpback whale *Megaptera novaeangliae* behaviour. They provided detailed noise signatures of five different vessels. Behavioural responses reported were similar to previous reports for humpbacks in Hawaiian waters (e.g. abrupt course changes,

Table 1. Summary of non-ADD/AHD studies

Study	Field, laboratory or modelling	Species	Sound source	Signal characteristics	RLs (dB re 1 uPa) modelled values in italics	Deployment details	General results
Patenaude <i>et al.</i> (2002)	Field	<i>Balaena mysticetus</i>	Helicopter (Bell 212)	Mainly tones between 10 and 500 Hz; aircraft at 150 m and 450 m altitude	120 and <i>110</i> at 3 m depth 112 and <i>107</i> at 18 m depth	Sound levels inversely related to altitude	Short dive durations, abrupt dives, orienting away from noise
Erbe (2002)	Modelling	<i>Orcinus orca</i>	Twin Otter (De Havilland Twin Otter) Zodiac whale watch boats (twin 150 hp engines)	Fewer prominent tones than for helicopter; aircraft at 150 m and <i>450 m</i> , at 150 kts Broadband	120 and <i>113</i> at 3 m depth 120 and <i>110</i> at 18 m depth <i>120</i> *		Audible, masking, behavioural, TTS (5 dB after 30–50-minute exposure) and PTS (2–5 dB, 8-hour daily exposure zones, 5 days/week for 50 years) zones at 1600, 1400, 200, 450, 1000 m, respectively
Williams <i>et al.</i> (2002)	Field	<i>Orcinus orca</i>	'Leapfrogging' whale watch boats	100 Hz, boats > 100 m from whales	115	RL measured at 100 m	Movement paths less direct and less predictable
Buckstaff (2004)	Field	<i>Tursiops truncatus</i>	Recreational boats		115–138 (planing boats); 114–121 (plowing boats); and 113–116 (idling boats)	Boats maintained 20 m from focal dolphin	Higher whistle rate at onset of noise than during or after exposure
Erbe & Farmer (2000b)	Modelling	<i>Delphinapterus leucas</i>	Ice-breaking boat	1/3 octave band level centred at 5 kHz	<i>81 (disturbance threshold)</i>		Audible and masking zones at 35–78 km, 14 (bubblers)/40 (ramming) km, respectively; TTS of 12–8 dB within 40 m of bubbler/120 m of ramming for >30 minutes, or of 4.8 dB within 1–2 km of bubbler/2–4 km of ramming 20 minutes

Madsen & Møhl (2000)	Field	<i>Physeter macrocephalus</i>	Detonations		<179	No behavioural or acoustic effects
Todd <i>et al.</i> (1996)	Field	<i>Megaptera novaeangliae</i>	Detonations	400 Hz	140–153*	No detectable changes in respiration rates, or surface reactions, or differences in re-sighting rates
Finneran <i>et al.</i> (2000)	Laboratory	<i>Tursiops truncatus</i>	Simulations of distant underwater explosions		196/209 (disturbance threshold)**	Behavioural alterations at these RLs; no TTS >6 dB up to 221 dB re 1 µPa p-p
Finneran <i>et al.</i> (2000)	Laboratory	<i>Delphinapterus leucas</i>	Simulations of distant underwater explosions		220 (disturbance threshold)**	Behavioural alterations at this RL; no TTS >6 dB up to 221 dB re 1 µPa p-p
McCaulley <i>et al.</i> (2000)	Field	<i>Megaptera novaeangliae</i>	Commercial (C) and experimental (E) seismic arrays		258 (C) and 227 (E)**	Avoidance response at 160–170 dB re 1 µPa p-p for both C and E arrays
Goold (1996)	Field	<i>Delphinus delphis</i>	Seismic survey air guns	a. 250 Hz, b. 2 kHz, c. 10 kHz, d. 20 kHz	a. 170, b. 140, c. 115, d. 90***	Greater number of vocalizations per hour before than during seismic surveys
Goold & Fish (1998)	Field	<i>Delphinus capensis</i>	Seismic survey air guns	a. 200 Hz, b. 20 kHz	a. 140, b. 90***	Smaller proportion of acoustic contact during emissions (4%) then when air guns were not in use
Madsen <i>et al.</i> (2002)	Field	<i>Physeter macrocephalus</i>	Seismic survey vessels	210–260 Hz	146 **	No behavioural or acoustic responses observed
Miller <i>et al.</i> (2000)	Field	<i>Megaptera novaeangliae</i>	SURTASS-LFA sonar	130–160 Hz ('low' component) and 260–320 Hz ('high' component)	130–150*	Song length significantly longer during playbacks than before or after
Fristrup <i>et al.</i> (2003)	Field	<i>Megaptera novaeangliae</i>	SURTASS-LFA sonar	130–160 Hz ('low' component) and 260–320 Hz ('high' component)	130–150*	Songs longer if overlapped by pings; effects up to 2 hours after pings
						Detonations at 1.8 km
						Arrays of 2678 (C) and 20 cu. (E)
						80–100 m depth, 5 km from source
						750 m from source
						18 transducer array towed at 60–180 m depth
						18 transducer array towed at 60–180 m depth

Table 1. (Continued)

Study	Field, laboratory or modelling	Species	Sound source	Signal characteristics	RLs (dB re 1 μ Pa) modelled values in italics	Deployment details	General results
Croll <i>et al.</i> (2001)	Field	<i>Balaena musculus</i> and <i>B. physalus</i>	SURTASS-LFA sonar	130–160 Hz ('low' component) and 260–320 Hz ('high' component)	120–150*	Playbacks had 42 every 6–10 minutes	No clear trends in distribution or vocalizations before, second transmissions during and after transmissions
Frankel & Clark (1998)	Field	<i>Megaptera novaeangliae</i>	ATOC	75 Hz (centre frequency)	≤ 130	10–80 m depth, 100–2000 m from source	Whales dived longer and travelled farther between dives
	Field	<i>Megaptera novaeangliae</i>	ATOC	75 Hz (centre frequency)	≤ 130	8000–12 000 m from source	Dive time and distance covered between dives increased with estimated RL
Au <i>et al.</i> (1997)	Laboratory	<i>Pseudorca crassidens</i>	ATOC, pure tone	75 Hz (centre frequency)	<i>142 \pm 2 (pure tone) and 141 \pm 1 (ATOC; hearing thresholds)</i>		Sound would only be audible directly above source at 400 m depth
Au <i>et al.</i> (1997)	Laboratory	<i>Grampus griseus</i>	ATOC, pure tone	75 Hz (centre frequency)	<i>139 \pm 1 (ATOC; hearing thresholds)</i>		Sound would only be audible directly above source at 400 m depth
Nowacek <i>et al.</i> (2001)	Field	<i>Eubalaena glacialis</i>	Synthetic alert signal	1000 Hz	148***	Alerting signal, with pure tones, a downsweep and amplitude modulated tones	5 of 6 swam to the surface and remained ~5 m below the surface

Included are only studies that reported measured or modelled RLs. We have summarized each study for accessibility, though we direct the reader to the text for more in-depth discussion of individual studies. Units for RLs are: *rms; **peak-peak; ***re 1 μ Pa/sgrt(Hz). Studies discussing responses of cetaceans to anthropogenic sound that did not report RL are included in the literature library included as an appendix to our paper.

RL, received level; TTS, temporary threshold shifts; PTS, permanent threshold shifts; ATOC, Acoustic Thermometry of the Ocean Climate; rms, root mean square.

abnormally long dives) (Green, 1998). Au & Green (2000) did not report details of the behavioural data collected, only that the 'strongest reaction' was in response to the loudest boat. Due to the sound field created by the singing whales, only outboards and SWATH vessels had RL higher than whale chorus when the vessels passed the receiver at 91 m (100 yards, the approach distance restriction in Hawaii).

Au & Green (2000) concluded that because vessel source levels were lower than levels produced by singing whales, vessel noise would not damage hearing. The robust data that Au & Green (2000) provide for vessel noise signatures are very helpful, but drawing conclusions about the lack of threat to hearing loss based on vessels being quieter than singing whales is premature. The authors draw a false dichotomy between sounds produced by boats and those made by whales. They state that whales approach singers without apparent hearing damage, but if a whale spent sufficient time at close range to a singer, the threats to hearing would be the same as if they were exposed to another source such as vessel noise. The authors are correct, however, when they state that the ramifications of documented behavioural responses are open to assessment, and marine mammal scientists as a community are currently grappling with that very question (NRC, 2005).

Ice-breaking

Erbe & Farmer (2000b) also used the acoustic exposure model developed by Erbe & Farmer (2000a) to investigate the impacts of ice-breaker noise on beluga whales *Delphinapterus leucas*. Bubbler system and propeller cavitation noises from an ice-breaker ship were examined separately, and the model suggested that ice-breakers were audible to beluga whales at maximum distances of 35–78 km, depending on location and propagation conditions. Based on previous studies of beluga call recognition that showed the importance of the low-frequency portions of the calls (Erbe & Farmer, 1998; Erbe, 2000), intraspecific calls were considered to be masked if the levels of the two lower-frequency peaks in the band of interest of the call were below the noise. Zones of masking were substantially smaller than zones of audibility (Table 1). The authors used a disturbance threshold of 81 dB re 1 μ Pa for the 1/3 octave band level centred at 5 kHz; this threshold was based on previous work reviewed by Richardson *et al.* (1995). Using this threshold, disturbance zones were only slightly smaller than zones of audibility. Caution should be used when interpreting these 'disturbance thresholds' given the lack of information and review of the criterion. Moreover, we discuss elsewhere the broader problem of using a single dB level to 'decide' that disturbance will occur. Zones of potential hearing damage were estimated using data from bottlenose dolphin and pinniped species, and TTSs of 4.8 and 12–18 dB were predicted from the model. A TTS of 12–18 dB would result if belugas stayed within 40 m of bubbler noise or within 120 m of ramming noise for more than 30 minutes, which the authors note is unlikely due to the high mobility of these animals. However, the model predicts that a TTS of 4.8 dB would occur if animals were within 1–2 km of bubbler noise or within 2–4 km of ramming noise for 20 minutes.

Marine construction: drilling, dredging and explosions

Marine construction activities, explosions in particular, have significant potential to produce physical damage (Ketten, 1995) as well as behavioural disturbance. We include responses to any construction-related stimuli in this section because little work has been reported on the subject since 1995, and the majority of these report on explosions. As coastal construction will likely continue for many years, the effects on species using these areas should be explored.

Madsen & Møhl (2000) reported that sperm whales *Physeter macrocephalus* showed no behavioural or acoustic responses to distant detonations resulting in RLs of ≤ 179 dB rms

re 1 μPa . While their conclusion of no acoustic disturbance was supported and based upon a sample of eight animals, they reported only a single behavioural observation. The authors, while providing caveats for their small sample sizes, attempt to draw conclusions based upon their findings. They cite the similarity in power spectra between the distant detonations and sperm whale clicks as a possible explanation for the lack of response, inferring that the clicks of conspecifics have caused habituation to this type of signal. Demonstrating habituation requires careful experimentation (see Seyfarth & Cheney, 1990), so it is impossible to prove interpretations about habituation from eight opportunistically observed whales.

Todd *et al.* (1996) studied humpback whale movements and behaviour in relation to blasts occurring during the development of a concrete oil production platform in Trinity Bay, Newfoundland. RLs of sound from Tovex charges varied with the size of charge detonated but were measured, at a distance of 1.8 km, to be 140–153 dB re 1 μPa rms with the peak amplitude occurring at \sim 400 Hz. The authors stated that the techniques used to document potential response(s) were designed to provide equal sampling of areas less than and greater than 10 km from the source, but the effort was not otherwise quantified as a function of range from the source. Short-term behavioural parameters of whales were apparently collected on an *ad lib* basis, and the authors reported no detectable changes in respiration rates nor occurrence of abrupt surface reactions around the time of blasts, though *ab lib* sampling would not necessarily produce data sufficient to detect changes. There were no reported differences in re-sighting rates or distance to the site of the explosions between blasting and non-blasting periods during the observation period. Most of the whale sightings during this study occurred between 3 and 9 km from the blast site, and individual animals within 10 km of the blast site were sighted significantly more often than animals in greater range categories: 10–20 and 20+ km. Residency times (the mean number of days a whale was resighted) were longest closest to the blast site, though again the effort with various range categories was not reported. No other received sound characteristics were reported, though the authors did measure the temperature profile of the water column, which would permit modelling of the sound propagation. Collecting even limited data on the physical environment, as these authors did, is useful and relatively easy. The authors suggested that the lack of response to the blasts could be attributed to high prey abundance, and while they mention field observations showed abundances of prey throughout and outside the study area, no quantitative estimates were provided. Also, to account for the lack of response, they suggested that the whales may have habituated to the blasts since the study occurred midway through the blast schedule. Associated with this same construction activity, Borggaard, Lien & Stevick (1999) studied humpback whale abundance, distribution and movements in relation to blasting, dredging and vessel traffic in Trinity Bay over a longer period (1992–95), though the authors did not report any information about the levels or characteristics of sound received by the whales.

Finneran *et al.* (2000) conducted a series of experiments with two captive bottlenose dolphins and one beluga whale *Delphinopterus leucas* to investigate the behavioural and auditory responses to sounds that simulated distant underwater explosions. This experiment involved only one exposure to an intense sound per day. Finneran *et al.* (2000) observed no auditory shift (i.e. TTS) greater than 6 dB to levels up to 221 dB re 1 μPa p-p (peak-peak). Behavioural ‘alterations’ were observed at 196 and 209 dB re 1 μPa p-p for the two dolphins, respectively, and at 220 dB re 1 μPa p-p for the beluga, and these alterations continued at higher levels (Table 1). These behavioural alterations included delaying approach to the test station and avoiding the ‘start’ station. The authors appropriately provided two caveats to

their study: (i) the signals they used were for truly distant signals, i.e. signals from nearby explosions would differ in both level and structure; and (ii) they measured masked-hearing thresholds. Masked-hearing thresholds are measured in the presence of introduced background noise, in this case to mask the high levels of background noise in the test pens. Studies of TTS in humans ([Humes, 1980](#)) indicate that the presence of masking noise increases hearing thresholds, simulating a pre-exposure loss in hearing sensitivity, and decreases the amount of observed TTS. [Finneran *et al.* \(2000\)](#) discussed this issue and acknowledged that larger threshold shifts might have been measured in the absence of the masking noise. Perhaps even more important than this masking noise was the behavioural context of the test. These trained animals were tested in a context where they were being rewarded for tolerating high levels of noise. Given this training, it is likely that behavioural disruption would likely be observed at lower levels in other contexts.

The US Navy periodically tests the integrity of new classes of ships with underwater explosions. The latest of these 'ship shock trials' occurred in 2001 when the Navy tested the USS *Winston S. Churchill* off the coast of north-eastern Florida. Extensive aerial, boat-based, and passive acoustic monitoring was undertaken prior to detonations in an attempt to clear a safety zone, and again subsequent to explosions to monitor for any marine mammals injured or killed by the blasts ([Clarke & Norman, 2005](#)). The mitigation protocol called for detonations to be postponed upon any evidence of mammals or turtles in the safety zone, which occurred three times, but the protocol did not include monitoring either behaviour or RL for animals outside the zone. We have included this work here to highlight the need to monitor these activities, which did occur, but also to call attention to how 'safety zones' are defined. In this case, the choice of the safety zone size was intended to prevent serious injury or mortality, with a buffer added to account for animal movement towards the zone itself. In the case of these ship shock trials, there were only three explosions, so the behavioural effects of animals outside the injury/mortality zone may be of less concern. In the case of repeated detonations (e.g. excavations), however, the effects on animals outside the injury/mortality zone should also be monitored.

Only two other studies have examined marine construction-related activities since [Richardson *et al.* \(1995\)](#). [Schick & Urban \(2000\)](#) examined bowhead whale distribution around a drilling rig in the Alaskan Beaufort Sea during 1992 and 1993, and though they estimated an 'audible range', they reported no RL information. [Koschinski *et al.* \(2003\)](#) observed reactions of harbour porpoises and harbour seals *Phoca vitulina* to the broadcast of wind generator noise, and reported source levels but did not report any information on RL. [Madsen *et al.* \(2006a\)](#) reviewed a broader array of studies on responses of marine mammals to sounds of windmill construction and operation.

Marine geophysical surveys

Seismic exploration can produce short duration broadband impulse sounds with high peak source levels (220–255 dB re 1 μ Pa peak at 1 m). Airgun arrays used for geophysical exploration are designed to direct low-frequency (<300 Hz) sound downwards to maximize energy reflected back from strata under the seafloor. The sound levels in surrounding waters are less well understood, but considerable energy is introduced, including higher frequencies up to at least 15 kHz ([Goold & Fish, 1998](#); [Madsen *et al.*, 2006b](#)). [Richardson *et al.* \(1995\)](#) warned that studies of seismic effects on marine mammals are often not comparable because measures of source levels from seismic exploration are reported differently by various researchers. [Madsen \(2005\)](#) reviewed these different measures and made recommendations about which were most relevant for studying effects on marine mammals.

While there is higher frequency energy in the seismic pulses, the vocalizations and estimated hearing range of baleen whales overlap the highest peaks of the power spectrum of airgun sounds and consequently these animals may be more affected by this type of disturbance than toothed whales. Studies of baleen whales up to 1995 found that whales showed behavioural changes after exposure to this noise (Reeves, Ljungblad & Clarke, 1984; Richardson, Wursig & Greene, 1986; Ljungblad *et al.*, 1988).

McCauley *et al.* (2000) observed humpback whales exposed to seismic surveys, both as part of oil exploration surveys and controlled exposures using a smaller source. Humpbacks were observed during their southward migration past North-west Cape, near Exmouth, Western Australia. Using aerial surveys, individual focal follows, and acoustic measurements from the vessels and by remote sensors, the authors observed no displacement or deflections of the humpback's migration along the coast. The whales did, however, consistently execute course and speed changes to avoid close encounters with operating seismic arrays. Interestingly, despite the vast difference in the size of the commercial (2678 cu. in., 258 dB re 1 μ Pa peak at 1 m) and the experimental (20 cu. in., 227 dB re 1 μ Pa² p-p at 1 m) arrays, the whales' avoidance response occurred at similar estimated RLs of 160–170 dB re 1 μ Pa p-p. The whales avoided the commercial array at a distance approximately three times larger than the experimental array.

Also worth-noting were several observations of whales at the surface approaching an operating seismic array to within 100 m, then swimming quickly away by changing direction. McCauley *et al.* (2000) estimated that at a distance of 100 m from the air guns these whales were exposed to levels up to 192 dB re 1 μ Pa p-p, well above the levels at which avoidance responses were observed throughout their study. It is possible, even likely, that whales 100 m from the array at the surface would receive a far lower dose than one below the surface for two reasons. First, the arrays are constructed to direct sound energy downwards to maximize their exploratory capabilities, though the extent of this directionality has not been rigorously documented. The second reason for reduced sound levels near the surface is the pressure release, which causes the sound pressure energy to approach zero within a fraction of a wavelength of the surface (Urick, 1983).

While odontocetes were thought to be less sensitive than baleen whales to airgun sounds, since Richardson *et al.* (1995) there has been increased evidence of high-frequency energy from airguns and of responses of odontocetes to airgun signals. Goold (1996) studied the vocal reactions of common dolphins *Delphinus delphis* to seismic surveys in the Irish Sea. The two-dimensional seismic array had a peak source spectrum level of 205 dB re 1 μ Pa²/Hz at 200 Hz, and seismic surveys were conducted for 6 weeks in the middle of the 3-month observation period. Sound pressure spectrum levels 5 km from the seismic source at 80–100 m depth are reported in Table 1. The number of hours of acoustic contact from dolphin whistles and low-frequency (<22 kHz) components of echolocation clicks per hour of acoustic recordings were greater in the weeks prior to seismic activity than during seismic surveys, and were generally greater between seismic shotlines (i.e. when guns were off between lines) than when air guns were active. Post-seismic levels of dolphin vocalizations were greater than levels during seismic surveys, but were substantially lower than in weeks before seismic activity, which the author attributed to previously documented seasonal migration of dolphins offshore. During 1 week of observation during seismic activity, dolphin vocalization levels were similar during and between seismic pulses. Since dolphins were more than 1 km from the source during this time and were close to the source during other weeks of seismic activity, Goold (1996) suggested that animals may be more tolerant of seismic activity at distances over 1 km. While the use of acoustic contact as the metric for presence/absence of

dolphins appeared effective, [Goold \(1996\)](#) made no attempt to estimate the range at which dolphins could be detected, and this range may have differed during seismic and non-seismic periods.

[Goold & Fish \(1998\)](#) also studied effects of seismic survey air gun emissions on common dolphins. The power of seismic pulses from a 2120 cubic inch array between 200 Hz and 22 kHz was investigated at various distances. At 750 m from the source, spectrum levels ranged from approximately 140 dB re $1 \mu\text{Pa}^2/\text{Hz}$ at the 200 Hz end of the range to 90 dB re $1 \mu\text{Pa}^2/\text{Hz}$ at the 20 kHz end (Table 1). Acoustic monitoring of common dolphins during seismic surveys suggested a localized disturbance; only 4% of acoustic contacts occurred during air gun emissions, while 96% were observed when the guns were not in use. However, there were several flaws in the analyses and assumptions made during this study, as highlighted by [Ridgway *et al.* \(1999\)](#). [Goold & Fish \(1998\)](#) used data from October 1994 to compare with sound field data from March of 1996. Not only could the air gun emissions have been different during these periods, but the emissions were conducted during different seasons where different environmental conditions could have caused marked differences in sound propagation. Based on a single observation of dolphins which remained just over 1 km from the seismic ship for 2 hours, [Goold & Fish \(1998\)](#) concluded that dolphins were tolerant to the air gun array at approximately 1 km – or a power of 133 dB as inferred from the acoustic data from 1996. The authors then suggested that this level of noise would be ‘distressing’ to the dolphins. [Ridgway *et al.* \(1999\)](#) pointed out that since whistles of common dolphins can produce source levels of 170 dB at 1 m, a 133 dB level is likely frequently encountered by dolphins within 100 m of a whistling dolphin.

[Madsen *et al.* \(2002\)](#) observed foraging male sperm whales in northern Norway during exposure to distant (>20 km) seismic survey vessels. Because of multi-path propagation, each seismic pulse had several arrivals over a longer period of time than the duration of the initial pulse and had a –10 dB spectrum of 210–260 Hz. With an estimated maximum RL of 146 dB re $1 \mu\text{Pa}$ p-p, no behavioural or acoustic responses were observed.

Military sonar

There are two basic types of sonar: passive, which is used to listen to and receive sounds, and active sonar that detects objects by examining echoes of produced sounds. The latter can be of concern relative to marine mammals. Military sonars often produce intense sounds, with source levels above 210 dB re $1 \mu\text{Pa}$ at 1 m, and they range from low frequency (<1000 Hz) and mid-frequency (1–10 kHz) to high frequency (>10 kHz). [Richardson *et al.* \(1995\)](#) reviewed several studies of the effects of sonars on various species of marine mammals, including examples of human reactions to low-frequency military sonar to highlight the need for research in this area. Additionally, the response of marine mammals to non-military high-frequency sonar (e.g. ‘whale-finding sonar’) was reviewed previously by [Richardson *et al.* \(1995\)](#). Since their review, studies have examined effects of military sonar on marine mammal behaviour or have suggested possible connections between marine mammal strandings and mid-frequency military sonars. Investigations of vocalization rates in relation to sonar broadcasts indicate that some marine mammal species increase the rate of redundancy of vocalizations in the presence of sonar, possibly to compensate for interference ([Rendell & Gordon, 1999](#); [Miller *et al.*, 2000](#)). Spatial and temporal coincidence between marine mammal strandings and military tests of naval sonar indicate that military sonar may cause some whale species to strand ([Simmonds & Lopez-Jurado, 1991](#); [Frantzis, 1998](#); [Cox *et al.*, 2006](#)).

[Miller *et al.* \(2000\)](#) found that humpback whale songs were longer during playbacks of the US Navy’s SURTASS-LFA system (an array of 18 transducers towed between 60 and 180 m

depth, hereafter 'LFA', or Low Frequency Active sonar). Both 'low' and 'high' components of the signals, 130–160 Hz and 260–320 Hz, respectively, were broadcast during the experimental periods. Maximum RLs at the whales ranged between 130 and 150 dB re 1 μ Pa rms, and 42-second LFA signals were broadcast every 6 minutes (Table 1). During playbacks of LFA signals, whale songs were significantly longer than those recorded before and after playbacks. Themes within a song were longer but occurred in the same order before, during and after playbacks. Song length returned to pre-exposure levels after playbacks, indicating that this vocal response did not extend beyond the exposure duration. [Miller et al. \(2000\)](#) proposed that whales lengthened their songs to counter interference from sonar signals. Research conducted by [Fristrup et al. \(2003\)](#) in parallel with the [Miller et al. \(2000\)](#) study examined the effects of time of year, time of day and broadcast factors of the same LFA source on humpback song length. Rather than studying differences in song length before, during, and after LFA broadcasts, [Fristrup et al. \(2003\)](#) used a multivariate approach. Their work showed that in humpback whale songs overlapped by LFA pings, the latter part of the song was longer than songs that were not overlapped by pings. Unlike [Miller et al. \(2000\)](#), the [Fristrup et al. \(2003\)](#) study documented a delayed response to the LFA pings with songs showing effects up to 2 hours after the final ping.

[Croll et al. \(2001\)](#) examined the distribution, abundance and vocal behaviour of baleenopterid whales before, during and after broadcast periods of LFA sonar signals. As in the study on Hawaiian humpbacks, both 'low' and 'high' components of the LFA signals, 130–160 Hz and 260–320 Hz, respectively, were broadcast during the experimental period, and ship-based hydrophones measured most LFA sounds at between 120 and 150 dB re 1 μ Pa rms. Transmissions of LF sounds were intermittent (playbacks consisted of a 42 second transmission every 6–10 minutes and were conducted only during daylight hours) and were broadcast over a period of approximately 2 weeks. Whale sighting rates were too low for statistical comparisons but did not appear to be related to the sound field generated by the sonar; they appeared instead to have been linked to food abundance. Also, there were no clear trends in vocal rates in relation to the periods before, during and after LFA broadcasts. Photo-identification studies showed few re-sightings of individual blue whales and fin whales *Balaenoptera physalus* (for example, three of 23 fin whales were re-sighted), suggesting that these whales were transitory. In reference to the hypothesis put forward by [Richardson et al. \(1995\)](#) that marine mammals would not remain in areas of continuous noise of more than 140 dB re 1 μ Pa at sound frequencies relevant to a given species, [Croll et al. \(2001\)](#) noted that the sounds broadcast in their own study were intermittent and that whale presence in the area was transitory and therefore was not an ideal test of the [Richardson et al. \(1995\)](#) hypothesis.

Strandings and military sonars

Over the past decade, there has been growing evidence for a possible link between military sonar exercises and strandings of cetaceans, especially beaked whales of the genera *Ziphius* and *Mesoplodon*. [Simmonds & Lopez-Jurado \(1991\)](#) reported on several mass strandings of beaked whales in the Canary Islands that occurred during the 1980s when naval ships were present in the area. [Frantzis \(1998\)](#) reported an atypical mass stranding of Cuvier's beaked whales *Ziphius cavirostrus* that coincided with military 'sound-detecting system trials'. Over 2 days, 12 Cuvier's beaked whales stranded over a 38 km stretch of coastline in Greece with an average separation distance of 3.5 km. This mass stranding was atypical as animals involved in such events usually occur close together in time and space (i.e. they involve one

group of whales coming ashore together in the same location). [Frantzis \(1998\)](#) reported that these strandings of Cuvier's beaked whales were closely related in space and time with military testing of an acoustic sonar system for submarine detection. The sonars used in the military tests broadcast at 226–228 dB re 1 μ Pa in waveforms centred at frequencies of 600 and 3000 Hz.

Cox *et al.* (2006) provided a recent review of additional data on beaked whale strandings, including one in the Bahamas in 2000 that occurred during a naval exercise using mid-frequency sonars that operated at 2.6 or 3.3 kHz at source levels near 235 dB re 1 μ Pa rms at 1 m or at 6.8–8.2 kHz at source levels near 223 dB re 1 μ Pa rms. The Bahamas and Greek strandings were the only ones for which information on sound transmissions were available. While the sound fields were modelled in both cases, it is not known where the whales were before they stranded, so it is impossible to reconstruct the RLs of sonar sounds at the whales.

Jepson *et al.* (2003, 2005) and Fernández *et al.* (2004, 2005a) conducted necropsies of beaked whales from strandings that coincided with naval sonar exercises, and they reported the presence of gas or fat emboli, which they interpreted as consistent with a decompression-like syndrome. There has been scientific disagreement about these findings, which differ from reports of decompression in humans ([Piantadosi & Thalmann, 2004](#)). Reports of emboli stimulated additional interest in the Crum & Mao (1996) report that exposure of supersaturated tissue to intense sound *in vitro* could cause or enhance bubble growth.

Cox *et al.* (2006) reviewed various hypotheses that were proposed to explain the link between sonar exposure and atypical mass strandings. Intense sound can physically damage tissue, either directly or by bubble growth, but these direct physical effects require sound exposure levels (SELs) so intense that they occur only within 100 m or so of the source. It seems unlikely that the geographical pattern of atypical strandings coincident with sonar, spread over tens of kilometres of beach, could require whales to be within 100 m of the source. It has also been suggested that the proximate cause of death in these strandings may simply be the physical and thermal trauma of stranding itself. However, Fernández *et al.* (Fernández *et al.*, 2005b) reported that decomposed beaked whales washed up on beaches in the Canary Islands several days after an offshore naval exercise, suggesting that actual stranding was a prerequisite for lethal injuries. Furthermore, some of the pathologies described in live-stranded whales (Fernández *et al.*, 2004, 2005a) were not consistent with effects of stranding alone. Jepson *et al.* (2003) suggested that exposure to sonar might trigger a behavioural reaction at RLs below those required for physical injury, and that the behavioural reaction may cause decompression-like symptoms in addition to causing strandings. After reviewing all hypotheses proposed to date, Cox *et al.* (2006; p 177) concluded 'gas-bubble disease, induced in supersaturated tissue by a behavioural response to acoustic exposure, is a plausible pathologic mechanism for the morbidity and mortality seen in cetaceans associated with sonar exposure'.

Ocean science: ATOC/NPAL

At least nine studies have been undertaken to gauge potential effects of the Acoustic Thermometry of the Ocean Climate (ATOC) project, which aimed to acoustically measure the ocean's temperature (Munk, Worcester & Wunsch, 1995), though not all of these studies reported RL for their study subjects. The centre frequency of the ATOC signal is 75 Hz with a source level of 195 dB rms re 1 μ Pa at 1 m in the 60–90 Hz band. One of the ATOC signals was essentially a sine wave with a series of phase reversals, designed to enable precise measurement of signal arrival times. These phase reversals spread the spectrum of the signal across a small frequency band ([Munk *et al.*, 1995](#)).

Frankel & Clark (1998) measured responses of humpback whales to the playback of a scaled-amplitude version of the ATOC signal along the coast of Hawaii. This signal was identical in structure but was projected at 172 dB or -23 dB relative to the full ATOC source. Whales showed a detectable response to the experimental stimulus and undertook longer dives with more distance covered between surfacings during the exposure. Frankel & Clark (1998) concluded, however, that if the whales wanted to minimize their RL, they should have spent more time near the surface rather than at depth, but they were not able to measure the whales' dive depths during the experiment. The RLs in this study, estimated from a propagation model that the authors verified, ranged from below ambient to 130 dB re 1 μ Pa, averaged over depths of 10–80 m.

Results of a study conducted near Kauai in 1998 to examine behavioural responses of humpback whales to full-scale ATOC signals were similar to the 1996 study (Frankel & Clark, 2000). As in the study of scaled-amplitude ATOC playbacks, the time and distance travelled between successive surfacings increased with the estimated RL of sound, while there was no clear effect of ATOC transmissions on other behavioural parameters examined. The authors pointed out that the similarity of results between these studies, which were conducted in different seasons, indicated that ATOC signals produced consistent, though subtle, effects on humpback whale distribution. Though RLs from the sound source were similar in both studies (from <105 dB to 130 dB re 1 μ Pa rms), observations were conducted at different distances from the sound source (approximately 100–2000 m in 1996 vs. 8000–12 000 m in 1998). The similarity of the behavioural results, in spite of these differences in distance to the source, were noted as support for the idea that RL of sound was a better predictor of behavioural response than distance to the source. Frankel & Clark (2002) and Mobley (2005) investigated the effects of ATOC signals on humpback whale distribution and abundance near Kauai, but these reports contained no information about RL of the ATOC signals in the vicinity of the whales.

Au, Nachtigall & Pawloski (1997) conducted experiments on false killer whales *Pseudorca crassidens* and Risso's dolphins *Grampus griseus* to determine hearing sensitivity to both ATOC signals and 75 Hz pure tone signals. Hearing thresholds for the pure tone and ATOC signals were similar for both species (142 ± 2 dB and 141 ± 1 dB re 1 μ Pa, respectively, for Risso's dolphins and 141 ± 1 dB and 139 ± 1 dB re 1 μ Pa, respectively, for false killer whales). The authors then compared these sensitivities with a propagation loss profile of the ATOC source. They concluded from this analysis that Risso's dolphins and false killer whales swimming directly above the source would not hear ATOC signals unless they dived to 400 m. This study highlights the importance of relating signals to the audiogram of the species under study, when available.

Playbacks/controlled experiments using novel stimuli

Nowacek, Johnson & Tyack (2004) conducted CEEs with tagged North Atlantic right whales *Eubalaena glacialis*. The whales were tagged with a multi-sensor acoustic recording tag (Johnson & Tyack, 2003), which, along with the acoustic data, simultaneously recorded orientation and movement of the whales, water depth and temperature. The whales, while feeding at ~150 m, were exposed to vessel noise, whale social sounds, silence and a synthetic signal designed to alert the whales (Nowacek *et al.*, 2004). This alerting signal contained three parts including pure tones, a downsweep and amplitude modulated tones. In response to the alert signal, five of six exposed whales broke off their foraging dive and swam abnormally towards the surface taking a lower ascent angle than normal and significantly increasing their fluke stroke rate. These whales did not, however, stay at the surface but continued to swim at

~5 m depth, surfacing only to breathe; the sixth whale showed no detectable response compared with baseline behaviour and compared with the other exposures. The documented response, uniform across the five responding whales, was triggered at low RLs. The maximum level received by any of the five responding whales was 148 dB re 1 $\mu\text{Pa}/\sqrt{(\text{Hz})}$ at 1000 Hz, and these RLs were not significantly different than the vessel noise stimulus [Nowacek *et al.* \(2004\)](#) used in this experiment. [Nowacek *et al.* \(2004\)](#) concluded that alarm stimuli were a poor option in attempts to mitigate vessel collisions with whales, since the reaction of most animals in the study likely placed them at greater risk of vessel collision.

Synthetic acoustic signals, ADDs and AHDs

There have been several advances in our understanding of how high-frequency anthropogenic noise can effect marine mammals since the review by [Richardson *et al.* \(1995\)](#), driven primarily by novel applications of acoustic alarms in attempts to modify the behaviour of marine mammals for by-catch mitigation and deterrence. This includes documentation of noise effects on target species, primarily small cetaceans and seals, along with demonstrations of unwanted or unpredicted effects of noises on non-target species. Previous to [Richardson *et al.* \(1995\)](#), acoustic alarms had been used to deter humpbacks from interacting with cod traps in eastern Canada ([Lien, Todd & Guigne, 1990](#); [Lien *et al.*, 1992](#); [Todd, Lien & Verhulst, 1992](#)), and to some extent in attempts to keep seals from approaching certain areas ([Wickens *et al.*, 1992](#)). Much of this material was reviewed by [Reeves *et al.* \(1996\)](#).

Two classes of high-frequency acoustic alarms, ADDs and AHDs, have been identified ([Reeves *et al.*, 1996](#)), although these terms are often used interchangeably ([Johnston, 2002](#)). They can be differentiated by source level and the way in which they are deployed ([Reeves *et al.*, 1996](#); [Johnston & Woodley, 1998](#)). For example, ADDs, most commonly referred to as pingers, are generally low-power devices (<180 dB rms re 1 μPa at 1 m) deployed on moveable or transient gear, such as gillnets and set nets. In contrast, AHDs are generally high-power sound sources (>180 dB rms re 1 μPa at 1 m) deployed permanently on structures such as fish pens and dams. Other forms of underwater acoustic signals have been tested for effects on marine mammals, including signals aiming to alert whales or deter them from approaching moving ships (e.g. [Nowacek *et al.*, 2004](#)) and sounds produced through the use of underwater communications equipment ([Kastelein *et al.*, 2005](#)). In the latter case, acoustic devices form the communication backbone of a developing maritime safety data collection and communication network (referred to as ACME) in several European countries. These communication devices produce several types of sounds, including chirps and frequency sweeps. Details on these sounds are presented in Table 2. Increasing the source level of sounds produced by ACME sources elicited avoidance responses from captive harbour porpoises at RL of ≤ 116 dB rms re 1 μPa ([Kastelein *et al.*, 2005](#)). This study explicitly recognized the potential for habitat exclusion effects for porpoises and developed models of discomfort zones for animals exposed to ACME sounds to help guide further development of the network.

ADDs

The largest number of recent studies examining the effects of high-frequency anthropogenic noise on marine mammals relate to the use of ADDs or 'pingers' on nets to reduce by-catch of small cetaceans (Table 2). There are a number of devices being tested or in use, and several have been tested experimentally in a variety of conditions to assess their efficacy. Initial tests of ADDs with harbour porpoises were consistent with avoidance responses (e.g. [Lien *et al.*, 1995](#)) and the most widely cited demonstration was a sea trial experiment with ADDs in an

Table 2. Details on the types of acoustic deterrent devices (ADDs) employed in recent studies including signal characteristics, modelled or measured RLs, deployment methods and general results of the study

Study	Field, laboratory or modelling	Species	Sound source	Signal characteristics	Received (dB re 1 μ Pa rms at 1 m) levels <i>modelled values in italics</i>	Deployment details	General results
Kastelein <i>et al.</i> (2005)	Laboratory	<i>Phocoena phocoena</i>	ACME underwater communications	8–16 kHz chirps, spread-spectrum blocks, frequency sweeps and modulated frequency shifts, 116–130 dB	Discomfort at ≤ 116	Deployed in enclosure with two male harbour porpoises	Avoidance of sound source as source levels increased
Kraus <i>et al.</i> (1997)	Field	<i>Phocoena phocoena</i>	Dukane Netmark 1000	10 kHz pulses every 4 seconds; 132 dB	≥ 98 at the net*	ADDs deployed on actively fishing gillnets	Reduced by-catch, reduced catch of Atlantic herring (<i>Clupea harengus</i>)
Kastelein <i>et al.</i> (1997)	Laboratory	<i>Phocoena phocoena</i>	Loughborough signal generator	Clicks, sweeps and tones, 17.5–140 kHz	≤ 107	Deployed in tank with single female harbour porpoise	Avoidance of sound source
	Laboratory	<i>Phocoena phocoena</i>	Memorial University ADD (MUN)	Tones, 2.5 kHz; 110–131 dB	≤ 107	Deployed in tank with single female harbour porpoise	Avoidance of sound source
	Laboratory	<i>Phocoena phocoena</i>	Scanmar netminder	110 kHz; 158 dB	≤ 107	Deployed in tank with single female harbour porpoise	Avoidance of sound source
	Laboratory	<i>Phocoena phocoena</i>	Tri-tech ROV scanning sonar	325 kHz; 179 dB	≤ 107	Deployed in tank with single female harbour porpoise. ²⁴ ° horizontal beam angle, 4.5° vertical beam angle; sonar scanned across the pool at various angles	Avoidance of sound source
Kastelein <i>et al.</i> (2000)	Laboratory	<i>Phocoena phocoena</i>	Dukane Netmark 1000	10 kHz pulses every 4 seconds; 132 dB	≤ 124	Deployed in tank with harbour porpoises	Avoidance of sound source

Kastelein <i>et al.</i> (2000)	Laboratory	<i>Phocoena phocoena</i>	Dukane prototype	10 kHz pulses, randomized production; 132 dB	≤124	Deployed in tank with harbour porpoises	Avoidance of sound source
Kastelein <i>et al.</i> (2001)	Laboratory	<i>Phocoena phocoena</i>	Bird Alarm	Sweeps between 2 and 3.5 kHz; 100 dB	≤90 at 3.5 kHz	Deployed in tank with harbour porpoises	Avoidance of sound source
	Laboratory	<i>Phocoena phocoena</i>	Dukane XP-10	16 tones (constant pulse width and interval – 6% duty cycle) between 9 and 15 kHz; 145 dB	≤138 at 33 kHz	Deployed in tank with harbour porpoises	Avoidance of sound source, increased respiration rates
	Laboratory	<i>Phocoena phocoena</i>	Dukane 2MP	16 tones (randomized pulse width and interval – 8% duty cycle) between 9 and 15 kHz; 145 dB	≤140 at 12 kHz	Deployed in tank with harbour porpoises	Avoidance of sound source, increased respiration rates
	Laboratory	<i>Phocoena phocoena</i>	HS20-80	0.1-second upsweep and 0.2-second downsweep, 20–80 kHz; 96–118 dB; 4.6% duty cycle	≤90 at 65 kHz	Deployed in tank with harbour porpoises	Avoidance of sound source, increased respiration rates
Kastelein <i>et al.</i> (2006)	Laboratory	<i>Phocoena phocoena</i> , <i>Stenella coeruleoalba</i>	Dukane XP-10	16 tones (constant pulse width and interval) between 9 and 15 kHz; 145 dB	≤138 at 33 kHz	Deployed in tank with harbour porpoise and striped dolphin	Sound source avoided by <i>P. phocoena</i> , no reaction from <i>S. coeruleoalba</i>
Culik <i>et al.</i> (2001)	Field	<i>Phocoena phocoena</i>	PICE pinger	Sweeps between 20 and 169 kHz; 145 dB	102 at COA	Deployed on experimental net and during sea trials in active fishery	Avoidance of sound source, COA to active
Koschinski & Culik (1997)	Field	<i>Phocoena phocoena</i>	MUN	Tones, 2.5 kHz; 115 dB	72 at COA	Deployed during sea trials in active fishery	Avoidance of sound source, COA to active device = 130 m
Gearin <i>et al.</i> (2000)	Field	<i>Phocoena phocoena</i>	Custom pinger	Broadband with peaks at 3 and 20 kHz; 122–125 dB	≥90 at the net*	Deployed during sea trials in ctive fishery	Reduced by-catch device = 130 m

Table 2. (Continued)

Study	Field, laboratory or modelling	Species	Sound source	Signal characteristics	Received (dB re 1 μ Pa rms at 1 m) levels modelled values in italics	Deployment details	General results
Trippel <i>et al.</i> (1999)	Field	<i>Phocoena phocoena</i>	Dukane Netmark 1000	10 kHz pulses every 4 seconds; 132 dB	Detection range of 0.1–0.6 km for 80–90 dB RL	Deployed during sea trials in active fishery	Reduced by-catch
Carlstrom <i>et al.</i> (2002)	Field	<i>Phocoena phocoena</i>	Dukane Netmark 1000	10 kHz pulses every 4 seconds; 132 dB	≥ 98 at the net*	Deployed during sea trials in active fishery	No by-catch recorded
Barlow & Cameron (2003)	Field	Various pinnipeds and cetaceans	Dukane Netmark 1000	10 kHz pulses every 4 seconds; 132 dB	≥ 99 at the net*	Deployed during sea trials in active fishery	Reduced by-catch
Cox <i>et al.</i> (2001)	Field	<i>Phocoena phocoena</i>	Dukane Netmark 1000	10 kHz pulses every 4 seconds; 132 dB	118–122 dB (ambient noise levels) at 125 m	Deployed individually on mooring	Exclusion distance decreased by 50% after 4 days
Bordino <i>et al.</i> (2002)	Field	<i>Pontoporia blainvillei</i> , <i>Otaria flavescens</i>	Dukane Netmark 1000	10 kHz pulses every 4 seconds; 132 dB	≥ 104 at the net*	Deployed during sea trials in active fishery	Reduced by-catch and increased depredation
Cox <i>et al.</i> (2004)	Field	<i>Tursiops truncatus</i>	Dukane Netmark 1000	10 kHz pulses every 4 seconds; 132 dB	120 dB at approximately 100 m	Deployed during sea trials in active fishery	No difference in COA for active and inactive devices
Stone <i>et al.</i> (1997)	Field	<i>Cephalorhynchus hectori</i>	Dukane Netmark 1000	10 kHz pulses every 4 seconds; 132 dB	86 at median COA*	Deployed individually on mooring	Avoidance of sound source, COA = 552

*For field studies in which the goal was to measure by-catch rates, we modelled the RL based on the maximum possible distance between a pinger and an animal at the net and the relationship that $TL = 20 \log(r)$ where 'r' is the range in metres (Urick, 1983). This RL at the net is the operational sound level of interest, i.e. the RL at an animal at the net is the level at which the device must repel an animal to prevent entanglement. RL, received level; rms, root mean square; COA, closest observed approach.

active groundfish bottom-set gillnet fishery on the east coast of the USA in 1996 (Kraus *et al.*, 1997). This study clearly documented a statistically significant reduction in the number of porpoises by-caught in nets outfitted with active Dukane ADDs, which produced sound repeatedly every 4 seconds at a fundamental frequency of 10 kHz and a source level of 132 dB re 1 μ PA at 1 m. Interestingly, [Kraus *et al.* \(1997\)](#) also documented a reduction in the catch of Atlantic herring *Clupea harengus* – one of the primary prey of porpoises – in nets fitted with active ADDs, suggesting that fish behaviour might also have been altered by the sounds. These results confounded interpretations of how ADDs actually worked to reduce by-catch – for example, it was difficult to discern if porpoises avoided the nets or perhaps followed fish that were avoiding nets.

Some acoustic alarms have been tested with captive harbour porpoises and other cetaceans. For example, [Kastelein *et al.* \(1997\)](#) examined the effects of several candidate high-frequency sounds on a single female harbour porpoise with the aim of developing effective ADDs for fisheries interactions. Specifically, they examined the effects of nine different types of sounds on the surfacing position of this porpoise in a floating pen (the sounds were produced by five different sources and deployed as they would on nets in a gillnet fishery – see Table 2 for details on each signal type). The sound sources employed were two signal generators produced by Loughborough University, an ADD produced by Memorial University (MUN), a Scanmar 400 netminder system for assessing the geometry of trawl gear when deployed, and a scanning sonar system frequently used for navigating remotely operated vehicles. This latter device was used by scanning the pool across various angles. Porpoises reacted to all combinations of sounds produced by these sources by surfacing away from the source, with the greatest and most consistent reactions to broadband sweeps produced by the Loughborough signal generators and the high source level (SL) Scanmar netminder. More recently, [Kastelein *et al.* \(2000\)](#) tested how three different signals – two commercial ADDs (Standard Dukane Netmark 1000 and a prototype Dukane device producing a randomized signal with the same properties as the standard Dukane device) and a test signal referred to as a ‘bird alarm’ (see Table 2 for details on devices used in this study) altered the swimming patterns and respiration rates of captive porpoises. For all three treatments it was apparent that animals increased their respiration rates and maximized their distance from the active sound sources at surfacings, clearly displaying avoidance behaviour. The authors acknowledged the limitations of their study, in particular the problems extrapolating their findings to porpoises in the wild, as startle effects could not be eliminated in the captive scenario and these controlled exposures clearly did not mimic how animals would experience approaching a net with active pingers on it in a natural setting.

More recently, [Kastelein *et al.* \(2001\)](#) assessed the effects of signals with randomized production times on harbour porpoise surfacing positions and respiration rates. Three signals were used: the Dukane XP-10 that produced 16 tones with constant pulse width and interval selected in a random order, the Dukane 2MP producing similar sounds with pulse intervals randomized between 2 and 5 seconds. The third signal (HS20-80) consisted of a frequency sweep with a constant pulse duration (see Table 2 for further details on these devices and RL information). The porpoise avoided all three alarms and also exhibited increased respiration rates. The largest effects were found for trials using the XP-10 signal. Comparisons between the reactions of a captive harbour porpoise and a striped dolphin *Stenella coeruleoalba* to the XP-10 signal were recently conducted ([Kastelein *et al.*, 2006](#)). In this case, the harbour porpoise exhibited similar avoidance of the sound source and an increased respiration rate, whereas the striped dolphin showed no reaction to the alarm signal. Care must be taken in extrapolating results demonstrated with one animal to predict responses by other animals in

a population, especially when only one animal per species is tested. In the above cases, there is some evidence that porpoises in general may avoid these sounds, as similar responses were elicited from a number of porpoises. In the case of the striped dolphin, however, only one animal has been tested and this result may or may not represent how other animals would react in the wild.

Subsequent to the initial successful field tests, a growing number of studies documented the exclusionary effects of ADDs on small cetaceans, including observations of experimental deployments and further sea trials in active fisheries. For example, [Culik *et al.* \(2001\)](#) assessed the behaviour of porpoises in relation to one type of ADD as well as the catches of fish in relation to three different ADD designs in an active fishery. They tested the Dukane Netmark 1000 used in many other studies, as well as a PICE pinger and an MUN pinger (see Table 2) – also see [Lien *et al.* \(1995\)](#). They found that porpoises avoided nets fitted with PICE alarms and that catches of Atlantic herring were not affected by any of the ADDs used in the Baltic Sea herring fishery. The authors cautioned against widespread use of higher powered pingers. The mean closest observed approach (COA) to a single PICE pinger was 414 m, and large-scale fisheries using these devices could effectively exclude porpoises, and perhaps other non-target species sensitive to these sounds, from large portions of habitat. The MUN ADDs, which only produce at 115 dB re 1 μ Pa at 1 m, were previously tested with porpoises by [Koschinski & Culik \(1997\)](#). They found that animals were excluded on average from the area within a radius of ≥ 130 m of an active device, but RLs at this distance were not measured or modelled.

Other field studies employing ADDs in active fisheries revealed reduced by-catch of porpoises. In the US Pacific North-west, [Gearin *et al.* \(2000\)](#) deployed ADDs in a salmon gillnet fishery in northern Washington. The alarms produced a broadband signal with peaks at 3 and 20 kHz, with mean source levels between 122 and 125 dB re 1 μ Pa at 1 m. They observed significant reductions in porpoise by-catch in nets fitted with these ADDs, with no changes in catch rates between nets with ADDs and those without. Similarly, in the Bay of Fundy on the East Coast of Canada, [Trippel *et al.* \(1999\)](#) deployed ADDs (Dukane Netmark 1000) in demersal gillnets targeting groundfish in an attempt to reduce harbour porpoise by-catch. In two separate experiments, porpoise by-catch declined by an average of 77%, with limited changes in catches of fish (reduced catch of Atlantic pollock *Pollachius pollachius* in one season).

In European waters, [Carlstrom *et al.* \(2002\)](#) deployed pingers in a bottom-set gillnet fishery in the Swedish Skagerrak. In this study, fish catches did not vary between controls and treatment sets, and no porpoises were caught in either the controls or treatments – a significant deviation from the by-catch rates recorded for that fishery and region in previous years. A spatial analysis of concurrent control and treatment net sets revealed that a large portion of the study area (including control net sets) was likely exposed to the sounds of active pingers and that porpoises maybe have been excluded entirely from the local environs, resulting in the recorded zero by-catch rates. The authors concluded that while pingers may be effective in reducing by-catch through displacement of animals, care must be taken not to reduce or eliminate access to critical habitats. Dukane Netmark 1000 ADDs have also been deployed experimentally in drift gillnet fisheries off the west coast of the USA and successfully reduced total by-catch of all cetacean and pinniped species that interact with this fishery ([Barlow & Cameron, 2003](#)).

Most studies on the effects of acoustic alarms on small cetaceans are relatively brief and cannot address potential issues of habituation or sensitization over longer time frames. Exposed animals may exhibit a reduced response to the sound stimulus over time, thereby

reducing the efficacy of the acoustic deterrent through habituation. For example, Cox *et al.* (2001) found that non-captive harbour porpoises appear to habituate to Dukane Netmark 1000 pingers relatively rapidly – they documented that the initial exclusion distanced diminished by 50% in just 4 days to a Dukane Netmark 1000 ADD. Few other studies have addressed the important issue of habituation by marine mammals to high-frequency anthropogenic sounds, which provides an interesting contrast to situations where ADDs appear effective at reducing small cetacean by-catch for a number of years.

Acoustic deterrent devices have been tested on other species of cetaceans, with varying degrees of success. For example, an experimental study using Dukane Netmark 1000 ADDs to reduce by-catch of Franciscana *Pontoporia blainvillei* in Argentina found that the acoustic devices reduced by-catch rates but increased depredation rates of nets fitted with ADDs by South American sea lions *Otaria flavescens* – see [Bordino *et al.* \(2002\)](#). This result resembles the attractive ('dinner bell') effect described for early attempts to use sounds to eliminate depredation of fishing gear by pinnipeds ([Reeves *et al.*, 1996](#)). Bottlenose dolphins appear to be less influenced than porpoises by sounds produced by standard Dukane ADDs. For example, [Cox *et al.* \(2001\)](#) tracked bottlenose dolphins with a digital theodolite in relation to nets fitted with Dukane Netmark 1000 pingers. They found no difference in the COA of animals to the nets between controls and treatments, or in the number of animals observed in the study area when active ADDs were on the nets. However, fewer animals entered a buffer zone of 100 m around active sound sources compared with inactive controls. The authors cautioned against extrapolating these latter results further and indicated that pingers were not appropriate for by-catch mitigation with this species as they appeared aware of the nets and that little avoidance of alarms occurred. These results may be consistent with the lack of avoidance exhibited by striped dolphins exposed to the XP-10 signals described above – see [Kastelein *et al.* \(2006\)](#). ADDs have also been tested with Hector's dolphins *Cephalorhynchus hectori* in New Zealand. In that study, [Stone *et al.* \(1997\)](#) used a coastal observation point overlooking an experimental mooring with active and inactive pingers deployed from it. Active and inactive trials were conducted in a blind manner and observers tracked the movements of dolphins in relation to the mooring with a theodolite. The results suggested that Hector's dolphins avoided the area during pinger active periods, with a mean distance from the mooring during active trials of 552 m in comparison with 299 m during inactive periods. RLs of sound at these distances were not measured or modelled. Because of their similarities to phocoenids in terms of echolocation and hearing, cephalorhynchid dolphins may be more likely than other delphinids to avoid alarms designed to work on harbour porpoises.

AHDs

Advances in technology during the 1990s led to the development of AHDs with louder source levels at frequencies known to be within the most sensitive hearing range of pinnipeds (and most odontocetes). In some cases these devices produced a single tone in a repetitive manner, while in other cases they are programmed to produce sound at a number of frequencies in a random fashion; this was an attempt to reduce habituation of target species to the stimulus (see [Johnston & Woodley, 1998](#); [Terhune, Hoover & Jacobs, 2002](#) for details of two forms of AHDs currently used in North America). The signal characteristics, transmitted and received, and a brief description of the animals' responses are summarized in Table 3. The effectiveness of using AHDs to deter seals from approaching salmon cages in the Bay of Fundy was assessed by [Jacobs & Terhune \(2002\)](#). They found that harbour seals approached within 45 m of an active Airmar AHD (apparently operating at a reduced SL of 172 dB re

Table 3. Details on the types of acoustic harassment devices (AHDs) employed in recent studies including signal characteristics, deployment mode, modelled or measured RLS and the general result of the study

Study	Field, laboratory or modelling	Species	Sound source	Signal characteristics	RLs (dB re 1 uPa at 1 m) modelled values in italics	Deployment details	General results
Johnston & Woodley (1998)	Field	<i>Phocoena phocoena</i>	Various	180–200 dB	<i>122 at max range of influence*</i>	Assessed extent of AHD use on salmon farms in lower Bay of Fundy	Large percentage of sites using AHDs, possible habitat exclusion
Terhune et al. (2002)	Field	<i>Phoca vitulina</i> <i>Phocoena phocoena</i>	Airmar, Ferranti Thompson 4X special	195 and 166 dB, respectively, 10–19 kHz	95 dB at 2.92 km for Airmar; 94 dB at 1.3 km for Ferranti Thompson	AHDs deployed experimentally from small boat or on active salmon farms	NA
Jacobs & Terhune (2002)	Field	<i>Phoca vitulina</i> , <i>Phocoena phocoena</i>	Airmar	172 dB	158–164 dB at approximately 45 m	AHDs deployed on active salmon farms	Seals avoided sound source, COA = 45 m
Olesiuk et al. (2002)	Field	<i>Phocoena phocoena</i>		180 dB	<i>≤134 at 200 m exclusion zone*</i>	AHDs deployed on active salmon farms	Porpoises avoided sound source – none observed within 200 m)
Johnston (2002)	Field	<i>Phocoena phocoena</i>	Airmar	180 dB	<i>125 dB at mean COA 991 m*</i>	AHD deployed on mooring	Porpoise avoided sound source, COA to active AHD = 645 m
Taylor et al. (1997)	Modelling	<i>Phocoena phocoena</i>	Various	180–200 dB	<i>>130 dB at 1 km for 200 dB source</i>	Modelled various zones of acoustic influence	AHDs may exclude non-target species from important habitats

*For field studies in which the goal was to measure by-catch rates, we modelled the RL based on the maximum possible distance between a pinger and an animal at the net and the relationship that $TL = 20 \log(r)$ where 'r' is the range in metres (Urick, 1983). This RL at the net is the operational sound level of interest, i.e. the RL at an animal at the net is the level at which the device must repel an animal to prevent entanglement. RL, received level; NA, not available; COA, closest observed approach.

1 μPa at 1 m, 8 dB down from 180 dB SL of these devices of reported elsewhere – see Table 3 for details on this device) and reported RL at this distance between approximately 158 and 164 dB re 1 μPa at 1 m from over nine different recording sites. They witnessed several seals in the vicinity of active AHDs at salmon farms and concluded that local seals had habituated to the sound sources or, less likely, that the observed seals were hearing impaired. Detection distances by seals and porpoises for two types of AHDs were calculated by Terhune *et al.* (2002). They found that an extraordinarily loud AHD, transmitting at 195 dB re 1 μPa rms, would be detected by harbour porpoises at 2.92 km (Table 3).

Many aquaculture sites using AHDs to reduce depredation by one species are located in areas frequented by a variety of other species that may be sensitive to the sounds produced by the AHDs. This includes a number of marine mammals (primarily odontocete cetaceans) and some species of fish. There is growing evidence that AHDs can have significant effects on non-target species, and may actually exclude them from potentially important habitat. For example, concerns have been raised over the potential exclusionary effect of AHDs on harbour porpoises (Johnston & Woodley, 1998). This is of particular concern where intensive salmon farming activities overlap spatially and temporally with areas that are thought to be important habitat for marine mammals and where salmon farming activities overlap with the ranges of marine mammal species considered to be at risk, as in western Canada where the ranges of killer whales (which are currently listed as threatened by the Committee on the Status of Endangered Wildlife in Canada) are known to overlap with salmon farming regions (Morton & Symonds, 2002).

In attempts to address some of these concerns, the effects of AHDs on harbour porpoises have been tested experimentally to determine if they are likely to avoid areas influenced by AHDs and, if so, the distances at which they may be excluded from active AHDs. Data were available from two of these studies: one conducted in the waters of British Columbia in Retreat Passage (Olesiuk *et al.*, 2002), the other conducted off Grand Manan Island in the lower Bay of Fundy (Johnston, 2002). Both of these experiments were conducted with similar sound sources (Airmar AHD – see Table 3 for details).

Taylor, Johnston & Verboom (1997) calculated theoretical zones of influence of various AHDs (see Table 2 for sound source details) for harbour porpoises, based on porpoise and human hearing models. These estimates may not have accurately reflected the actual distances to which porpoises may perceive or be affected by AHDs, and they clearly did not account for variation in the motivational state of the animals which may greatly affect the way they respond to such stimuli. However, these theoretical models may help interpret exclusion distances determined experimentally. Taylor *et al.* (1997) produced estimates for the ‘zone of audibility’, the ‘zone of severe disturbance and discomfort’ and the ‘zone of hearing damage and injury’ for three underwater acoustic alarms. Depending on the source level and spectrum of the device, it was estimated that AHDs might be audible to porpoises up to distances as great as 12 km. The greatest ‘zone of severe disturbance and discomfort’ (estimated RL >130 dB re 1 μPa) was predicted to be within 1 km of the most powerful AHD considered (200 dB re 1 μPa at 1 m, broadband).

Olesiuk *et al.* (2002) provided data on the effects of the Airmar AHD, described above, on the relative abundance of porpoises at various distances from the device as assessed by visual scans of an area. No porpoises were observed in scans within 200 m of an active AHD, and only 8.1% of the expected porpoise abundance was observed in scans within 3.5 km of the active AHD. They also concluded that the effect of the AHD on porpoises was pronounced and highly significant, and likely extended further than their greatest observation distance (3.5 km) (Olesiuk *et al.*, 2002).

Johnston (2002) provided COA and mean COA of porpoises (tracked from a cliff-top by a digital theodolite) to both an active and inactive AHD – the same type of device tested in Olesiuk *et al.* (2002) – as well as changes in porpoise relative abundance associated with the presence of an active AHD. Fewer porpoises were found in the study area during active AHD periods. Porpoises did not surface within 645 m (estimated RL of 128 dB re 1 μ Pa) of the active AHD, and on average were likely to be excluded from within 991 m (estimated RL of 125 dB re 1 μ Pa) of the device when it was on. These results were similar to those of Olesiuk *et al.* (2002), with both studies illustrating a significant decline in porpoise abundance within 1500 m of an active AHD. The COA of porpoises to the AHD (645 m) tested in Johnston (2002) was similar to the predicted ‘zone of severe disturbance and discomfort’ (estimated RL of 130 dB re 1 μ Pa or greater) calculated for similar AHDs in Taylor *et al.* (1997).

Other species also appeared to avoid areas influenced by the sounds produced by AHDs. For example, Morton & Symonds (2002) observed a precipitous decline in the number of killer whales in the Broughton Archipelago, Canada when Airmar AHDs were introduced into the area. When the devices were turned off, these animals resumed using the areas, supporting the hypothesis that they were avoiding areas influenced by AHD sounds. Similarly, Morton (2000) reported a decline in the local abundance of Pacific white-sided dolphins *Lagenorhynchus obliquidens* in the same region, concurrent with the deployment of AHDs. In both cases, however, no RL measurements were made or modelled, and limited data on actual exposure were presented.

PHYSIOLOGICAL RESPONSES TO SOUND IN CETACEANS

To date, measuring non-behavioural, physiological responses of cetaceans to sound has been limited. Heart rate has been measured by several studies (Fletcher *et al.*, 1996; Andrews *et al.*, 1997; Miksis *et al.*, 2001), though only with limited capacity or with captive animals. Measuring TTS has received greater attention in recent years. Finding the point at which this damage occurs is clearly important in setting an upper bound on acceptable exposure to noise. TTS has been measured by both behavioural means as well as by auditory evoked potentials, a well-developed technique that has been utilized with marine mammals since 1981 (Ridgway *et al.*, 1981) and has been refined and used with more species recently (Cook *et al.*, 2006). The study by Finneran *et al.* (2000) was discussed earlier as it also contained behavioural results. Several other studies have explored TTS exclusively, using different signal types. Another study exposed dolphins to broadband noise and measured TTS both behaviourally (Nachtigall, Pawloski & Au, 2003) and using auditory evoked potentials (Nachtigall *et al.*, 2004). Schlundt *et al.* (2000) and Finneran *et al.* (2005) measured TTS in response to intense tones. The onset of TTS varied somewhat between animals, with TTS onset at lower amplitudes at frequencies where the animals are most sensitive. Finneran *et al.* (2005) reported levels in SEL, a metric that is intended to capture the effects of prolonged exposure and so includes a time-based term and is reported in μ Pa²-s.

The hearing in the animals in these studies did recover, and we are unaware of any studies that have induced PTS. Finally, these final two studies reported some data on behavioural disturbance that resulted from the sound exposure trials. These studies were not designed to measure behavioural disturbance, and, as such, suffered from potential biases when reporting these results. For example, Finneran *et al.* (2005) reported ‘altered behaviour’ as a percentage of trials, but there is no information about the order in which the experiments that yielded this behaviour occurred, i.e. the order effects of presentation were not considered. Additionally, though not reported, an animal failing to return to the experimental station for additional exposures is usually ‘punished’ in some way, which can certainly affect the level(s) it is willing

to withstand. A study designed to measure behavioural thresholds would follow a different paradigm than one intended to investigate the physiological limits of the auditory system. TTS in response to noise sources that we discussed have also been measured in pinnipeds (e.g. [Kastak *et al.*, 1999](#)) and fishes ([Scholik & Yan, 2002](#); [Smith, Kane & Popper, 2004](#); [Popper *et al.*, 2005](#)), and these data are valuable as we need to include in this discussion all marine species that are potentially affected by anthropogenic noise.

Noise-induced stress has also received attention recently. [Romano *et al.* \(2004\)](#) examined plasma concentrations of catecholamines, aldosterone and lymphoid cell subsets as well as cortisol in a beluga whale and a bottlenose dolphin before and after exposure to impulsive sounds from a seismic water gun and/or single pure tones similar to sonar pings. Exposure to water gun impulses ranged from 198 to 226 dB re 1 μ Pa peak pressure, and tones from 130 to 201 dB re 1 μ Pa. Detrimental changes in the various parameters (i.e. a decrease in some is considered negative effects) increased with increasing sound levels and reached highly significant levels at exposures that induced TTS. The authors noted that studies investigating impacts of noise as a potential 'stressor' are difficult to interpret; stress is a difficult concept to define, and differences between studies, such as the type, intensity and duration of sounds used, make it difficult to compare results between studies. We agree with this assessment while realizing that noise can result in significant stress-related physiological problems, as have been measured in terrestrial mammals (e.g. [Harlow *et al.*, 1987](#); [Krebs *et al.*, 1997](#)).

CONCLUSIONS

A great deal has been learned since [Richardson *et al.*'s \(1995\)](#) review of underwater sound and marine mammals, and significant insights into the effects of anthropogenic sound on cetaceans have been gained. Some positive uses of sound in reducing by-catch have been found, thereby improving conservation of cetaceans. However, a growing number of studies have detailed the negative effects of marine anthropogenic sound on cetaceans, including incidental effects on non-target species in purposeful deployments as well as those occurring from uses of sound in the marine environment that are completely disconnected from cetacean biology and conservation.

In our review, we have discussed both acoustic and behavioural responses to noise exposure. The acoustic responses can be immediate or delayed, perhaps indicating a need to expand the normal 'pre' and 'post' exposure windows in order to avoid missing responses. One of the most common behavioural responses is displacement. To better assess the significance of displacements, though, it is necessary to know areas to which animals relocated, the quality of that habitat and the duration of the displacement if they indeed return to the pre-disturbance area. A short-term displacement from even an important location may not be of great concern, unless such a disturbance happens repeatedly. Furthermore, a long-term displacement may or may not be of concern, depending on the quality of available habitat. The critically endangered population of western gray whales *Eschrichtius robustus*, for example, can ill afford even a temporary displacement from its summer feeding grounds, where it is subject to potential impact from extensive oil and gas exploration ([Reeves *et al.*, 2005](#)). Overall, we advocate a continued increase in the sophistication of measuring and reporting of noise signals, animal responses and analysis of those responses, particularly in light of the recent report of a disconnect between short-term responses and long-term trends ([Bejder *et al.*, 2006a,b](#)).

Data gaps and uncertainties

Despite significant advances in our understanding, there remain fundamental uncertainties and data gaps regarding how anthropogenic sounds affect marine mammals. The two most

glaring data deficiencies are: (i) the lack of studies linking animal responses to RL data, and (ii) gaps in species representation. In addition, we are also deficient in our understanding of habituation, sensitization and tolerance. These phenomena have been explored in terrestrial and marine mammal species ([Espmark & Langvatn, 1985](#); [Conomy et al., 1998](#)), but they have been investigated only in the context of ADDs and in one behavioural study of cetaceans ([Cox et al., 2001, 2004](#)). [Richardson et al. \(1995\)](#) recognized the importance of their investigation; yet, very few studies have even attempted to address habituation, sensitization or tolerance. We recognize that such studies are difficult, particularly with large whales or those that are difficult to maintain in captivity; yet, the importance of understanding these phenomena has certainly not diminished.

Similarly, there is a need for studies to differentiate between what may be harmless, short-term responses to a sound from responses that could have an adverse impact on populations or species. NRC (2005) argues that if the primary goal of policy is to conserve marine mammal populations, behavioural criteria should try to link acoustic exposure and harassment with population-level effects. NRC (2005) provided a model for approaching this problem, called Population Consequences of Acoustic Disturbance or PCAD, but data for critical steps in that model are almost completely lacking. Two recent studies by [Bejder et al. \(2006a,b\)](#) presented a solid initial foray into this area. These studies documented a decline in the abundance of a population of coastal dolphins exposed to increasing tourist boat traffic. However, in recent experiments, this group of dolphins actually responded less strongly to approaching boats than a nearby, less exposed dolphin population, indicating either some level of tolerance in the 'tourist affected' population or that the more sensitive animals departed. Though the source of documented disturbance was not solely an acoustic one, these studies suggest that pronounced short-term behavioural responses, or the lack thereof, must be interpreted carefully and with a longitudinal perspective.

Active acoustics in by-catch reduction and ship-strike avoidance are two of the most prominent examples, and while by-catch reductions have been realized, they are not uniform or consistent. For example, there are a number of studies that detail the effects of various noises on harbour porpoises in captivity; yet, most of these have not been further evaluated under field situations or in the presence of other species (target or otherwise). Most controlled field studies detail avoidance of ADDs by porpoises and provide documented exclusion distances; yet, only one study has touched upon how large scale deployments of ADDs in fisheries may affect the behaviour and movements of these animals in the wild ([Carlstrom et al., 2002](#)). This may be especially important in regions where extensive fishing effort employing ADDs overlaps with important feeding or breeding habitat. Such results suggest the critical need to minimize unwanted side effects, while maximizing desired results and monitoring effectiveness in every situation/species. The use of active acoustics must not be thought of as a panacea for all marine mammal fishery interactions. Indeed, further research is required to assess the viability of using ADDs and AHDs in fisheries applications, and ideally they should be considered on a case-by-case basis. In some situations it may be logistically and economically difficult to deploy and maintain acoustic devices. Moreover, relying on active acoustics to solve by-catch problems may be dangerous or counter-productive. For example, [Dawson, Read & Slooten \(1998\)](#) illustrated that the uncertainties associated with using ADDs (e.g. habituation, proper implementation and mechanism of deterrence) in attempts to conserve small populations of cetaceans may compromise the overall conservation efforts, and [Bordino et al. \(2002\)](#) illustrated that nets with sound sources on them still may suffer from the 'dinner bell' effect – possibly increasing harmful marine mammal fishery interactions. Furthermore, one of the best documented uses of active

acoustics that might be applied to ship-strike avoidance resulted in surfacing responses by right whales that would likely increase their risk of collision (Nowacek *et al.*, 2004). There is a clear need for careful monitoring in each application of active acoustics, particularly in light of our lack of knowledge of the mechanism and the lack of information regarding habituation, sensitization and tolerance. Additionally, the potential for effects on non-target species must be considered.

To account for potential effects of purposeful as well as incidental sound, the marine mammal bioacoustic community is currently developing criteria for acceptable exposure levels (B. L. Southall *et al.*, unpublished). While preventing exposure to levels that would cause hearing damage (PTS) is relatively straightforward if one assumes that TTS data can be extrapolated across species to estimate non-PTS exposure, determining a single acceptable level of disturbance or harassment for a whole species is more difficult because of variability in responsiveness. While PTS and TTS may have specific amplitudes associated with them, behavioural disturbance may be more or less likely to occur depending on potential variation in response as a function of differences between species, age-sex classes, the motivational state of the animals and effects of repeated exposures. As such, we caution against the use of a single number of dBs as an absolute threshold, but rather develop a risk function or dose : response curve that captures the variability in responsiveness.

Future directions

Experimental methods available for measuring responses to sound have increased in recent years, particularly with the advent of sophisticated tags (Johnson & Tyack, 2003; Goldbogen *et al.*, 2006). Tagging animals is only one means of collecting data on potential effects of sound, but regardless of the methodology, it is imperative to account for as many variables as possible and to strictly define the questions and therefore the data to be collected. As we found, many studies have attempted to document effects of different noise sources, though most of them did not even report the received sound characteristics, which is a highly significant variable. The propagation of sound in the ocean can be complex, but many tools exist for estimating the exposure at individual cetacean receivers. Controlling the source of the sound is one means of controlling the exposure, but then either measuring or modelling the RL is still important as propagation conditions change (Urlick, 1983). It is sometimes desirable to have the actual noise source, e.g. a ship, be the source of noise so as to have a realistic exposure, particularly when the source is difficult to reproduce. Even with this more 'natural' exposure, however, measuring or modelling RL is still an imperative.

Controlled exposure experiments (CEEs) have been used by many investigators, and these experiments allow for the maximum control of acoustic variables and, in many ways, can acquire the most robust data for linking exposure with response(s). Once CEEs have demonstrated causal links between sound exposure and responses, observational studies can be designed to evaluate the significance of such responses over a variety of temporal and spatial scales. It is only through a combination of long-term studies and controlled experiments, not necessarily just CEEs, that we can extrapolate to population-level effects with any confidence. The results of CEEs can also be used to inform observational studies, as a means to achieve larger sample sizes, and to validate experimental results with more realistic exposures.

Controlled exposure experiments are, however, time-consuming and expensive. Combining CEEs with observational experiments as performed by McCauley *et al.* (2000) represents a solid compromise between experimental control and sample size needs. A particularly powerful way to combine these methods involves first conducting CEEs with sensitive response measures to develop specific hypotheses regarding responses, and secondly to follow up with

observational studies to increase the sample size in order to define the relationship between acoustic dose and behavioural response, under realistic conditions.

Both experimental and observational approaches raise animal welfare concerns. CEEs do add additional exposure to noise for some subset of animals, but the exposure is controlled more carefully than in observational studies, limiting risk to the subjects. The types, levels and durations of such harassment, which are acceptable for understanding the effects of noise, need to be resolved. In the case of CEEs in which animals are followed and/or tagged, a few individuals receive intense attention for relatively short periods of time, on the order of hours to days. In observational experiments, the research efforts focused on a given individual may be less intense, but there are typically many more animals harassed, often for periods of similar duration.

Humans continue to introduce sound into the marine environment, both purposefully and as a by-product of human activities, with an extremely limited understanding of its effects on cetaceans. For example, we have no information on how commercial sonars, depth finders and fisheries acoustics gear may influence the distribution and behaviour of cetaceans. These devices are prevalent and many produce sound in the frequency ranges used by cetaceans, e.g. 38 kHz. For example, depth finders available at virtually every boating supply store use acoustic pulses to measure the depth of water under a boat, with many using frequencies audible to cetaceans. While the energy from these devices is generally directed downwards, the sheer number of them, particularly in coastal waters, makes them a concern. Moreover, while our understanding of effects has indeed deepened and broadened since the Richardson *et al.* (1995) review, significant research effort continues to be expended without collecting data we consider to be absolutely necessary for exploring the effects of noise, specifically acoustic exposure measured or predicted at the animal and quantifiable behavioural responses. Harassing individual cetaceans during experiments of any type should be minimized, but we find it unacceptable to carry out experiments without collecting data that are necessary to evaluate observed responses. Even if an effect is observed, how can management efforts be directed without basic information such as exposure levels? This would be akin to reporting responses of human patients to drug trials without measuring or reporting the dose. Discovering how cetaceans perceive and respond to anthropogenic noise sources is important for their conservation, and maximizing the information available for exposures will maximize our ability to predict the scope of responses.

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Supplementary material

The following supplementary material is available for this article.

Appendix S1. Literature considered in the preparation of the review

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